# DRAFT U.S. PACIFIC MARINE MAMMAL STOCK ASSESSMENTS: 2021

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Stock assessment reports and appendices revised in 2021 are highlighted; all others can be found at the NOAA marine mammal stock assessment homepage.

**PINNIPEDS**



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



**CETACEANS – HAWAII & 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 National Marine Mammal Laboratory (NMML, Seattle, WA), and the Northwest Fisheries Science Center (NWFSC, Seattle, WA). The 2021 Draft Pacific marine mammal stock assessments include revised reports for 22 Pacific marine mammal stocks under NMFS jurisdiction, including 6 "strategic" stocks: Hawaiian monk seal, Southern Resident killer whale, California/Oregon/Washington humpback whale, Eastern North Pacific blue whale, California/Oregon/Washington fin whale, and Main Hawaiian Islands Insular false killer whale. New abundance estimates are available for 19 stocks: Northern elephant seal, Hawaiian monk seal, Southern Resident killer whale, 4 stocks of U.S. West Coast harbor porpoise, Dall's porpoise, Pacific white-sided dolphin, Common bottlenose dolphin, Striped dolphin, Short-beaked common dolphin, Long-beaked common dolphin, Northern right whale dolphin, Baird's beaked whale, California/Oregon/Washington stocks of humpback, fin and minke whales, and Eastern North Pacific blue whale. New information on human-caused sources of mortality and serious injury is included for those stocks where new data are available or resulted in a significant change compared with previously-documented levels of anthropogenic mortality and injury. 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 (2016). The authors solicit any new information or comments which would improve future stock assessment reports. Draft versions of the 2021 stock assessment reports were reviewed by the Pacific Scientific Review Group (PSRG) at the March 2021 online 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:

NOAA. 2016. Guidelines for Preparing Stock Assessment Reports Pursuant to the 1994 amendments to the MMPA. https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks

Revised ~~7/31/2015~~ 08/07/2021

# 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.

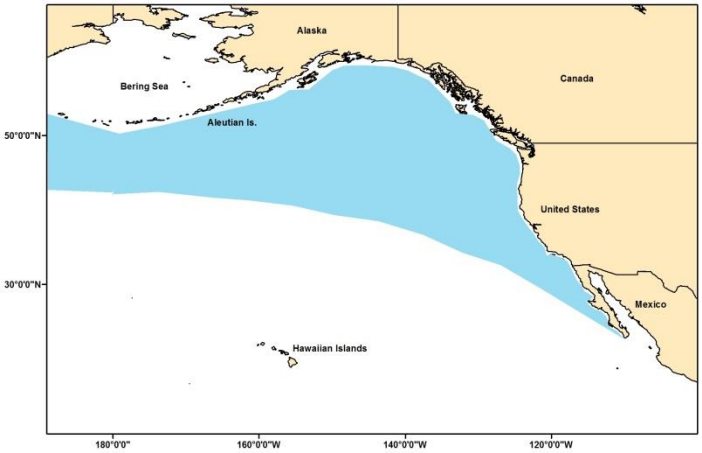


**Figure 1.** Pelagic range of northern elephant seals in the eastern North Pacific. Major breeding rookeries occur along the west coast of Baja California and the California coast, as described in Lowry *et al*. (2014).

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 ~~2010~~2013, Lowry *et al*. (~~2014~~2020) reported ~~that 40,684~~ 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*. (2014) applied a multiplier of 4.4 to extrapolate from total pup counts to a population estimate of approximately 179,000 elephant seals. This multiplier is derived from life tables based on published elephant seal fecundity and survival rates, and reflects a population with~~ 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 ~~2010~~ 2013 can be estimated ~~very~~ conservatively as ~~81,368~~ 85,369 seals, which is equal to twice the ~~observed pup count~~ estimated statewide pup count (to account for the pups and their mothers).

**Current Population Trend**

The population is reported to have grown at ~~3.8~~ 3.1% annually since 1988 (Lowry *et al*. ~~2014~~ 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 ~~2010~~ 2013, the population is estimated to have grown ~~3.8~~ 3.1% annually (Lowry *et al*. ~~2014~~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 (~~81,368~~ 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 ~~4,882~~ 5,122 animals per year.

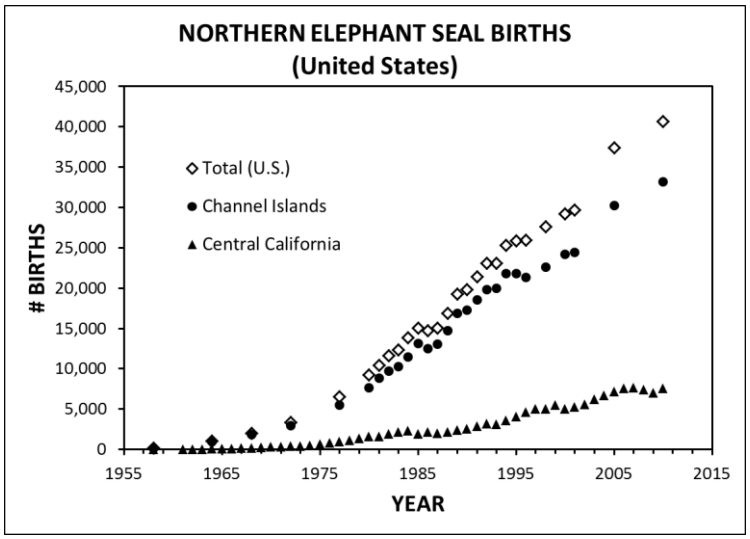


**Figure 2.** Estimated number of northern elephant seal births in California 1958-2010. Multiple independent estimates are presented for the Channel Islands 1988-91. Estimates are from Stewart *et al*. (1994), Lowry *et al*. (1996), Lowry (2002), Lowry *et al*. (2014), and unpublished data from Sarah 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**

~~**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*".~~

**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 ≥ 4.9 elephant seals annually. ~~More detailed information on these fisheries is provided in Appendix 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 ~~2008-2012~~ 2015-2019, ~~mortality~~ deaths and serious injuries from the following non-commercial fishery sources were documented: shootings (~~9~~ 2); marine debris entanglement (~~7~~ 4); hook and line fisheries (~~3~~ 2); ~~power plant entrainment (2); research-related~~ dog attack (1); unidentified human interaction (2); harassment (7); vehicle collision (1); tar/oil (~~1~~ 22); and vessel strike (1) (Carretta *et al*. ~~2014b~~

2021). These non-commerical fishery sources of mortality and serious injury total ~~24~~ 42 animals, or an average of ~~4.8~~ 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 and Enriquez 2009, 2010, 2012a, 2012b, Carretta *et al*. 2014a~~ Carretta et al. 2020a, 2020b, Jannot *et al*. 2018). n/a indicates information is not available. Mean annual takes are based on ~~2008-2012~~ 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** | ~~2008~~<br>~~2009~~<br>~~2010~~<br>~~2011~~<br>~~2012~~<br>2015-2019 | observer ~~data~~ | ~~13.5%~~<br>~~13.3%~~<br>~~11.9%~~<br>~~19.5%~~<br>~~18.6%~~<br>21% | ~~0~~<br>~~0~~<br>~~0~~<br>~~0~~<br>~~0~~<br>3 | ~~0~~<br>~~0~~<br>~~0~~<br>~~0~~<br>~~0~~<br>10.8 (0.41) | ~~0 (n/a)~~<br>2.2 (0.41) |
| **CA halibut and white seabass set gillnet fishery** | ~~2008~~<br>~~2009~~<br>~~2010~~<br>~~2011~~<br>~~2012~~<br>2017 | observer ~~data~~ | ~~0%~~<br>~~0%~~<br>~~12.5%~~<br>~~8.0%~~<br>~~5.5%~~<br>~~10%~~ | ~~n/a~~<br>~~n/a~~<br>~~0~~<br>~~0~~<br>~~0~~<br>0 | ~~n/a~~<br>~~n/a~~<br>~~0~~<br>~~0~~<br>~~0~~<br>0 | 0 (n/a) |
| ~~**CA small-mesh drift gillnet fishery for white seabass, yellowtail, barracuda, and tuna**~~ | ~~2010~~<br>~~2011~~<br>~~2012~~ | ~~observer data~~ | ~~0.7%~~<br>~~3.3%~~<br>~~4.6%~~ | ~~0~~<br>~~0~~<br>~~0~~ | ~~0~~<br>~~0~~<br>~~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)** | ~~2005~~<br>~~2006~~<br>~~2007~~<br>~~2008~~<br>~~2009~~<br>2012-2016 | observer data | 98% to 100% of tows in at-sea hake fishery<br>~~Generally less than 30% of landings observed in other groundfish sectors~~ | ~~0~~<br>~~1~~<br>~~3~~<br>~~7~~<br>~~2~~<br>2 | ~~0 (n/a)~~<br>~~1 (n/a)~~<br>~~3 (n/a)~~<br>~~9 (n/a)~~<br>~~2 (n/a)~~<br>2 | ~~3 (n/a)~~<br>0.4 (n/a) |
| ~~**Unknown gillnet fishery**~~ | ~~2008-2012~~ | ~~stranding~~ | ~~n/a~~ | ~~1~~ | ~~1 (n/a)~~ | ~~≥1~~ |
| **Total annual takes** | | | | | | ~~≥4.0 (n/a)~~<br>≥ 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. ~~Because their~~Total annual human-caused mortality (~~≥8.8~~

commercial fishery (5.3) + other sources (8.4) = 13.7) is ~~much~~ less than the calculated PBR for this stock (~~4,882~~ 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 ≥~~4.0~~ 5.3) ~~also appears to be~~ 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 ~~2010~~2013, the population grew at an annual rate of ~~3.8~~ 3.1% (Lowry *et al*. ~~2014~~ 2020). The population continues to grow, with ~~most~~ ~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*. ~~2014b~~ 2021).

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Revised ~~4/15/2021~~ 8/5/2021

# 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 ~~1,437~~1,437 (no change from 2020 SAR) (95% confidence interval ~~1,367~~1,369 ~~-~~– ~~1,549~~1,532); CV = 0.03), (Table 1, Johanos ~~2019a~~2021a,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). 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, Niihau 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 ~~2018~~2019, total enumeration was achieved at ~~Laysan and~~ Lisianski Island~~s,~~ and Kure Atoll, and at ~~capture-recapture estimate was obtained for~~ Midway Atoll, Laysan Island, ~~. At~~ French Frigate Shoals, and Pearl and Hermes Reef, abundance estimates were obtained using discovery curve analysis (Table 1). Counts at Necker and Nihoa Islands are 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. ~~In recent years, a~~A small number of surveys of Ni'ihau and nearby Lehua Islands ~~have been~~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 ~~recent~~ 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 ~~2018~~2019. 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 | ~~181~~188 | ~~40~~35 | ~~221~~223 | ~~181~~186 | 35~~40~~ | 221 | DC |
| Laysan | ~~200~~194 | ~~30~~40 | ~~230~~234 | ~~200~~193 | 40~~30~~ | ~~230~~233 | ~~EN~~DC |
| Lisianski | ~~130~~139 | ~~15~~19 | ~~145~~158 | ~~130~~139 | 19~~15~~ | ~~145~~158 | EN |
| Pearl and Hermes Reef | ~~124~~120 | ~~26~~21 | ~~150~~141 | ~~124~~120 | 21~~26~~ | ~~150~~141 | DC |
| Midway | ~~71~~70 | ~~13~~10 | ~~84~~80 | ~~65~~70 | 10~~13~~ | ~~78~~80 | ~~CR~~DC |
| Kure | ~~91~~81 | ~~11~~13 | ~~102~~94 | ~~91~~81 | 13~~11~~ | ~~102~~94 | EN |
| Necker | ~~76~~62 | ~~7~~8 | ~~83~~70 | ~~64~~52 | 8~~7~~ | ~~71~~60 | CC |
| Nihoa | ~~100~~72 | ~~6~~4 | ~~106~~76 | ~~85~~61 | 4~~6~~ | ~~91~~65 | CC |
| MHI_(without Ni'ihau/ Lehua) | ~~145~~161 | ~~30~~25 | ~~175~~186 | ~~145~~161 | 25~~30~~ | ~~175~~186 | Min |
| Ni'ihau/Lehua | ~~124~~138 | ~~7~~23 | ~~131~~161 | ~~104~~115 | 23~~7~~ | ~~111~~138 | CC |
| **Range-wide** | ***~~1252~~1239*** | ***~~185~~19 8*** | ***1437*** | ***~~1189~~1178*** | ***~~185~~19 8*** | ***~~1374~~13 76*** | |

**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 (~~1,374~~1,376) are presented in Table 1.

**Current Population Trend**

Range-wide abundance estimates are available from 2013 to ~~2018~~ 2019 (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-~~2018~~ 2019 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.~~00~~01, 1.~~04~~03). Thus, the best estimate is that the population grew at an average rate of about 2% per year from 2013 to ~~2018~~2019. Less than 1% of the distribution was below 1, indicating that there is greater than a 99% chance that the monk seal population increased during 2013- ~~2018~~2019.

**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 ~~2019a~~2021a). 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).

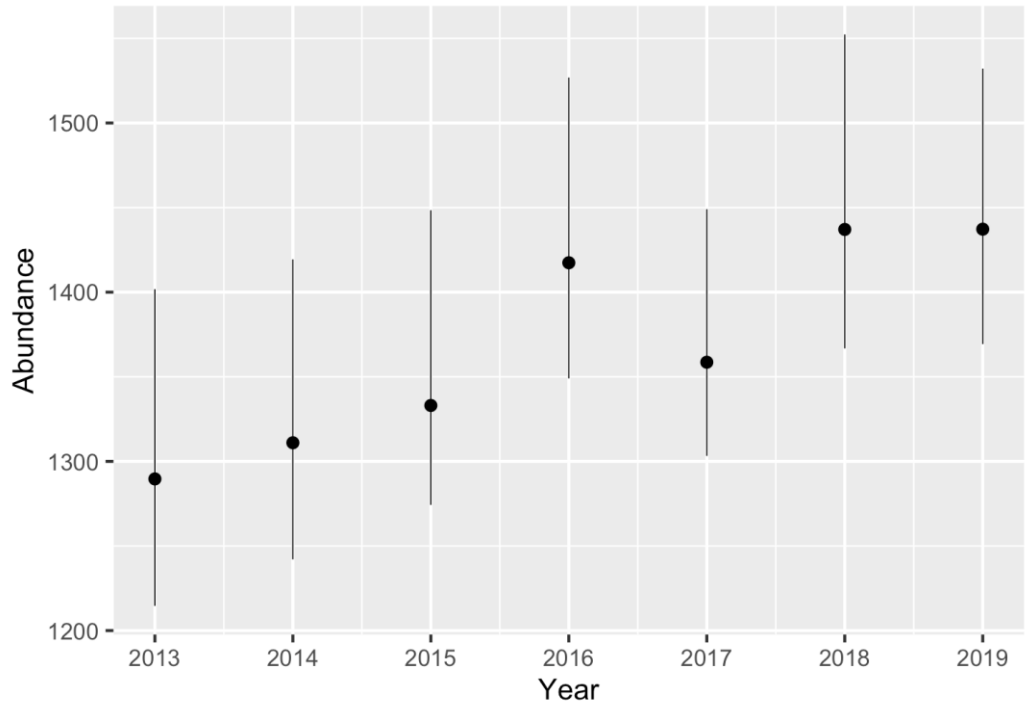


**Figure 1**. Range-wide abundance of Hawaiian monk seals, 2013-~~2018~~2019. Medians and 95% confidence limits are shown. Estimates prior to ~~2018~~ 2019 are re-estimated based on new data and represent negligible changes compared with values reported in the ~~final 2019~~previous final stock assessments.

**POTENTIAL BIOLOGICAL REMOVAL**

Using current minimum population size (~~1,374~~1,376), $R_{max}$ (0.07) and a recovery factor ($F_r$) for ESA endangered stocks (0.1), yields a Potential Biological Removal (PBR) of 4.8.

**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 20th century (Rice 1960), most subpopulations again declined. This second decline has not been fully explained, but long-term trends at several sites appear to have been driven both by variable oceanic productivity (represented by the Pacific Decadal Oscillation) and by human disturbance (Baker *et al*. 2012, Ragen 1999, Kenyon 1972, Gerrodette 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).

**Table 2**. Intentional and potentially intentional killings of MHI monk seals, and anthropogenic mortalities not associated with fishing gear during ~~2014~~2015-~~2018~~-2019 (Johanos ~~2019d~~2021d, Mercer ~~2020~~2021). There were no confirmed cases in 2016 nor 2019.

| Year | Age/sex | Island | Cause of Death | Comments |
|---|---|---|---|---|
| ~~2014~~ | ~~Adult male~~ | ~~Oahu~~ | ~~Suspected trauma~~ | ~~Intent unconfirmed~~ |
| ~~2014~~ | ~~Pup female~~ | ~~Kauai~~ | ~~Skull fracture, blunt force trauma~~ | ~~Likely intentional~~ |
| 2015 | Pup male | Kauai | Dog attack/bite wounds | 4 other seals injured during this event |
| 2015 | Juvenile male | Kauai | Probable boat strike | |
| 2015 | Adult male | Laysan | Research handling | Accidental, specific cause undetermined |

| | | | | |
|---|---|---|---|---|
| 2017 | Adult female | Kauai | Trauma | Suspect intentional |
| 2017 | Juvenile female | Molokai | Blunt force trauma | Suspect intentional |
| 2018 | Juvenile female | Molokai | Blunt force trauma | Intentional |

~~It is extremely unlikely that all carcasses of intentionally killed monk seals are discovered and reported.~~ 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. ~~Studies of the recovery rates of carcasses for other marine mammal species have shown that the probability of detecting and documenting most deaths (whether from human or natural causes) is quite low (Peltier *et al*. 2012; Williams *et al*. 2011; Perrin *et al*. 2011; Punt and Wade 2010).~~

**Fishery Information**

Fishery interactions with monk seals can include direct interaction with gear (hooking or entanglement), seal consumption of discarded 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 ~~2018~~2019, ~~ten~~ 17 seal hookings were documented, ~~two~~ one of which ~~were~~ was classified as serious and ~~eight~~ 16 as non-serious injuries. Of the non-serious injuries, ~~two~~ five would have been deemed serious had they not been mitigated (Henderson 2019a, Mercer ~~2020~~2021). The hooks involved included circle, treble and J-hooks of widely varying sizes. Two seals that had been previously scored as having serious injuries from hooking in 2018 have been reclassified as having non-serious injuries upon further review. Also, two mitigated serious injuries reported as fishery interactions in 2018 were reclassified as debris entanglements. Monk seals also interact with nearshore gillnets, and several confirmed deaths have resulted. ~~One confirmed gillnet mortality occurred in 2017, and three mortalities in 2016-2017 are considered suspect net mortalities (Mercer 2018), based on necropsy findings of probable peracute underwater entrapment (drowning) (Moore et al. 2013). In 2018, a fisherman reported releasing a monk seal from a gillnet he was tending off Oahu. The seal was reportedly lethargic but the event was deemed non-serious because the seal was released and subsequently has been resighted alive. Two seals were entangled in monofilament fishing gear on Oahu in 2018; both were deemed non-serious because they were disentangled, but would have been serious had they not been mitigated.~~ In 2019, the deaths of two seals were attributed to net drowning based on necropsy and other information. A third seal was suspected of having drowned in a net but the carcass was at sea and could not be recovered. No mortality or ~~serious~~ injuries have been attributed to the MHI bottomfish handline fishery (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. Percent observer coverage for the deep and shallow-set components, respectively, of the pelagic longline fishery, are shown. 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). ~~Data for MHI bottomfish and nearshore fisheries are based upon incidental observations (i.e., hooked seals and those entangled in active gear). All hookings not clearly attributable to either fishery with certainty were attributed to the bottomfish fishery, and hookings which resulted in injury of unknown severity were classified as serious.~~ 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 | % Obs. | Observed/Reported | Estimated | Non-serious | Mean |
|---|---|---|---|---|---|---|---|

| | | Type | coverage | Mortality/Serious Injury | Mortality/ Serious Injury | (Mitigated serious) | Takes (CV) |
|---|---|---|---|---|---|---|---|
| **Pelagic Longline** | ~~2014~~<br>2015<br>2016<br>2017<br>2018<br>2019 | ~~observer~~<br>observer<br>observer<br>observer<br>observer<br>observer | ~~20.8% & 100%~~<br>20.6% & 100%<br>20.1% & 100%<br>20.4% & 100%<br>20.4% & 100%<br>20.5 & 100% | ~~0~~<br>0<br>0<br>0<br>0<br>0 | ~~0~~<br>0<br>0<br>0<br>0<br>0 | ~~0~~<br>0<br>0<br>0<br>0<br>0 | 0 (0) |
| **MHI Bottomfish** | ~~2014~~<br>2015<br>2016<br>2017<br>2018<br>2019 | Incidental observations of seals | none | ~~0~~<br>0<br>0<br>0<br>0<br>0 | n/a | ~~0~~<br>0<br>0<br>0<br>0<br>0 | n/a |
| **Nearshore** | ~~2014~~<br>2015<br>2016<br>2017<br>2018<br>2019 | Incidental observations of seals | none | ~~1~~<br>3<br>0<br>3<br>~~2~~0<br>3 | n/a | ~~13 (9)~~<br>8 (2)<br>11 (6)<br>19(6)<br>11(3)<br>17(5) | ≥ 1.8 |
| **Mariculture** | ~~2014~~<br>2015<br>2016<br>2017<br>2018<br>2019 | Incidental Observation | none | ~~0~~<br>0<br>0<br>1<br>0<br>0 | n/a | ~~0~~<br>0<br>0<br>0<br>0<br>0 | 0.2 (2.2) |
| **Minimum total annual takes** | | | | | | | ≥ 2.0 |

**Fishery Mortality Rate**

Total fishery mortality and serious injury is not considered to be insignificant and approaching a rate of zero. Monk seals are ~~being~~ regularly hooked and entangled in the MHI ~~at a rate that has not been reliably assessed but is certainly greater than zero~~ and the resulting deaths have substantially reduced the population growth rate (Harting et al. 2021). ~~The information above represents only reported direct interactions, and without directed observation effort, the true interaction rate cannot be estimated.~~ 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 pursuing a program 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 ~~(one of which occurred at Kure Atoll in 2018)~~ (Henderson 2001; Henderson 2019b, Mercer ~~2020~~2021). 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. Nearly all documented cases would have been deemed serious had they not been mitigated by field biologists. 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) |
|---|---|---|
| ~~2014~~ | ~~0~~ | ~~5(4)~~ |
| 2015 | 0 | 12(8) |
| 2016 | 0 | 3(2) |
| 2017 | 0 | 11(8) |
| 2018 | 1 | ~~13~~15(6) |
| 2019 | 0 | 16(10) |
| **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 annual population assessment activities at the main reproductive sites. 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 (~~2014~~2015-~~2018~~2019) five monk seal deaths (representing a minimum average of one death per year) have been directly attributed to toxoplasmosis (Mercer ~~2020~~2021). All five 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 severely restricted. 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/parasitism. 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 introduced from livestock, feral animals, pets or other carrier wildlife could be catastrophic to the immunologically naïve monk seal population. Key disease threats include West Nile virus, morbillivirus and influenza.

**Habitat Issues**

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 monk seal field 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.

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.

## 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 (~~2014~~2015-~~2018~~2019) was at least ~~4.8~~4.6 animals~~, which equals PBR~~, including fishery-related mortality in nearshore gillnets, hook-and-line gear, and mariculture (≥ 2.0/yr, Table 3), intentional killings and other human-caused mortalities (≥ 1.2~~6~~/yr, Table 2), entanglement in marine debris (≥ 0.2/yr, Table 4), and deaths due to toxoplasmosis (≥ 1.0/yr). Because 4.6 is a minimum rate of annual human-caused mortality, the true value almost certainly exceeds PBR (4.8).

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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 ~~Monterey~~ Morro 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; 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.

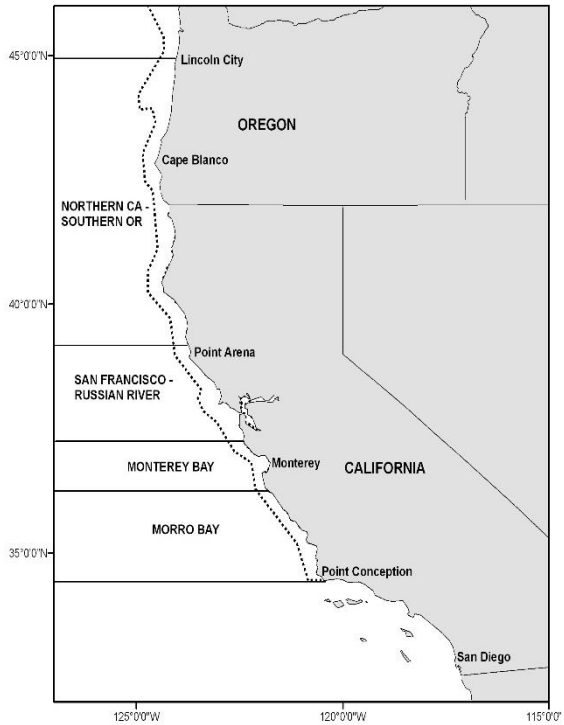


**Figure 1.** Stock boundaries and distributional range of harbor porpoise along the California and southern Oregon coasts. Dashed line represents harbor porpoise habitat (0-200 m) in this region.

~~In their assessment of harbor porpoise, Barlow and Hanan (1995) recommended that the animals inhabiting central California (defined to be from Point Conception to the Russian River) be treated as a separate stock. Their justifications for this were: 1) fishery mortality of harbor porpoise was limited to central California, 2) movement of individual animals appears to be restricted within California, and consequently 3) fishery mortality could cause the local depletion of harbor porpoise if central California is not managed separately. Although geographic structure exists along an almost continuous distribution of harbor porpoise from California to Alaska, stock boundaries are difficult to draw because any rigid line is (to a greater or lesser extent) arbitrary from a biological perspective. Nonetheless, failure to recognize geographic structure by defining management stocks can lead to depletion of local populations. Based on more recent genetic findings (Chivers *et al.*, 2002, 2007), California coast stocks were re-evaluated, and significant genetic differences were found among 4 identified sampling sites. Revised stock boundaries are presented here based on these genetic data and density discontinuities identified from aerial surveys, resulting in six~~

~~California/Oregon/Washington stocks where previously there had been four (Carretta *et al.* 2001a). The stock boundaries for animals that occur in California/southern Oregon waters are shown in Figure 1. For the Marine Mammal Protection Act (MMPA) Stock Assessment Reports, Pacific coast harbor porpoise stocks include: 1) a Monterey Bay stock, 2) a San Francisco-Russian River stock, 3) a northern California/southern Oregon stock, 4) a northern Oregon/Washington coast stock, 5) an Inland Washington stock, 6) a Southeast Alaska stock, 7) a Gulf of Alaska stock, and 8) a Bering Sea stock. Stock assessment reports for harbor porpoise stocks within waters of California, Oregon, and Washington appear in this volume. The three 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~~b~~). 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.* ~~2019~~2020) between 1986 and 2012 estimated a population size of ~~4,255~~4,191 (CV=~~0.562~~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 ~~2,737~~2,698 animals.

### Current Population Trend

A hierarchical Bayesian analysis of harbor porpoise trends between 1986 and 2012 (Forney *et al.* ~~2019~~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.

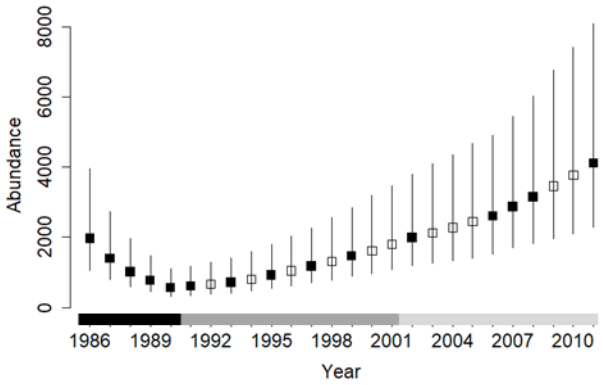


**Figure 2.** Population trends for the Morro Bay harbor porpoise stock, 1986-2012 (from Forney *et al.* 2020). Estimates represent median abundance (with 95% credible intervals) for years with survey effort (solid symbols) and without survey effort (open symbols). Shaded bars along the x-axis reflect 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 ~~9.7~~9.6% per year (95% credible interval: ~~6.4~~6.2% - ~~13.2~~13.0%) estimated by Forney *et al.* (~~2019~~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 ~~560-600~~571 porpoises when gillnet bycatch was reduced to low levels in 1991, and by 2012 the population had increased to ~~over 4,000~~an estimated 4,191 individuals.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (~~2,737~~2,698) times one half the estimated maximum net growth rate for this stock of harbor porpoise (½ of ~~9.7~~9.6%) times a recovery factor of 0.5 (for a stock of unknown status; Wade and Angliss 1997), resulting in a PBR of ~~66~~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 (~~2013-2017~~2015-2019), ~~one~~ no fishery-related strandings of harbor porpoise ~~was~~ were documented ~~south of~~within this stock's ~~primary~~ range (~~in 2013, Table 1,~~ Carretta *et al*. ~~2019~~2021). ~~The responsible fishery has not been identified.~~

**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 ~~2007-2011~~2015-2019 data, Carretta *et al*. (~~2019~~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 net fishery~~ Unknown fishery | ~~2013-2017~~ 2015-2019 | Stranding | n/a | ~~1~~none | n/a | ~~≥1~~ n/a | ~~≥ 0.2 (n/a)~~ 0 (n/a) |
| **Minimum total annual takes** | | | | | | | ~~≥ 0.2 (n/a)~~ 0 (n/a) |

~~**Other Mortality**~~

~~One harbor porpoise that was entangled in marine debris (a plastic bag) stranded in San Diego County and was attributed to the Morro Bay stock (Carretta *et al*. 2019), resulting in an average of ≥ 0.2 non-fishery, human-caused harbor porpoise deaths per year.~~

## 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*. ~~2019~~ 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 (~~≥ 0.4~~ zero harbor porpoise per year) is less than the PBR (~~66~~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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Revised ~~04/15/2020~~ 8/7/2021

# 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 ~~Monterey~~ Morro 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; 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.

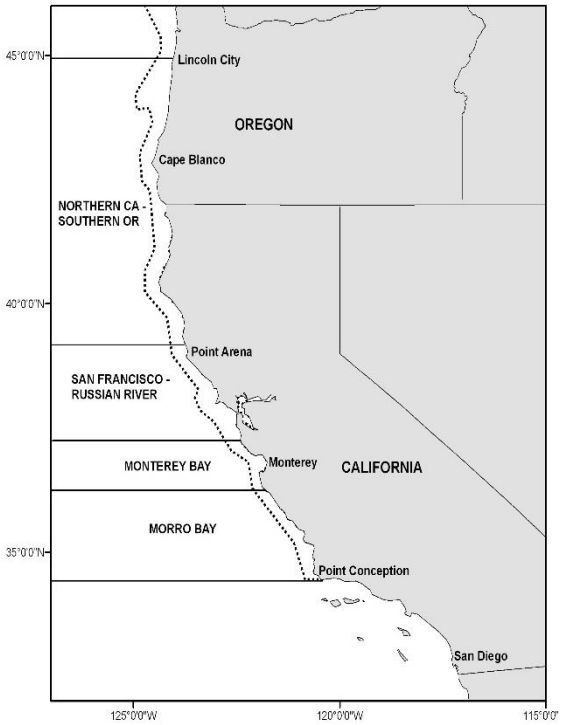


**Figure 1.** Stock boundaries and distributional range of harbor porpoise along the California/southern Oregon coast. Dashed line represents harbor porpoise habitat (0-200 m) along the U.S. west coast.

~~In their assessment of harbor porpoise, Barlow and Hanan (1995) recommended that the animals inhabiting central California (defined to be from Point Conception to the Russian River) be treated as a separate stock. Their justifications for this were: 1) fishery mortality of harbor porpoise was limited to central California, 2) movement of individual animals appears to be restricted within California, and consequently 3) fishery mortality could cause the local depletion of harbor porpoise if central California is not managed separately. Although geographic structure exists along an almost continuous distribution of harbor porpoise from California to Alaska, stock boundaries are difficult to draw because any rigid line is (to a greater or lesser extent) arbitrary from a biological perspective. Nonetheless, failure to recognize geographic structure by defining management stocks can lead to depletion of local populations. Based on more recent genetic findings (Chivers *et al*., 2002, 2007), California coast stocks were re-evaluated, and significant genetic differences were found among 4 identified sampling sites. Revised stock boundaries are presented here based on these genetic data and density discontinuities identified from aerial surveys, resulting in six~~

~~California/Oregon/Washington stocks where previously there had been four (Carretta *et al*. 2001a). The stock boundaries for animals that occur in California/southern Oregon waters are shown in Figure 1. For the Marine Mammal Protection Act (MMPA) Stock Assessment Reports, Pacific coast harbor porpoise stocks include: 1) a Morro Bay stock, 2) a San Francisco-Russian River stock, 3) a northern California/southern Oregon stock, 4) a northern Oregon/Washington coast stock, 5) an Inland Washington stock, 6) a Southeast Alaska stock, 7) a Gulf of Alaska stock, and 8) a Bering Sea stock. Stock assessment reports for harbor porpoise stocks within waters of California, Oregon, and Washington appear in this volume. The three 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~~b~~). 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*. ~~2019~~2020) between 1986 and 2013 estimated a population size of ~~3,455~~3,760 (CV=~~0.579~~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 ~~2,197~~ 2,421 animals.

**Current Population Trend**

A hierarchical Bayesian analysis of harbor porpoise trends between 1986 and 2013 (Forney *et al*. ~~2019~~2020) showed an increase in population size from a low of about 1,500 porpoises in 1987 to more than ~~3,400~~3,700 porpoises in 2013 (Forney *et al*. ~~2019~~2020). Most of this increase took place after gillnet fisheries were eliminated within the range of the Monterey Bay stock in ~~2003~~ 2002 (Figure 2).

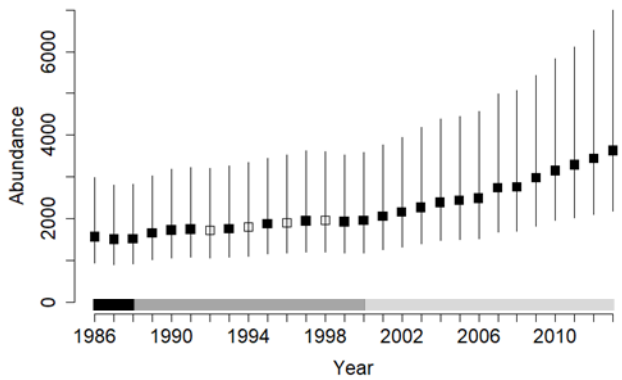


**Figure 2.** Population trends for the Monterey Bay harbor porpoise stock, 1986-2013 (from Forney *et al*. 2020). Estimates represent median abundance (with 95% credible intervals) for years with survey effort (solid symbols) and without survey effort (open symbols). Shaded bars along the x-axis reflect 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.*

(~~2019~~2020) estimated a growth rate of ~~4.2%~~5.8% per year (95% credible interval: ~~-1.9% - 6.3%~~ 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 (~~2,197~~2,421) times one half the maximum net growth rate estimated for this stock (½ of ~~4.2%~~5.8%) times a recovery factor of 0.50 (for a stock of unknown status; Wade and Angliss 1997), resulting in a PBR of ~~23~~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 (~~2013-2017~~2015-2019), one fishery-related stranding of harbor porpoise was documented within the range of the Monterey Bay stock (in 2015, ~~Table 1,~~ Carretta *et al.* ~~2019~~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 ~~2013-2017~~2015-2019 data, Carretta *et al.* (~~2019~~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.* (~~2019~~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 (~~23~~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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Revised ~~04/15/2020~~ 7/8/2021

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

## STOCK DEFINITION AND GEOGRAPHIC RANGE

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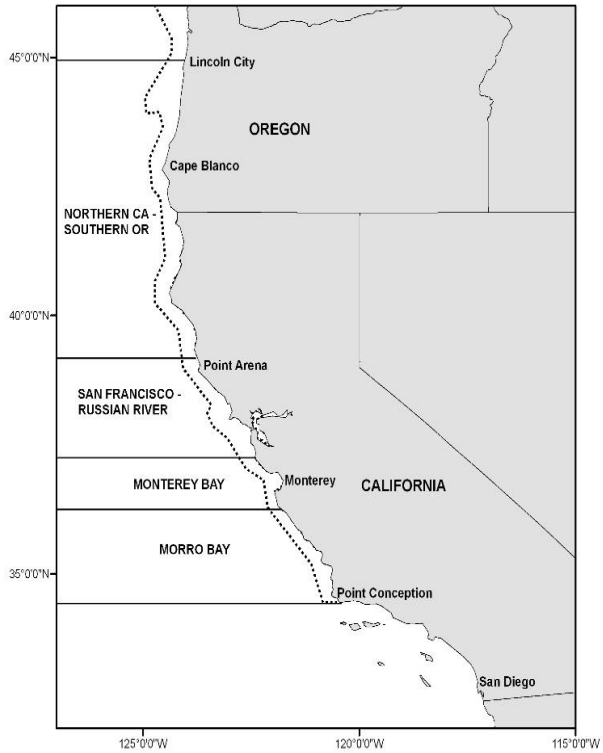


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**Minimum Population Estimate**

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 ~~4,801~~4,811 animals.

**Current Population Trend**

A hierarchical Bayesian analysis of harbor porpoise trends between 1986 and 2017 (Forney *et al.* ~~2019~~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 ~~14,500~~13,500 porpoises, and subsequently appeared to drop, leveling off at about 7,000-8,000 porpoises during 2010-2017 (Figure 2). ~~There are no known causes of this apparent decline, and~~ Forney *et al.* (~~2019~~2020) noted that the apparent decrease after 2005 could be artefact of the large uncertainty in the abundance estimates during 2002-2007. Alternately, ~~suggested that~~ a shift in the distribution of harbor porpoise in this region, including

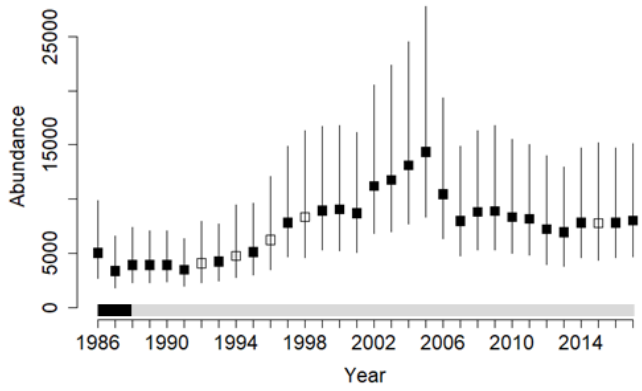


**Figure 2.** Population trends for the San Francisco-Russian River harbor porpoise stock, 1986-2017 (from Forney *et al.* 2020). Estimates represent median abundance (with 95% credible intervals) for years with survey effort (solid symbols) and without survey effort (open symbols). Shaded bars along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

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.

## 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.* (~~2019~~2020) estimated a growth rate of ~~2.4%~~6.1% per year (SE = 2.1%)~~95% credible interval: 0.1% – 4.8%)~~ 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.
~~, but this cannot be considered a maximum net productivity rate because it includes periods of apparent decline as well as increase. Because a reliable estimate of the maximum net productivity rate is not available for this 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 (~~4,801~~4,811) times one half the ~~default~~ maximum net growth rate ~~for cetaceans~~estimated for this stock (½ of ~~4%~~6.1%) times a recovery factor of 0.5 (for a stock of unknown status; Wade and Angliss 1997), resulting in a PBR of ~~48~~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 (~~2013-2017~~2015-2019), three fishery-related strandings of harbor porpoise were documented within the range of the San Francisco-Russian River stock (in ~~2013, 2014, and~~ 2015 and 2018; ~~Table 1,~~ Carretta *et al.* 2021~~19~~). Unidentified net fisheries were considered responsible for ~~all three~~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. ~~No fishery takes or fishery-related strandings were reported in this region between 2013 and 2017, Carretta *et al.* (2019).~~ 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** | ~~2013-2017~~ 2015-2019 | stranding | n/a | ~~3~~2 | n/a | ≥~~3~~ 2 (n/a) | ≥ ~~0.6~~0.4(n/a) |
| **Minimum total annual takes** | | | | | | | ≥ ~~0.6~~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.* (~~2019~~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 (≥ ~~0.6~~0.4 harbor porpoise per year) is less than the PBR (~~48~~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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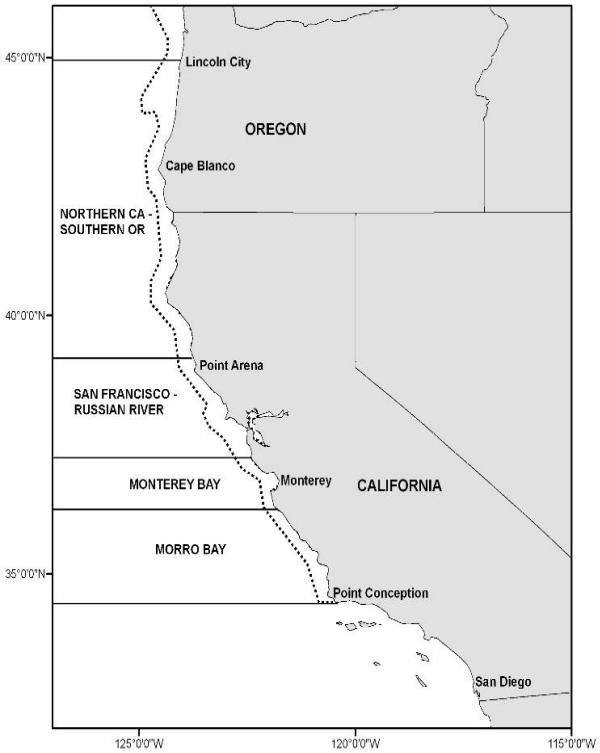
Revised ~~04/15/2020~~ 7/8/2021

# 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 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 ~~Monterey~~ Morro 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; 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.

~~In their assessment of harbor porpoise, Barlow and Hanan (1995) recommended that the animals inhabiting central California (defined to be from Point Conception to the Russian River) be treated as a separate stock. Their justifications for this were: 1) fishery mortality of harbor porpoise was limited to central California, 2) movement of individual animals appears to be restricted within California, and consequently 3) fishery mortality could cause the local depletion of harbor porpoise if central California is not managed separately. Although geographic structure exists along an almost continuous distribution of harbor porpoise from California to Alaska, stock boundaries are difficult to draw because any rigid line is (to a greater or lesser extent) arbitrary from a biological perspective. Nonetheless, failure to recognize geographic structure~~

~~by defining management stocks can lead to depletion of local populations. 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 were identified based on these genetic data and density discontinuities identified from aerial surveys (Figure 1). For the Marine Mammal Protection Act (MMPA) Stock Assessment Reports, Pacific coast harbor porpoise stocks include: 1) a Morro Bay stock, 2) a Monterey Bay stock, 3) a San Francisco-Russian River stock, 4) a northern Oregon/Washington coast stock, 5) an Inland Washington stock, 6) a Southeast Alaska stock, 7) a Gulf of Alaska stock, and 8) a Bering Sea stock. The stock assessment reports for harbor porpoise stocks within waters of California, Oregon, and Washington appear in this volume. The three 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).

A recent analysis of long-term trends in the northern California portion of this harbor porpoise stock between 1989 and 2016 (Forney *et al.* ~~2019~~2020) estimated a northern California population size of ~~11,670~~12,160 (CV=~~0.659~~0.663) porpoises during 2016. 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. The most recent estimate available for the entire northern California / southern Oregon stock is the sum of the 2016 California estimate of 12,160~~11,670~~ (Forney *et al.* ~~2019~~2020), plus the 2007-2011 southern Oregon estimate of 12,525 (CV = 0.48; Forney *et al.* 2014), totaling ~~24,195~~ 24,685 (CV = ~~0.40~~0.41).

**Minimum Population Estimate**

The minimum population estimate for harbor porpoise in northern California/southern Oregon is taken as the lower 20th percentile of the log-normal distribution of the abundance estimate given above, or ~~17,447~~17,713 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.* ~~2019~~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.

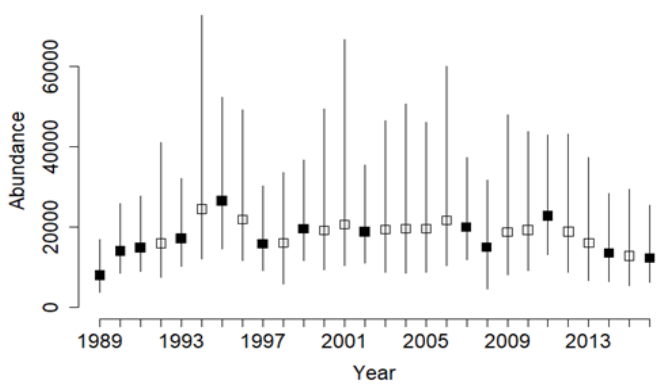



**Figure 2.** Population trends for the northern California portion of the Northern California / Southern Oregon harbor porpoise stock, 1986-2016 (from Forney *et al*. 2020). Estimates represent median abundance (with 95% credible intervals) for years with survey effort (solid symbols) and without survey 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 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. Because a reliable estimate of the maximum net productivity rate is not available for this 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 (17,713~~17,447~~) times one half the default maximum net growth rate for cetaceans (½ of 4%) times 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 ~~349~~354.

## HUMAN-CAUSED MORTALITY

### Fishery Information

There were no harbor porpoise strandings in this stock's range with evidence of fishery interactions during ~~2013-2017~~2015-2019 (Carretta *et al.* 2021).

**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 ~~2013-2017~~2015-2019 (Carretta *et al.* ~~2019~~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** | ~~2013-2017~~ 2015-2019 | Stranding | n/a | none | n/a | n/a | 0 (n/a) |
| | **Minimum total annual takes** | | | | | | 0 (n/a) |

~~**Other Mortality**~~

~~One harbor porpoise stranded with evidence of a fatal vessel strike during 2014 off Coos Bay, Oregon (Carretta *et al.* 2019), resulting in an average of ≥0.2 non-fishery, human-caused harbor porpoise deaths per year.~~

## 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 the known human-caused mortality or serious injury (~~≥ 0.2~~zero harbor porpoise per year) is less than the PBR (~~349~~354), 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 ~~2/9/2017~~ 8/7/2021

# 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~~ Ocean~~. Off the U.S. west coast, they are ~~commonly~~ seen in shelf, slope and offshore waters (Figure 1~~; Morejohn 1979~~). Sighting patterns from aerial and shipboard surveys ~~conducted~~ in California, Oregon and Washington waters (Green et al. 1992, 1993; Forney and Barlow 1998; Barlow 2016, Henry *et al.* 2020) suggest that ~~north-south~~ 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 ~~population's~~ stock's range is not well-documented, but they are ~~commonly~~ 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 ~~will emerge when data become available~~ may exist (Perrin and Brownell 1994). Although Dall's porpoises ~~are not restricted to~~ occur outside U.S. ~~territorial~~ 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.

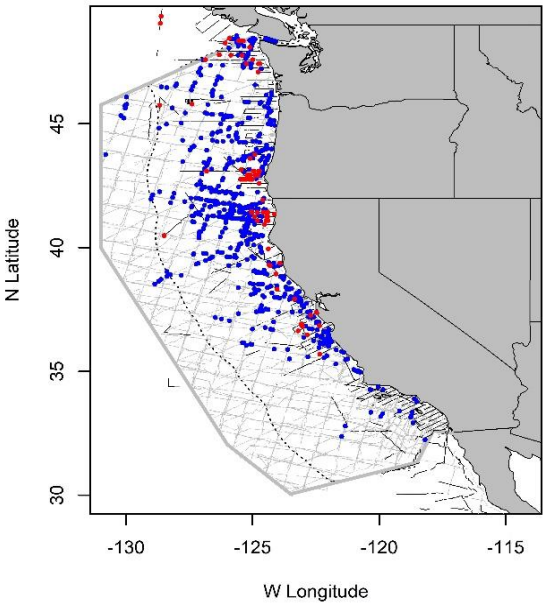


**Figure 1**. Dall's porpoise sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018. Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

## POPULATION SIZE

~~Dall's porpoise distribution in this region is highly variable between years and appears to be affected by 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 Dall's porpoise 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 25,750 (CV=0.45) animals (Barlow 2016). This estimate includes new correction factors for animals missed during the surveys.~~ 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 ~~2008-2014 average~~ 2018 abundance estimate for the outer coasts of California, Oregon and Washington waters is ~~17,954~~ 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 ~~considerably~~ 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. ~~but no longterm trends have been identified~~. 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.

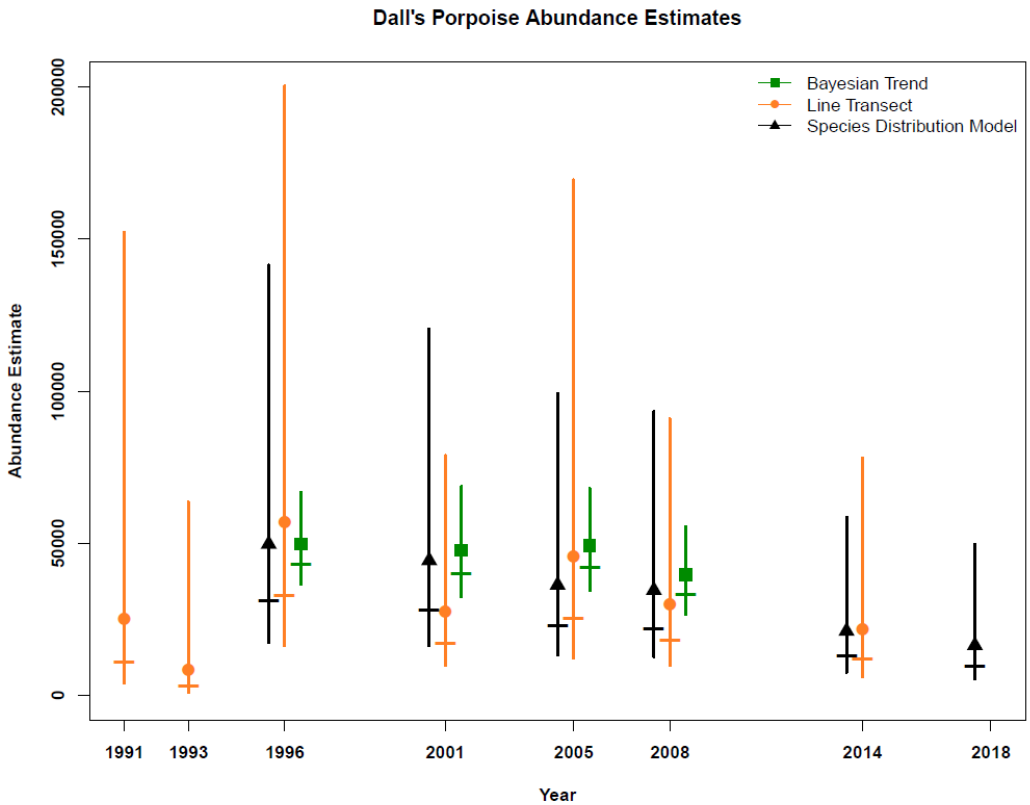


**Figure 2.** Abundance estimates and 95% confidence intervals from vessel-based line transect surveys (Barlow 2016), Bayesian trend models (Boyd *et al.* 2018), and species distribution models (Becker *et al*. 2020) within the California Current. Line-transect surveys in 1991 and 1993 did not include 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 (~~17,954~~ 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 ~~172~~ 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, ~~2010-2014~~ 2015-2019, averages ~~0.3~~ 0.46 animals per year (Carretta ~~*et al.* 2017~~ 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, ~~2009-2013~~ 2012-2016 (Jannot *et al*. ~~2011; NWFSC unpublished data~~ 2018). One animal was killed in an unidentified gillnet fishery in Washington state inland waters in 2016 (Carretta *et al*. 2021). ~~Gillnets have been documented to entangle marine mammals off Baja California (Sosa-Nishizaki et al. 1993), where Dall's porpoise may occasionally be found, but no recent bycatch data from Mexico are available.~~

**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 ~~*et al.* 2017~~ 2021, Carretta *et al.* 2021; Jannot *et al*. ~~2011~~ 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. ~~Mean annual takes are based on 2010-2014 data for the CA/OR swordfish drift gillnet fishery and 2005-2009 for groundfish fisheries.~~

| 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 | ~~2010~~<br>~~2011~~<br>~~2012~~<br>~~2013~~<br>~~2014~~<br>2015-2019 | ~~12%~~<br>~~20%~~<br>~~19%~~<br>~~37%~~<br>~~24%~~<br>21% | ~~0~~<br>~~0~~<br>~~0~~<br>~~0~~<br>~~1~~<br>0 | ~~0~~<br>~~0~~<br>~~0~~<br>~~0.2 (2.3)~~<br>~~1.1 (0.29)~~<br>2.3 (0.4) | ~~0.3 (0.53)~~<br>0.46 (0.4) |
| **WA, OR, CA domestic groundfish trawl fishery (includes at-sea hake and other limited-entry groundfish sectors)**<br><br>~~**WA/OR/CA groundfish (bottom trawl)**[a]~~ | observer | ~~2009-2013~~<br>2012-2016 | ~~23% (2009)~~<br>~~18% (2010)~~<br>~~100% (2011-2013)~~<br>98% - 100% | 0 | 0 | 0 |
| Unidentified gillnet fishery | Stranding | 2015-2019 | n/a | 1 | 1 | ≥ 0.2 |
| ~~WA/OR/CA groundfish (midwater trawl - at-sea hake sector)~~ | ~~observer~~ | ~~2009-2013~~ | ~~100%~~ | ~~0~~ | ~~0~~ | ~~0~~ |

| | | | | | | |
|---|---|---|---|---|---|---|
| ~~WA/OR/CA groundfish (midwater trawl - shoreside hake sector)[b]~~ | ~~observer~~ | ~~2011-2013~~ | ~~100%~~ | ~~0~~ | ~~0~~ | ~~0~~ |
| **Minimum total annual takes** | | | | | | ~~0.3 (0.53)~~ 0.66 (0.4) |

~~[a]The bottom trawl fishery was a limited entry fishery in 2010 and a catch shares fishery in 2011-2013.~~
~~[b]Fishery observers began monitoring the shoreside hake sector of the fishery in 2011.~~

## STATUS OF STOCK

The status of Dall's porpoises in California, Oregon and Washington relative to OSP is not known, and ~~there are insufficient data to evaluate potential~~ 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 (~~0.3~~ 0.66 animals) is estimated to be less than the PBR (~~172~~ 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 ~~2/9/2017~~ 8/7/2021

# 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 33ºN, (Southern California to Alaska), and southern specimens were obtained from about 36ºN 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).

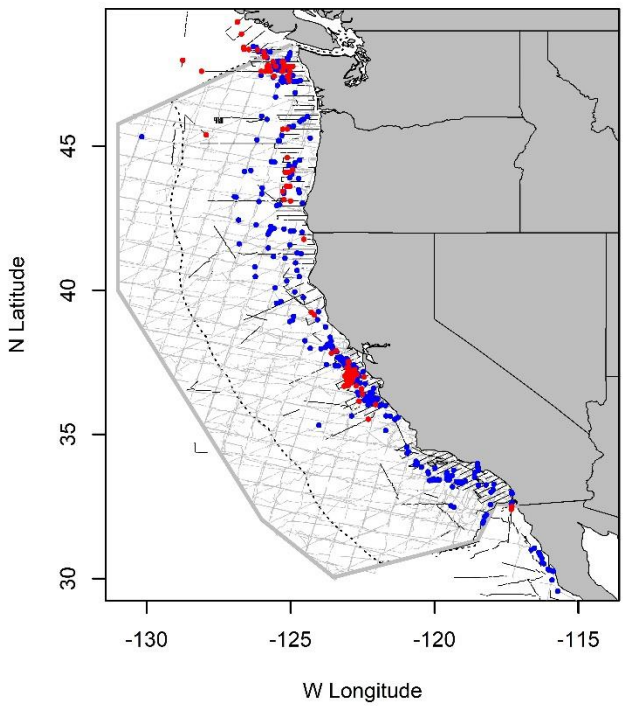


**Figure 1.** Pacific white-sided dolphin sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018. Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

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**

~~The distribution of Pacific white-sided dolphins throughout this region is highly variable, apparently in response to oceanographic changes on both seasonal and interannual time scales (Forney and Barlow 1998, Barlow 2016). As oceanographic conditions vary, Pacific white-sided dolphins may spend time outside the U.S. Exclusive Economic Zone, and therefore a multi-year average abundance estimate including California, Oregon and Washington is the most appropriate for management within U.S. waters. The most recent estimate of Pacific white-sided 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, 26,814 (CV=0.28) animals (Barlow 2016). This estimate includes new correction factors for animals missed during the surveys.~~ 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).

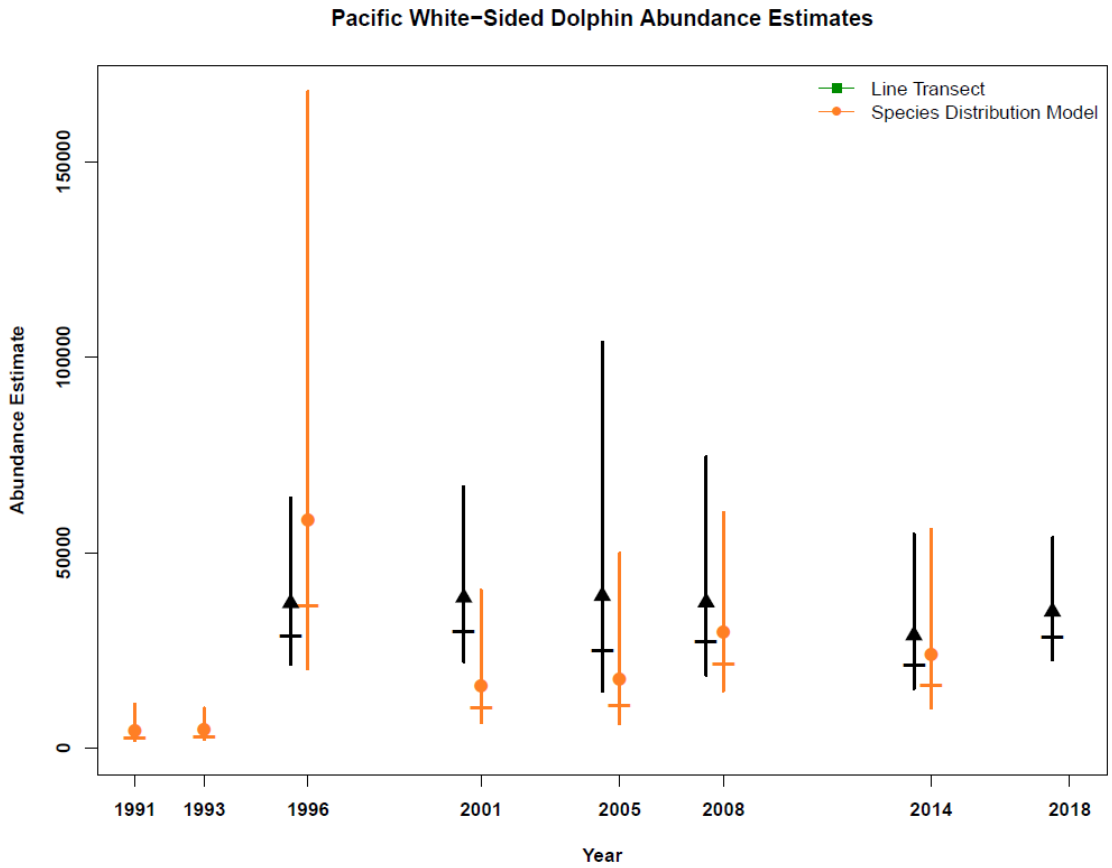


**Figure 2.** Abundance estimates and 95% confidence intervals from vessel-based line transect surveys (Barlow 2016) and species distribution models (Becker *et al*. 2020) within the California Current. Line-transect surveys in 1991 and 1993 did not include the waters of Oregon and Washington. Vertical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

**MINIMUM POPULATION ESTIMATE**

The log-normal 20th percentile of the ~~2008-2014 average~~ 2018 abundance estimate is ~~21,195~~ 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 (~~21,195~~ 29,090) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of ~~0.45~~ 0.48 (for a species of unknown status with a mortality rate CV between ~~0.6 and 0.8~~ 0.3 and 0.6; Wade and Angliss 1997), resulting in a PBR of ~~191~~ 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, ~~2010-2014, is 1.1 animals (CV=0.97)~~ 2015-2019 is 4.0 (CV=0.37) per year ~~(Carretta *et al*. 2017)~~. 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). ~~Although some Pacific-white sided 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.~~

**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 ~~*et al*. 2017~~ 2020; Jannot *et al*. ~~2011~~ 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. ~~Mean annual takes are based on 2010-2014 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 | ~~2010~~ | ~~12%~~ | ~~0~~ | ~~1.3 (2.5)~~ | ~~1.1 (0.97)~~ 2.4 (0.76) |
| | | ~~2011~~ | ~~20%~~ | ~~0~~ | ~~1.4 (2)~~ | |
| | | ~~2012~~ | ~~19%~~ | ~~0~~ | ~~0.8 (2.2)~~ | |
| | | ~~2013~~ | ~~37%~~ | ~~0~~ | ~~0.9 (1.5)~~ | |
| | | ~~2014~~ | ~~24%~~ | ~~0~~ | ~~0.9 (2)~~ | |
| | | 2015-2019 | 21% | 0 | 12.1 (0.76) | |

| Fishery Name | Data Type | Year(s) | Percent Observer Coverage | Observed Mortality | Estimated Annual Mortality | Mean Annual Takes (CV in parentheses) |
|---|---|---|---|---|---|---|
| **WA, OR, CA domestic groundfish trawl fishery (includes at-sea hake and other limited-entry groundfish sectors)**<br><br>~~WA/OR/CA groundfish (bottom trawl)~~ | observer | ~~2009-2013~~<br>2012-2016 | ~~23% (2009)~~<br>~~18% (2010)~~<br>~~100% (2011-2013)~~<br>98% - 100% | ~~0~~<br>4 | ~~0~~<br>4 (n/a) | ~~0~~<br>0.8 (n/a) |
| **Unidentified gillnet fishery** | stranding | 2015-2019 | n/a | 1 | ≥ 4 (0.46) | ≥ 0.8 (0.46) |
| ~~WA/OR/CA groundfish (midwater trawl – at-sea hake sector)~~ | ~~observer~~ | ~~2009-2013~~ | ~~100%~~ | ~~0~~ | ~~0~~ | ~~0~~ |
| ~~WA/OR/CA groundfish (midwater trawl – shoreside hake sector)~~ | ~~observer~~ | ~~2011-2013~~ | ~~100%~~ | ~~0~~ | ~~0~~ | ~~0~~ |
| **Minimum total annual takes** | | | | | | ~~1.1 (0.97)~~<br>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 ~~2010 through 2014~~ 2015 - 2019, there were ~~26~~ 14 deaths and ~~2~~ 1 serious ~~injuries~~ injury of Pacific white-sided dolphins in scientific research trawls, or an average of ~~5.6~~ 3.0 annually (Carretta *et al*. ~~2016a~~ 2021). ~~One Pacific white-sided dolphin stranded dead in Washington Inland waters during 2014, and the cause of death was determined to be a vessel strike (Carretta *et al*. 2016a). 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. (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). Applying this correction factor to the one stranded Pacific white-sided dolphin yields a minimum estimate of 4 vessel strike-related deaths during 2010-2014, or 0.8 animals annually. The average annual mortality and serious injury of Pacific white-sided dolphin from other anthropogenic activities during 2010-2014 is 5.6 (research takes), plus 0.8 animals (vessel strikes, corrected for undetected carcasses), or 6.4 animals per year.~~

**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 (~~1.1~~ 4.0 animals), plus other anthropogenic sources (~~6.4~~ 3.0) during ~~2010-2014 7.5~~ 2015-2019 (7.0 annually) is estimated to be less than the PBR (~~191~~ 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 (~~1.1~~ 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 ~~2/7/2017~~ 8/5/2021

# 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.

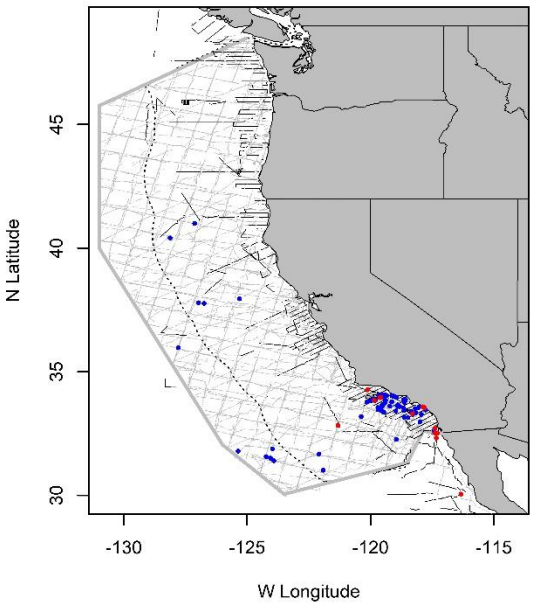


**Figure 1.** Bottlenose dolphin sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018. Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

## POPULATION SIZE

~~The most recent estimate of bottlenose 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, 1,924 (CV=0.54) animals (Barlow 2016). This estimate includes new correction factors for animals missed during the surveys.~~ 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 ~~2008-2014 geometric mean~~ 2018 abundance estimate is ~~1,255~~ 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).

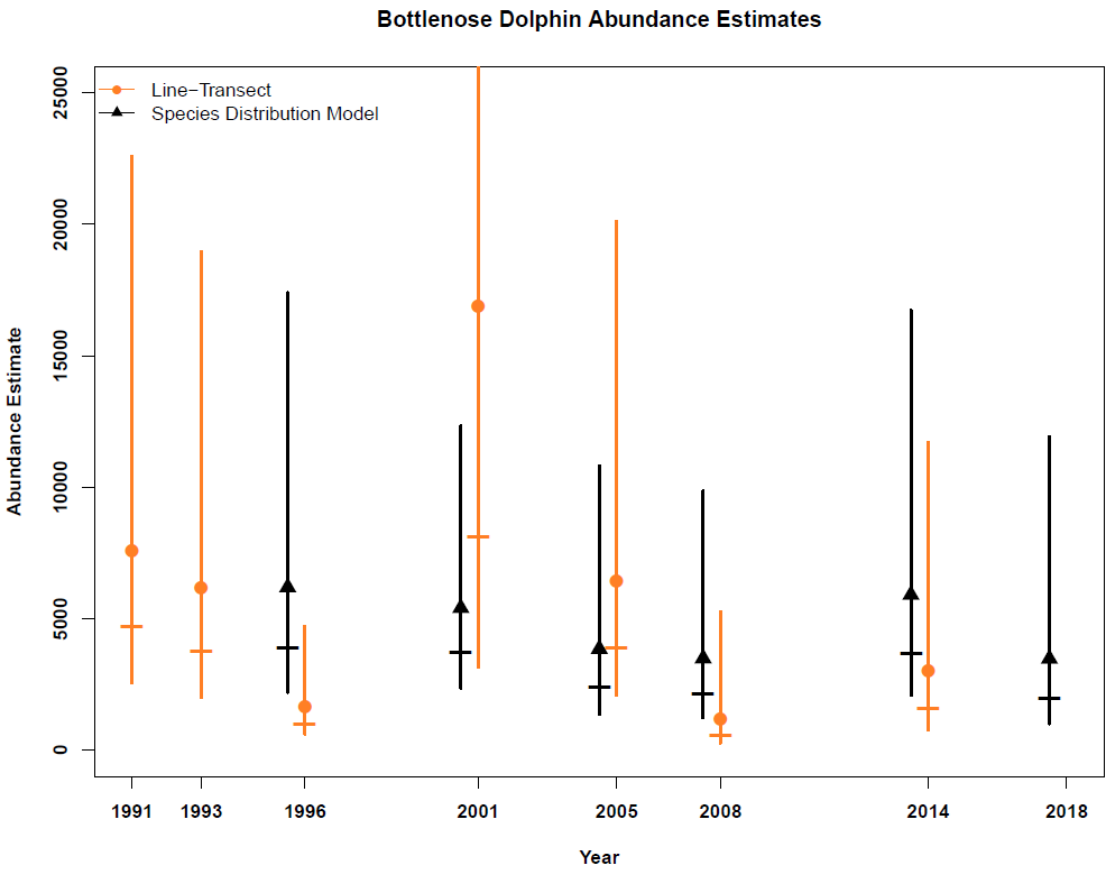


**Figure 2.** Abundance estimates and 95% confidence intervals from vessel-based line transect surveys (Barlow 2016) and species distribution models (Becker *et al.* 2020) within the California Current. Line-transect surveys in 1991 and 1993 did not include the waters of Oregon and Washington. Vertical 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 (~~1,255~~ 2,048) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of ~~0.45~~ 0.48 (for a species of unknown status with fishery mortality CV between ~~0.6 and 0.8~~ 0.3 and 0.6; Wade and Angliss 1997), resulting in a PBR of ~~11~~ 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, ~~2010-2014, is 6.9 (CV=0.74)~~ 2015-2019, is 0.8 (CV=0.52) individuals, or an average of ~~1.4 per year (CV=0.74) (Carretta *et al*. 2017)~~ 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). ~~One bottlenose dolphin was seriously injured in the limited entry fixed gear sablefish fishery during 2009, but no other deaths or injuries were reported in West Coast groundfish fisheries for the period 2009-2013 (Jannot *et al*. 2011). 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. 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 bottlenose dolphins (California/ Oregon/Washington Offshore Stock) in commercial fisheries that might take this species (Carretta *et al*. ~~2016, 2017~~2021, Carretta 2021; Jannot *et al*. ~~2011~~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 | ~~2010~~<br>~~2011~~<br>~~2012~~<br>~~2013~~<br>~~2014~~<br>2015-2019 | ~~12%~~<br>~~20%~~<br>~~19%~~<br>~~37%~~<br>~~24%~~<br>21% | ~~1~~<br>~~0~~<br>~~0~~<br>~~0~~<br>~~0~~<br>0 | ~~6.8 (0.75)~~<br>~~0.1 (7.6)~~<br>~~0 (n/a)~~<br>~~0 (n/a)~~<br>~~0 (n/a)~~<br>0.8 (0.52) | ~~1.4 (0.74)~~<br>0.16 (0.52) |
| **CA halibut / white seabass and other species set gillnet fishery** | observer | ~~2010-2014~~<br>2017 | ~~9%~~<br>~ 10% | ~~0~~<br>~~0~~<br>~~0~~<br>~~0~~<br>0 | 0 | 0 |
| ~~**California yellowtail, barracuda, and white seabass drift gillnet fishery**~~ | ~~observer~~ | ~~2010-2012~~ | ~~~4%~~ | ~~0~~ | ~~0~~ | ~~0~~ |
| ~~**CA lobster trap/pot**~~ | ~~At sea disentanglement~~ | ~~2008~~ | ~~n/a~~ | ~~0 (1)~~ | ~~1 (n/a)~~ | ~~0.2 (n/a)~~ |
| **Limited entry fixed gear ~~(longline) sablefish fishery~~** | ~~At sea disentanglement~~<br>observer | ~~2005~~<br>~~2006~~<br>~~2007~~<br>~~2008~~<br>~~2009~~<br>2012-2016 | ~~0.5%~~<br>~~1.5%~~<br>~~3.4%~~<br>~~1.5%~~<br>~~2.4%~~<br>~ 4% | 0 ~~(1)~~ | ~~1~~ 3.28 (n/a)* | ~~0.2~~ 0.656 (n/a) |
| **Minimum total annual takes** | | | | | | ~~≥1.6 (0.74)~~<br>≥ 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 ~~estimate of bycatch was derived from the one observation of a bottlenose dolphin released injured from sablefish gear~~ (Jannot *et al*. ~~2011~~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 (~~1.6~~ 0.82 /yr) are less than the calculated PBR (~~11~~ 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 8/7/2021 ~~02/09/2017~~

# 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.

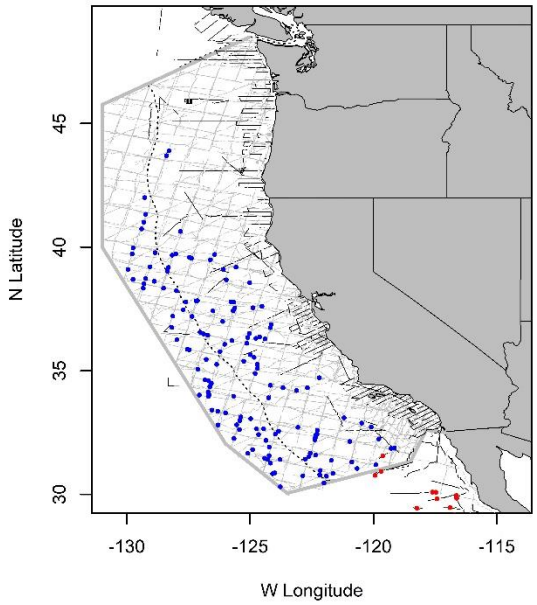


**Figure 1.** Striped dolphin sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018 (Barlow 2016, Henry *et al*. 2020). Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are 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). ~~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 striped 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, 29,211 (CV=0.20) animals (Barlow 2016). This estimate includes new correction factors for animals missed during the surveys.~~ 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 ~~2008-2014 average~~ 2018 abundance estimate is ~~24,782~~ 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*. ~~2012,~~ 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.

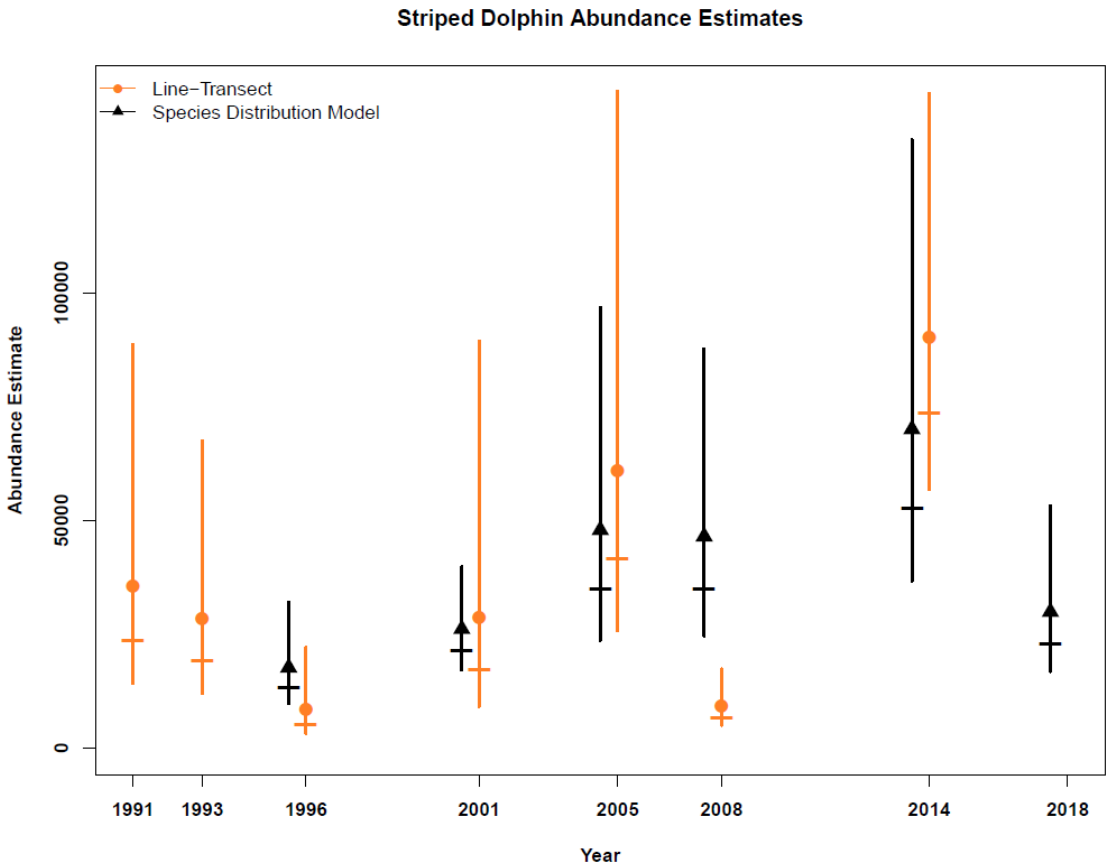


**Figure 2**. Abundance estimates and 95% confidence intervals from vessel-based line transect surveys (Barlow 2016) and species distribution models (Becker *et al*. 2020) within the California Current. Line-transect surveys in 1991 and 1993 did not include the waters of Oregon and Washington. Vertical 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 (~~24,782~~ 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 ~~238~~ 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, ~~2010-2014~~ 2015-2019, is zero animals per year (Carretta ~~et al. 2017~~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. ~~Carretta *et al*. (2016a) 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, i~~I 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~~a~~). ~~One~~ Five striped dolphin stranded during ~~2010-2014~~ 2015-2019 with evidence of fishery interactions (Carretta *et al*. ~~2016b~~ 2021), yielding a minimum estimate of ~~four~~ 20 fishery-related dolphin deaths. ~~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 striped dolphins (California/ Oregon/Washington Stock) in commercial fisheries that might take this species (Carretta *et al*. ~~2016a, 2016b, 2017.~~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). ~~Coefficients of variation for mortality estimates are provided in parentheses.~~

| 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 | ~~2010~~<br>~~2011~~<br>~~2012~~<br>~~2013~~<br>~~2014~~<br>2015-2019 | ~~12%~~<br>~~20%~~<br>~~19%~~<br>~~37%~~<br>~~24%~~<br>21% | ~~0~~<br>~~0~~<br>~~0~~<br>~~0~~<br>~~0~~<br>0 | 0 (n/a) | 0 (n/a) |
| **Unidentified fishery (includes unidentified gillnet)** | Stranding | ~~2010-2014~~<br>2015-2019 | n/a | ~~1~~<br>5 | ~~≥ 4~~<br>≥ 20 | ≥ ~~0.8~~ 4.0 (0.46) |
| **Minimum total annual takes** (includes correction for unobserved beach strandings) | | | | | | ≥ ~~0.8~~ 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 (≥~~0.80~~ 4.0) is less than 10% of the PBR (~~238~~ 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 8/6/2021 ~~02/07/2017~~

# SHORT-BEAKED COMMON DOLPHIN (*Delphinus 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 ~~between~~ from the coast ~~and~~ to at least 300 nmi distance from shore (Figure 1). The abundance of this species off California ~~has been shown to~~ 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 ~~have been~~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). ~~However, preliminary data on v~~ 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.

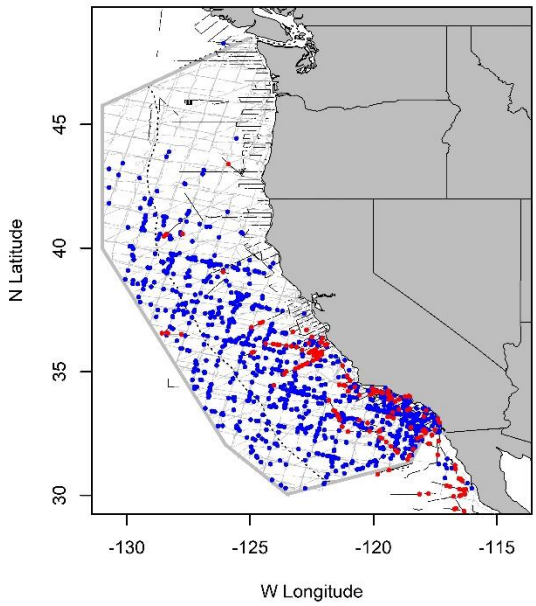


**Figure 1**. Short-beaked common dolphin sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018 (Barlow 2016, Henry et al. 2020). Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey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

## POPULATION SIZE

The distribution of short-beaked common dolphins ~~throughout~~ in this region is highly variable, ~~apparently~~ in response to oceanographic changes on both seasonal and interannual time scales (Heyning and Perrin 1994; Forney 1997; Forney and Barlow 1998). ~~As oceanographic conditions vary, short-beaked common 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 short-beaked common 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, 969,861 (CV = 0.17) animals (Barlow 2016). This estimate includes new correction factors for animals missed during the surveys.~~ 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 2008-2014 average~~ 2018 abundance estimate is ~~839,325~~ 888,971 short-beaked common dolphins.

**Current Population Trend**

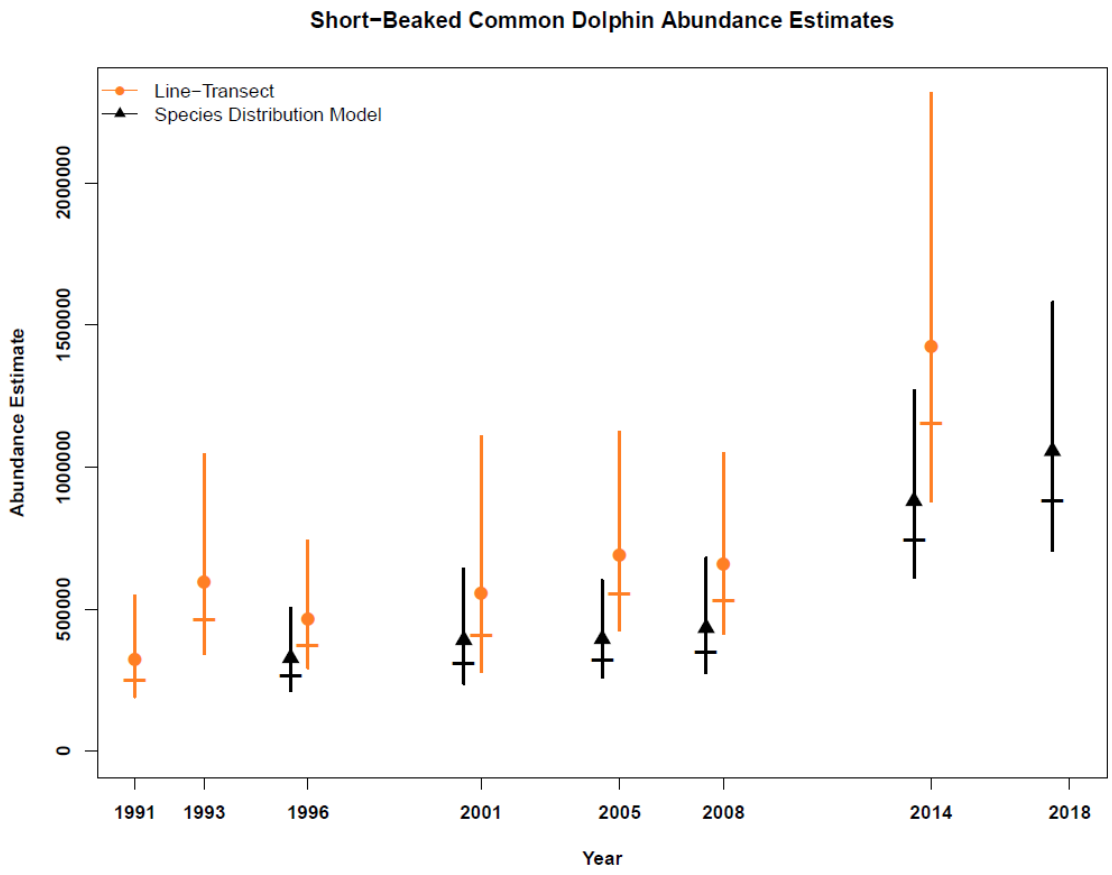


**Figure 2.** Abundance estimates and 95% confidence intervals from vessel-based line transect surveys (Barlow 2016) and species distribution models (Becker *et al.* 2020a) within the California Current. Line-transect surveys in 1991 and 1993 did not include the waters of Oregon and Washington. Vertical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

Short-beaked common dolphin abundance off the U.S. West Coast ~~is known to~~ increases during warm-water periods (Dohl *et al.* 1986, Forney and Barlow 1998, Barlow 2016). ~~The most recent~~ Estimated abundance increased significantly beginning in 2014 survey ~~was conducted~~ during extremely warm ocean conditions (Bond *et al*. 2015) and ~~resulted in the largest abundance estimate since large-scale surveys began in 1991~~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 (~~839,325~~ 888,971) 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 a mortality rate CV< 0.30; Wade and Angliss 1997), resulting in a PBR of ~~8,393~~ 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 ~~summed~~ estimate of mortality ~~and serious injury~~ for short-beaked common dolphin in the California drift gillnet fishery for ~~the five~~ most recent 5 year period of ~~s of monitoring~~, ~~2010-2014~~ 2015-2019~~, is approximately 100 individuals, or an average of 20 (CV=0.18)~~ averages 26.7 (CV=0.25) per year (Carretta *et al*. ~~2017~~2021, Carretta 2021) (Table 1). ~~No takes were documented by observers during the most recent five years of monitoring for other gillnet and purse seine fisheries that have interacted with short-beaked common dolphins in the past. However, two short-beaked common dolphins stranded with evidence of fishery interaction with an unidentified gillnet fishery.~~

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.

~~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) 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*. 2016a). Applying this correction factor to the two stranded short-beaked common dolphins yields a minimum estimate of 8 fishery-related dolphin deaths.~~

**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*. ~~2016b, 2017~~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 | ~~2010~~<br>~~2011~~<br>~~2012~~<br>~~2013~~<br>~~2014~~<br>2015-2019 | ~~12%~~<br>~~20%~~<br>~~19%~~<br>~~37%~~<br>~~24%~~<br>21% | ~~3~~<br>~~2~~<br>~~5~~<br>~~6~~<br>~~6~~<br>31 | ~~21.2 (0.53)~~<br>~~15.2 (0.46)~~<br>~~21.3 (0.41)~~<br>~~16.5 (0.25)~~<br>~~23.5 (0.31)~~<br>133.6 (0.25) | ~~20 (0.18)~~<br>26.7 (0.25) |
| ~~**CA squid purse seine**~~ | ~~observer~~ | ~~2004~~<br>~~2005~~<br>~~2006~~<br>~~2007~~<br>~~2008~~ | ~~unknown~~<br>~~1.1%~~<br>~~unknown~~<br>~~<5%~~<br>~~<5%~~ | ~~0~~<br>~~1~~<br>~~0 (1)~~<br>~~0~~<br>~~0~~ | ~~0~~<br>~~87 (0.98)~~<br>~~≥ 1 (n/a)~~<br>~~0~~<br>~~0~~ | ~~18 (0.98)~~ |
| **CA halibut / white seabass and other species set gillnet fishery** | ~~observer~~<br>strandings | ~~2010-2014~~<br>2015-2019 | ~~9%~~<br>n/a | ~~0~~<br>3 | ~~0~~<br>≥ 12 (0.46) | ≥ 3 ~~0~~ (~~n/a~~ 0.46) |
| **Hawaii Shallow Set Longline fishery** | observer | ~~2010-2014~~<br>2015-2019 | 100% | ~~0~~<br>~~0 (1)~~<br>~~0~~<br>~~0~~<br>~~0 (1)~~<br>0 | ~~0 (2)~~<br>0 | ~~0.4 (n/a)~~<br>0 |
| **Unidentified gillnet fishery** | Stranding | ~~2010-2014~~<br>2015-2019 | n/a | ~~2~~ 1 | ≥~~8~~ 4 (0.46) | ≥~~1.6~~ 0.8 (0.46) |
| **Minimum total annual takes** (includes correction for unobserved beach strandings) | | | | | | ≥ ~~40 (0.45)~~<br>≥ 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 there have not been any recent interactions observed. ~~Between 2004 and 2008, there were 377 sets observed in the squid purse seine fishery and one short-beaked common dolphin mortality was observed in 2005, with a resulting mortality estimate of 87 (CV=0.98) animals (Carretta and Enriquez 2006). It is likely, due to the low observer coverage that year (~1%), combined with a relatively rare entanglement event, that this estimate is positively-biased (Carretta and Moore 2014). In addition, there was one squid purse seine set in 2006 where 8 unidentified dolphins were encircled. Seven were released alive and the eighth was seriously injured. For purposes of this stock assessment report, it is assumed that the unidentified seriously injured dolphin was a short-beaked common dolphin, due to its high abundance within the fishing area and a previous record of this species having been killed in the fishery.~~

~~Two short-beaked common dolphins were reported released injured from the Hawaii shallow set longline fishery (one each in 2011 and 2014 with 100% observer coverage, Table 1). These interactions occurred outside of the U.S. EEZ just west of the California Current and likely involved dolphins from the CA/OR/WA stock of short-beaked common dolphins (NOAA Pacific Islands Regional Office 2017).~~

## Other Mortality

~~In the eastern tropical Pacific, 'northern common dolphins' have been incidentally killed in international tuna purse-seine fisheries since the late 1950's and are managed separately under a section of the MMPA written specifically for the management of dolphins involved in eastern tropical Pacific tuna~~

~~fisheries. Cooperative international management programs have dramatically reduced overall dolphin mortality in these fisheries in recent decades (IATTC 2015). Between 2007 and 2014, annual fishing mortality of northern common dolphins (potentially including both short-beaked and long-beaked common dolphins) ranged between 35 and 124 animals, with an average of 75 (IATTC, 2015). Although it is unclear whether these animals are part of the same population as short-beaked common dolphins found off California, the distributions of both of the species that comprise the 'northern common dolphins' appear to shift into U.S. waters during certain oceanographic conditions (IATTC 2006).~~ 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 ~~probably reflects~~ 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 ~~2010-2014 (40 animals)~~ 2015-2019 (≥ 30.5 animals) is estimated to be less than the PBR (~~8,393~~ 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 8/7/2021 ~~2/10/2017~~

# LONG-BEAKED COMMON DOLPHIN
# (*Delphinus ~~capensis~~ 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.

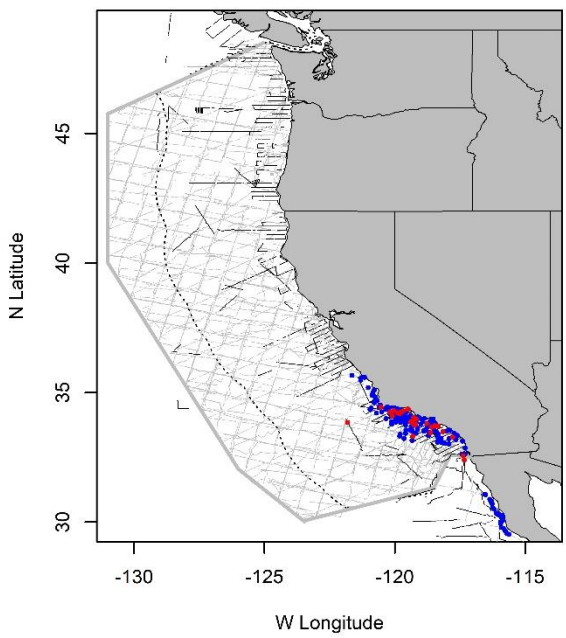


**Figure 1.** Long-beaked common dolphin sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018 (Barlow 2016, Henry *et al*. 2020). Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018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

## POPULATION SIZE

~~The distribution and abundance of long-beaked common dolphins off California varies inter-annually and seasonally (Heyning and Perrin 1994). As oceanographic conditions change, long-beaked common dolphins may move between Mexican and U.S. waters, and therefore a multi-year average abundance estimate is the most appropriate for management within the U.S. waters. The geometric mean abundance estimate for California, Oregon and Washington waters based on two ship surveys conducted in 2008 and 2014 (Barlow 2016) is 101,305 (0.49) long-beaked common dolphins. This estimate includes new correction factors for animals missed during the surveys. Although Carretta *et al*. (2011) also estimated abundance of this stock from a 2009 survey, that estimate did not include the correction factors and had high imprecision for one of the geographic strata, so it is not included in the multi-year average.~~ 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 ~~weighted 2008-2014~~ 2018 abundance estimate is ~~68,432~~ 69,636 long-beaked common dolphins.

**Current Population Trend**

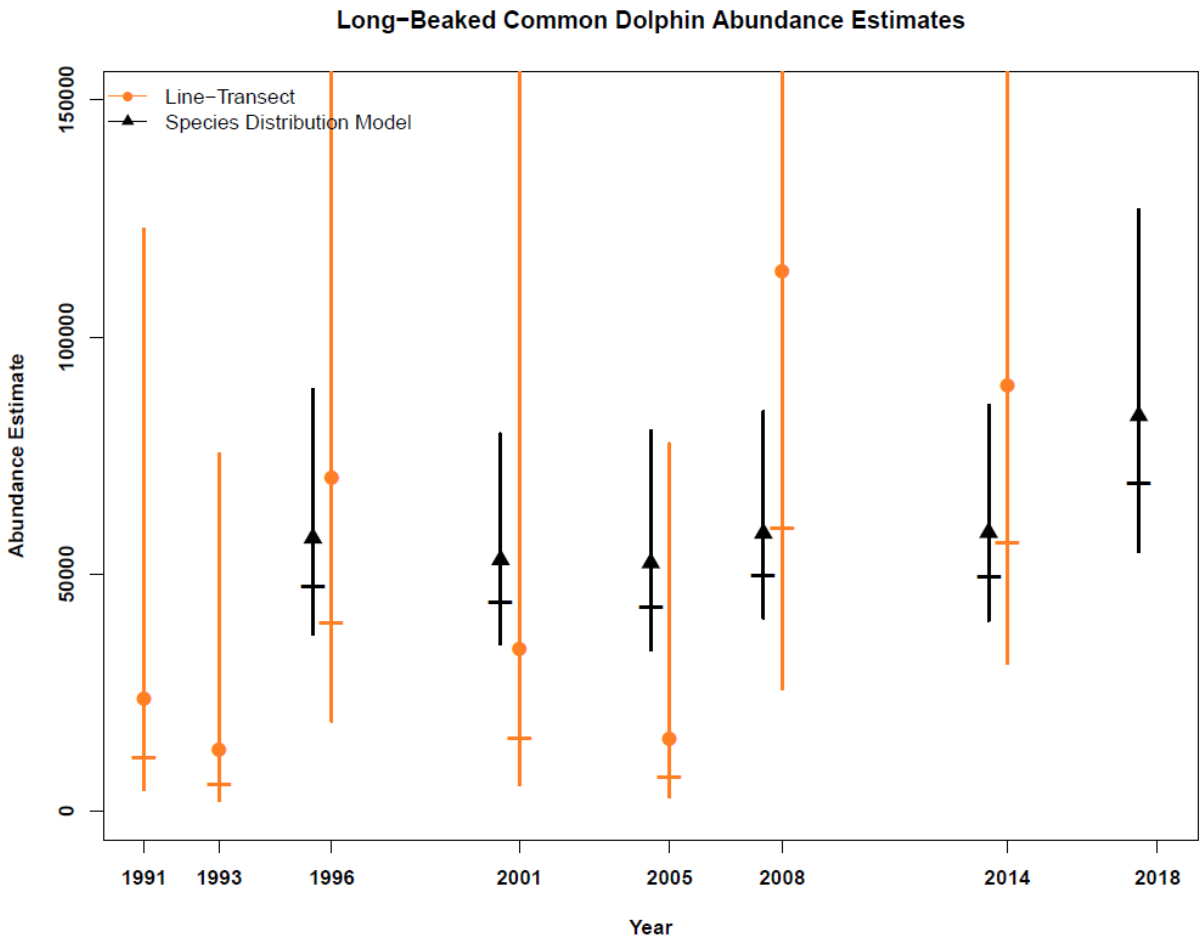


**Figure 2**. Abundance estimates and 95% confidence intervals from vessel-based line transect surveys (Barlow 2016) and species distribution models (Becker *et al.* 2020) within the California Current. Line-transect surveys in 1991 and 1993 did not include the waters of Oregon and Washington, but all 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. ~~While no formal statistical trend analysis exists for this stock of long-beaked common dolphin, abundance estimates for California waters from vessel-based line-transect surveys have been greater in recent years as water conditions have been warmer (Barlow 2016)~~. The ratio of strandings of long-beaked to short-beaked common dolphin in southern California has varied, suggesting that the proportions of each species present change as ocean

conditions vary (Heyning and Perrin 1994, Danil *et al*. 2010).~~ During a 2009 ship-based survey of California and Baja California waters, the ratio of long-beaked to short-beaked common dolphin sightings was nearly 1:1, whereas during previous surveys conducted from 1986 to 2008 in the same geographic strata, the ratio was approximately 1:3.5 (Carretta *et al*. 2011).~~ 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 (~~68,432~~ 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 ~~657~~ 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, ~~2010-2014~~ 2015-2019, averages ~~2.0~~ 1.7 (CV=~~0.99~~ 0.60) per year (Carretta ~~*et al*. 2017~~ 2020). ~~One interaction with the halibut set gillnet fishery was observed during 2010-2014, resulting in an estimate of 7 (CV=1.17) dolphins (Carretta and Enriquez 2012). No mortality or serious injury has been documented by observers during the most recent five years of monitoring for the small mesh gillnet fishery, which has interacted with long-beaked common dolphins in the past. However, 36 long-beaked common dolphins stranded with evidence of interaction with unidentified fisheries.~~ 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 and injury documentation is often based on stranding data, where~~ Human-caused mortality totals based on observed ~~raw~~ counts from strandings are negatively-biased because only a fraction of carcasses are detected (Carretta *et al*. 2016). ~~Carretta *et al*. (2016a) 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 (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 ~~36~~ 21 stranded long-beaked common dolphins yields a minimum estimate of ~~144~~ 84 fishery-related dolphin deaths, or an average of ~~29~~ 16.8 ~~per year~~ annually (Table 1). ~~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 long-beaked common dolphins (California Stock) in commercial fisheries that might take this species (Carretta 2021, Carretta *et al*. ~~2016b~~2021~~, 2017~~). 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 | ~~2010~~<br>~~2011~~<br>~~2012~~<br>~~2013~~<br>~~2014~~<br>2015-2019 | ~~12%~~<br>~~20%~~<br>~~19%~~<br>~~37%~~<br>~~24%~~<br>21% | ~~1~~<br>~~1~~<br>~~0~~<br>~~0~~<br>~~0~~<br>2 | ~~1.9 (1.1)~~<br>~~5.1 (1.3)~~<br>~~0 (n/a)~~<br>~~0.2 (2.2)~~<br>~~2.8 (1.4)~~<br>8.7 (0.60) | ~~2.0 (0.99)~~<br>1.7 (0.60) |
| ~~**CA small mesh drift gillnet fishery for white seabass, yellowtail, barracuda, and tuna**~~ | ~~observer~~ | ~~2010-2012~~ | ~~~4%~~ | ~~0~~ | ~~0~~ | ~~0 (n/a)~~ |
| **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** | ~~Observer~~<br>Strandings | ~~2010-2014~~<br>2015-2019 | ~~9%~~<br>n/a | ~~1~~<br>2 | ~~7 (1.17)~~<br>≥ 8 (n/a) | ~~1.4 (1.17)~~<br>≥ 1.6 (0.46) |
| **Unidentified gillnet fishery interaction** | Strandings | ~~2010-2014~~<br>2015-2019 | - | ~~36~~<br>21 | ~~≥144~~<br>≥ 84 | ~~≥29 (0.46)~~<br>≥ 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) | | | | | | ~~≥32 (0.42)~~<br>≥ 26.5 (0.39) |

## Other Mortality

~~Three long-beaked common dolphins died near San Diego in 2011 as the result of blast trauma associated with underwater detonations conducted by the U.S. Navy. Three days later, a fourth animal stranded approximately 70 km north of that location with similar injuries (Danil and St. Leger 2011). One long-beaked common dolphin was incidentally killed during fishery research during 2013 (Carretta *et al*. 2016b).~~ Stranding records from ~~2010-2014~~ 2015-2019 include ~~three additional~~ two deaths resulting from hook and line fishery entanglements (Carretta *et al*. 2021). ~~human-related long-beaked common dolphin deaths, including one animal that was struck by a vessel, one animal that had ingested marine debris, and one animal that had been cut in half (Carretta *et al*. 2016b).~~ Applying the minimum correction factor of 4 to account for undetected mortality (Carretta *et al*. 2016a), ~~this~~ yields an estimated ~~12~~ 16 human-caused long-beaked common dolphin deaths from hook and line fisheries or 3.2 annually for 2015-2019. ~~From all sources combined, this results in a total of 17 non-fishery human-caused deaths between 2010 and 2014, or an average of 3.4 dolphins per year.~~

'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).

~~In the eastern tropical Pacific, 'northern common dolphins' have been incidentally killed in international tuna purse-seine fisheries since the late 1950's and are managed separately under a section of the MMPA written specifically for the management of dolphins involved in eastern tropical Pacific tuna fisheries. Cooperative international management programs have dramatically reduced overall dolphin mortality in these fisheries (Joseph 1994). Between 2007 and 2014, annual fishing mortality of northern common dolphins (potentially including both short-beaked and long-beaked common dolphins) ranged between 35 and 124 animals, with an average of 75 (IATTC 2015). The distributions of both of the species that comprise the 'northern common dolphins' appear to shift into U.S. waters during certain oceanographic conditions (IATTC 2006).~~

## 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 ~~new~~ 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 (≥~~32~~ 26.5 dolphins /year) and other sources (~~3.4~~ 3.2 dolphins/year) is ~~35.4~~ 29.7 long-beaked common dolphins. This does not exceed the PBR (~~657~~ 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 (~~32~~ 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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Revised ~~2/10/2017~~ 8/7/2021

# 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 recent 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.

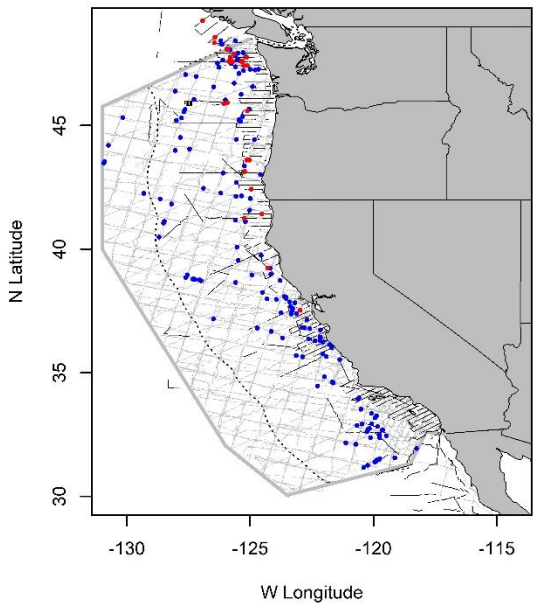


**Figure 1.** Northern right whale dolphin sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018. Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

## 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 interannual 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~~, and therefore a multi-year average abundance estimate is the most appropriate for management within U.S. waters. The most recent estimate of northern right whale 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, 26,556 (CV=0.44) animals (Barlow 2016). This estimate includes new correction factors for animals missed during the surveys~~. 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 ~~2008-2014 average~~ 2018 abundance estimate is ~~18,608~~ 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 interannual time scales (Forney and Barlow 1998, Becker *et al*. 2012, 2020, Barlow 2016), but no long term trends have been identified (Figure 2).

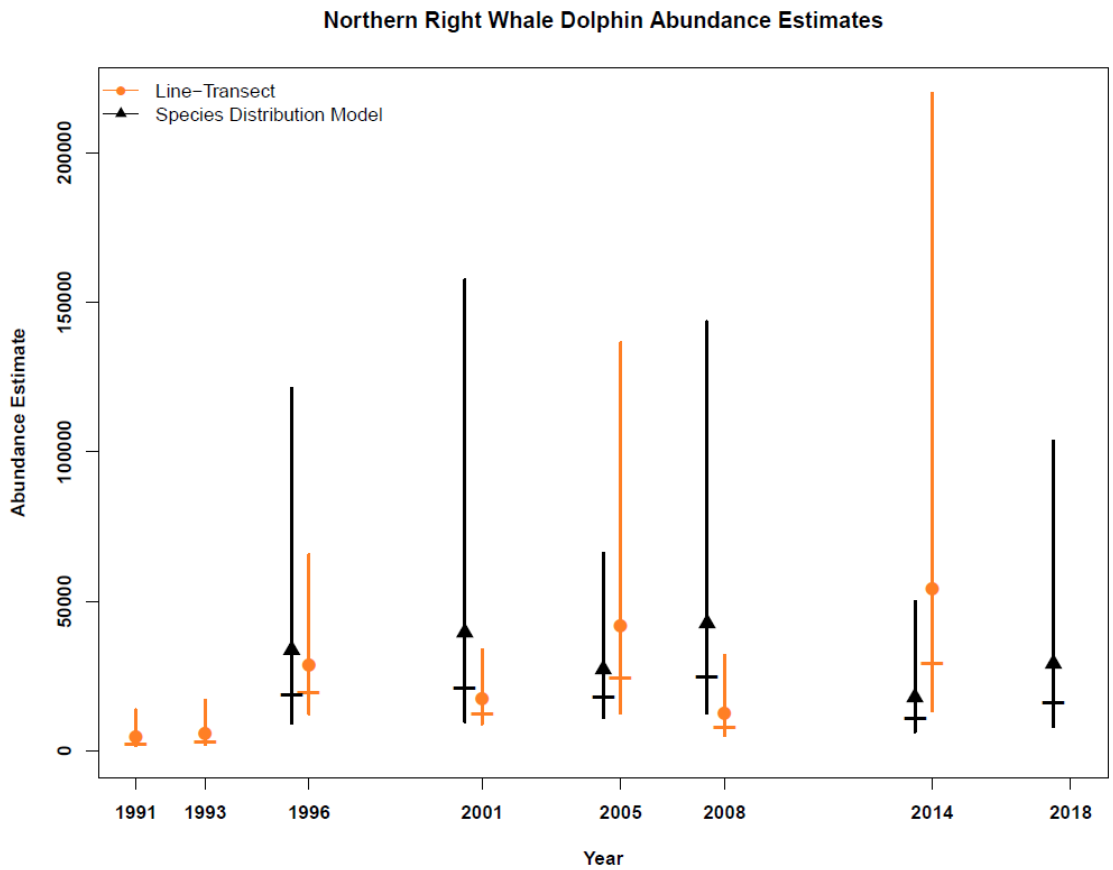


**Figure 2.** Abundance estimates and 95% confidence intervals from vessel-based line transect surveys (Barlow 2016) and species distribution models (Becker *et al*. 2020) within the California Current. Line-transect surveys in 1991 and 1993 did not include the waters of Oregon and Washington. Horizontal 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 (~~18,608~~ 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 ~~179~~ 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). ~~The estimate of mortality and serious injury for northern right whale dolphin in the California drift gillnet fishery for the five most recent years of monitoring, 2010-2014, is 17.6 (CV=0.36) individuals, or an average of 3.5 per year (Carretta *et al.* 2016). 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 northern right-whale dolphins (California/Oregon/Washington Stock) in commercial fisheries that might take this species (Carretta ~~*et al.* 2017~~ 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 | ~~2010~~<br>~~2011~~<br>~~2012~~<br>~~2013~~<br>~~2014~~<br>2015-2019 | ~~12%~~<br>~~20%~~<br>~~19%~~<br>~~37%~~<br>~~24%~~<br>21% | ~~1~~<br>~~1~~<br>~~1~~<br>~~2~~<br>~~1~~<br>7 | ~~3.9 (1)~~<br>~~5.5 (0.85)~~<br>~~3.7 (0.95)~~<br>~~3.3 (0.45)~~<br>~~2.5 (0.83)~~<br>28.1 (0.39) | ~~3.8 (0.40)~~<br>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** | | | | | | ~~3.8 (0.40)~~<br>≥ 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 ~~in 2010-2014~~ from 2015-2019 (~~3.8~~ 6.6 animals) is estimated to be less than the PBR (~~179~~ 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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Revised ~~02/28/2021~~ 08/04/2021

# 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).

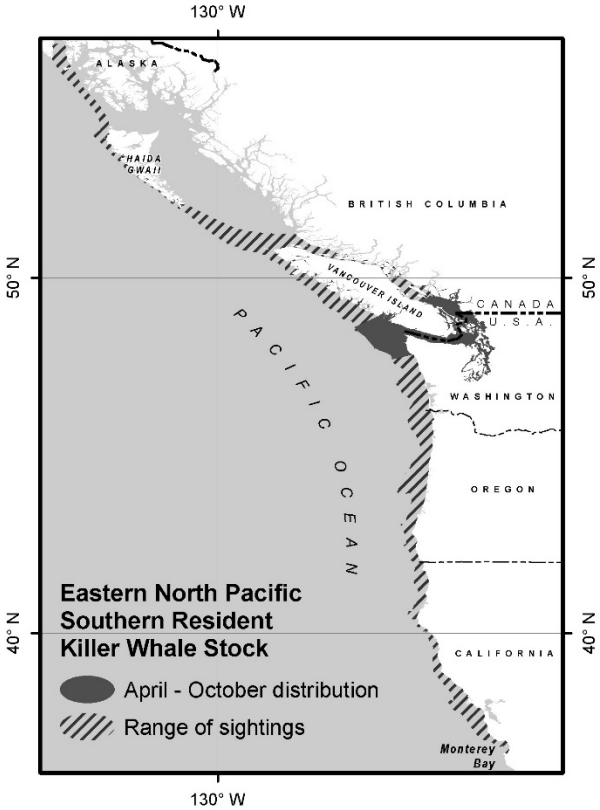


**Figure 1**. Approximate April - October distribution of the Eastern North Pacific Southern Resident killer whale stock (shaded area) and range of sightings (diagonal lines).

The range of southern resident killer whales is described in the draft biological report for the Proposed Revision of the Critical Habitat Designation for Southern Resident Killer Whales (NMFS 2019a, 2019b): "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. 2002). The whales also visit outer coastal waters off Washington and Vancouver Island, especially in the area between Grays Harbor and the Columbia River (Ford et al. 2000, Hanson et al. 2017), but travel 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 ~~73~~ 72 whales in ~~2019~~ 2020 (Fig. 2; Ford *et al.* 2000; Center for Whale Research 2019). The 2001-2005 counts included a whale born in 1999 (L-98) that was listed as missing during the annual census in May and June 2001 but was subsequently discovered alone in an inlet off the west coast of Vancouver Island. L-98 remained separate from L pod until 10 March 2006 when he died due to injuries associated with a vessel interaction in Nootka Sound. L-98 has been subtracted from the official 2006 and subsequent population censuses. The most recent census spanning 1 July ~~2018~~ 2019 through 1 July ~~2019~~ 2020 includes ~~two new calves and the deaths of a juvenile female, two young adult males, and a post-reproductive female~~the death of an adult male, but does not include the two calves that were born in fall 2020.

### Minimum Population Estimate

The abundance estimate for this stock of killer whales is a direct count of individually identifiable animals. It is thought that the entire population is censused every year. This estimate therefore serves as both a best estimate of abundance and a minimum estimate of abundance. Thus, the minimum population estimate ($N_{min}$) for the Eastern North Pacific Southern Resident stock of killer whales is ~~73~~ 72 animals.

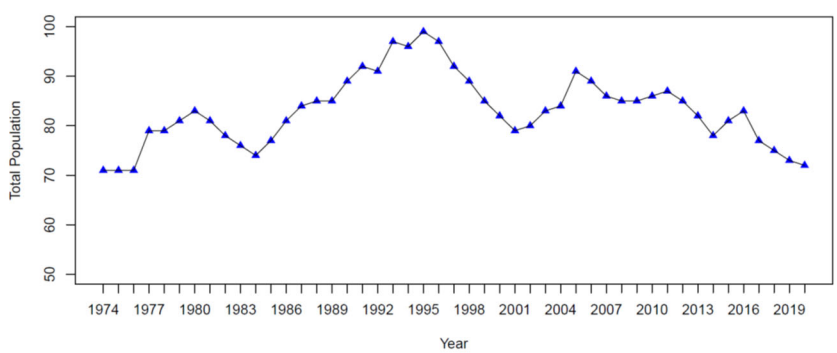


**Figure 2.** Population of Eastern North Pacific Southern Resident stock of killer whales, 1974-2020 ~~2019~~. Each year's count includes animals first seen and first missed; a whale is considered first missed the year after it was last seen alive (Ford *et al.* 2000; Center for Whale Research ~~2019~~2020).

### 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 ~~73~~ 72 animals as of the ~~2019~~ 2020 census (Ford *et al.* 2000; Center for Whale Research ~~2019~~2020).

## 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 (~~73~~72) 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

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 ~~2014-2018~~ 2015-2019 (Carretta ~~2020~~ 2021, Carretta *et al.* ~~2020~~ 2021).

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.

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). 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. In press). 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). 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 ~~2014-2018~~ 2015-2019 (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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Revised 8/5/2021 ~~6/4/2018~~

# 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 Ocean (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 have been seen less frequently and are presumed to be farther offshore during the colder water months of November through 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.

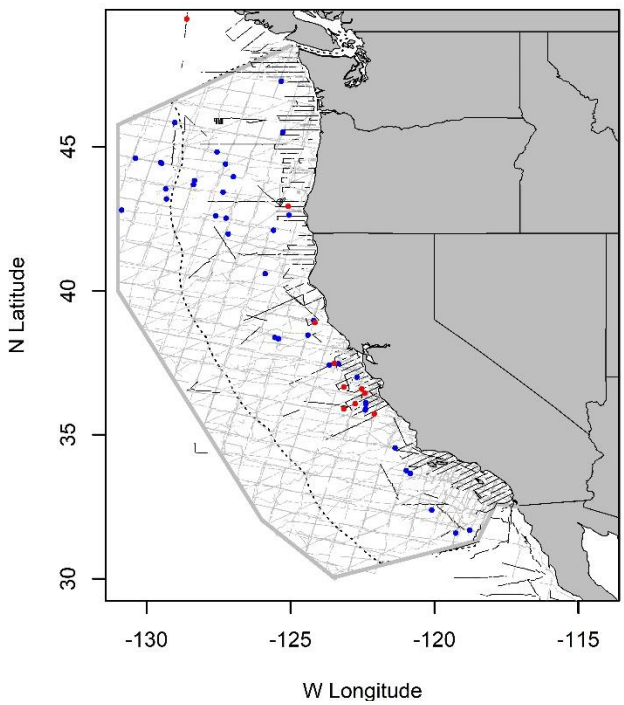


**Figure 1.** Baird's beaked whale sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018. Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

## POPULATION SIZE

~~Barlow (2016) recently estimated Baird's beaked whale abundance in the California Current at 5,394 (CV=0.83) and 7,960 (CV=0.93) whales for surveys conducted in 2008 and 2014, respectively. These estimates are higher than previously-published estimates for this region because they include lower estimates of trackline detection probability, *g(0)*, based on Beaufort sea-state specific estimates of detectability for *Mesoplodon* species (Barlow 2015). A trend-based analysis of line-transect data from all surveys conducted between 1991 and 2014 yielded an estimate of abundance of 2,697 (CV=0.60) whales (Moore and Barlow 2017); these were based on newer (lower) *g(0)* estimates from earlier analyses, but were not as low as those used by Barlow (2016) and thus the abundance estimates are not as high (Moore and Barlow 2017). Based on this analysis and weak evidence for any trend in abundance, the recent 2014 estimate of 2,697 (CV=0.60) Baird's beaked whales is the most appropriate estimate for this stock.~~ 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 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 log-normal 20th percentile of the 2014 abundance estimate is 1,633 Baird's beaked whales (Moore and Barlow 2017).~~ The minimum population size estimate is taken as the lower 20th percentile of the 2018 abundance estimate, or 894 whales (Becker *et al*. 2018).

**Current Population Trend**

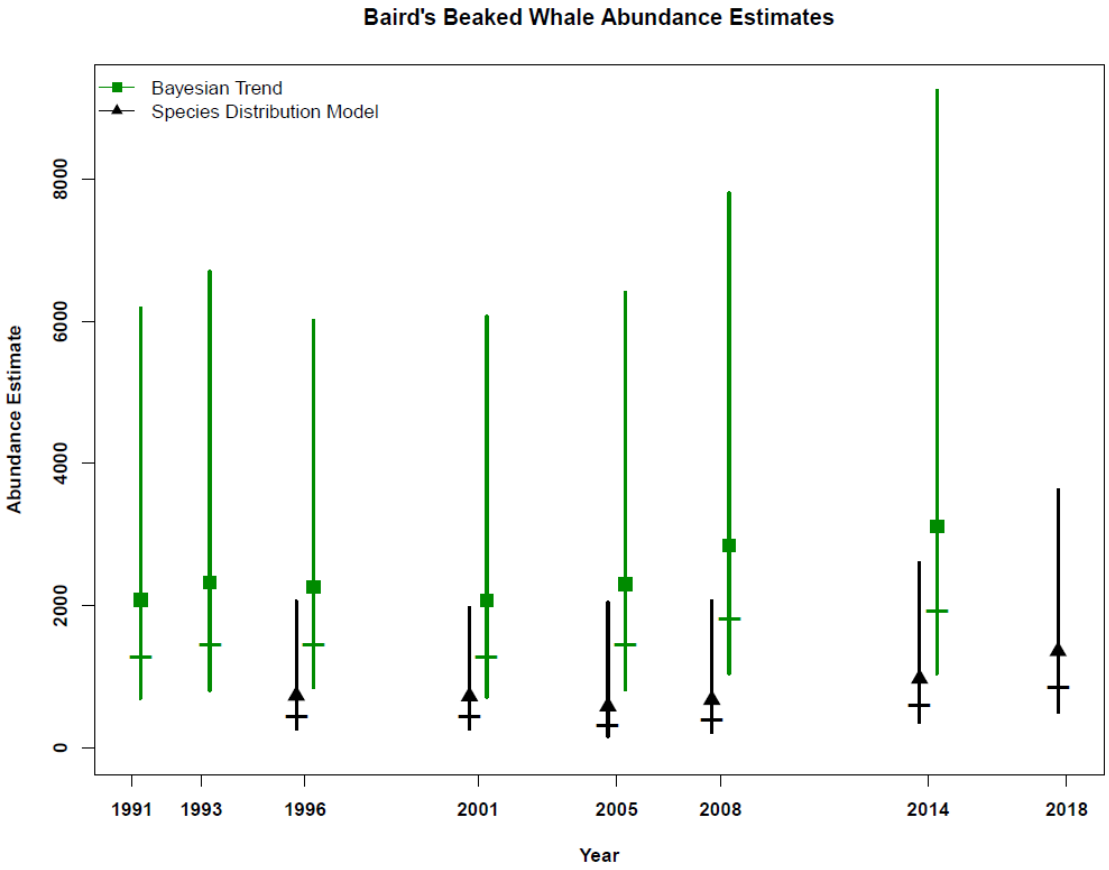


**Figure 2.** Baird's beaked whale abundance estimated from a Bayesian trend analysis (Moore and Barlow 2017) and habitat-based species distribution models based on 1991-2018 line-transect survey data (Becker *et al*. 2020). Vertical bars indicate approximate 95% log-normal confidence limits for Bayesian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

~~The analysis by Moore and Barlow (2013) did not suggest evidence of an abundance trend during 1991–2008 for Baird's beaked whale in waters off the U.S. west coast, but an updated analysis that includes 2014 survey data indicates that the~~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 (~~1,633~~ 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 ~~16~~ 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 ~~*et al.* 2017a~~ 2021). Mean annual takes in Table 1 are based on ~~2011-2015~~ 2015-2019 data. This results in an average estimated annual mortality of zero Baird's beaked whales (Carretta ~~*et al.* 2017a~~ 2021). ~~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 Baird's beaked whales (California/Oregon/Washington Stock) in commercial fisheries that might take this species. ~~The single observed entanglement resulted in the death of the animal.~~ Coefficients of variation for mortality estimates are provided in parentheses. Mean annual takes are based on ~~2011-2015~~ 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 | ~~2011~~<br>~~2012~~<br>~~2013~~<br>~~2014~~<br>~~2015~~<br>2015-2019 | ~~20%~~<br>~~19%~~<br>~~37%~~<br>~~24%~~<br>~~20%~~<br>21% | ~~0~~<br>~~0~~<br>~~0~~<br>~~0~~<br>~~0~~<br>0 | ~~0~~<br>~~0~~<br>~~0~~<br>~~0~~<br>~~0~~<br>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 ~~Washington state~~ California in ~~2003~~ 2016 and the cause of death was attributed to a ship strike (Carretta *et al*. 2021). No other human-caused mortality has been reported for this stock for the period ~~2011-2015~~ 2015-2019 (Carretta *et al.* ~~2017b~~ 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 ~~2011-2015~~ 2015-2019 is ~~zero~~ 0.2 animals/year (one vessel strike death). Because recent fishery and human-caused mortality is less than the PBR (~~16~~ 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 ~~04/15/2020~~ 8/7/2021

# HUMPBACK WHALE (*Megaptera novaeangliae*): California/Oregon/Washington Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

NMFS ~~has~~ conducted a global Status Review of humpback whales and ~~recently~~ revised the ESA listing of the species, based on identification of distinct population segments (DPSs) (Bettridge *et al*. 2015, NOAA 2016a). NMFS is evaluating the stock structure of humpback whales under the MMPA, but no ~~changes~~ revisions to the current stock ~~structure~~ definition are presented at this time. ~~However, effects of the ESA listing final rule on the status of the stock are discussed below.~~

North Pacific humpback whales (*M. novaeangliae kuzira*) comprise a distinct subspecies based on mtDNA and DNA relationships and distribution compared to North Atlantic humpback whales (*M. n. novaeangliae*) and those in the Southern Hemisphere (*M. n. australis*) (Jackson *et al*. 2014). Humpback whales occur throughout the North Pacific, with multiple populations recognized based on low-latitude winter breeding areas (Baker *et al*. 1998, Calambokidis *et al*. 2001, Calambokidis *et al*. 2008, Barlow *et al*. 2011, Fleming and Jackson 2011). North Pacific breeding areas fall broadly into three regions: 1) western Pacific (Japan and Philippines); 2) central Pacific (Hawaiian Islands); and 3) eastern Pacific (Central America and Mexico) (Calambokidis *et al*. 2008). Exchange of animals between breeding areas occurs rarely, based on photo-identification evidence ~~data of individual whales~~ (Calambokidis *et al*. 2001, Calambokidis *et al*. 2008). Photo-identification evidence also ~~suggests~~ shows strong site fidelity to feeding areas, but animals from multiple feeding areas converge on common winter breeding areas and whales from multiple breeding areas may share feeding areas (Calambokidis *et al*. 2008, Wade 2017). Baker *et al*. (2008) reported significant differences in mtDNA haplotype frequencies among different North Pacific breeding and feeding areas ~~in the North Pacific~~, reflecting strong matrilineal site fidelity to respective migratory destinations. The most significant differences in haplotype frequencies were found between ~~the~~ California/Oregon ~~feeding area~~ and Russian and Southeastern Alaska feeding areas (Baker *et al*. 2013). Among breeding areas, the greatest level of differentiation was found between Okinawa and Central America ~~and most other breeding grounds~~ (Baker *et al*. 2013). Genetic differences between feeding and breeding grounds were also found, even for areas where regular exchange of animals between feeding and breeding grounds is confirmed by photo-identification (Baker *et al*. 2013).

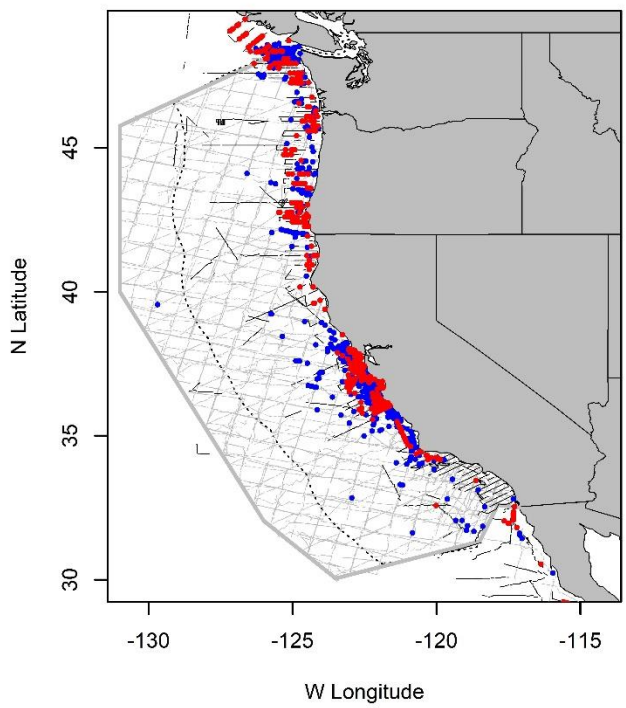


**Figure 1**. Humpback whale sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018. Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

~~Along the U.S. West Coast, NMFS currently recognizes one humpback whale stock that includes two separate feeding groups: (1) a California and Oregon feeding group of whales that includes whales from the endangered Central American and threatened Mexican distinct population segments (DPSs) defined under the ESA (NOAA 2016a), and (2) a northern Washington and southern British Columbia feeding group that primarily includes whales from the threatened Mexican DPS, but also small numbers of whales from the unlisted Hawaii and endangered Central American DPSs (Calambokidis *et al*. 2008, Barlow *et al*. 2011, Wade *et al*. 2016, Wade 2017). Very few~~

~~photographic matches between these feeding groups are documented (Calambokidis *et al*. 2008). Calambokidis *et al*. (2017) report that approximately 70% of whales photographed in the southern Mexico and Central America breeding grounds have been matched to California and Oregon waters. Seven 'biologically important areas' for humpback whale feeding 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). Sightings from large-scale research vessel surveys are largely concentrated near shelf waters (Fig. 1).~~

~~For the Marine Mammal Protection Act (MMPA) stock assessment reports, the~~ The California/Oregon/Washington Stock ~~is~~ as defined ~~to~~ includes humpback whales ~~that feed off the west coast of the United States, including animals from both the~~ from two feeding groups: California-Oregon and Washington-southern British Columbia ~~feeding groups~~ (Calambokidis *et al*. 1996, Calambokidis *et al*. 2008, Barlow *et al*. 2011). The California-Oregon feeding group includes whales from the Mexico and Central America DPSs and it is estimated that most Central America DPS whales use California-Oregon waters for feeding (NOAA 2016a, Wade *et al.* 2016, Wade 2017). The northern Washington and southern British Columbia feeding group includes primarily threatened Mexico DPS whales, with smaller numbers from the unlisted Hawaii DPS and endangered Central America DPS. Seven areas important for humpback whale feeding 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 in Puget Sound (the 'Salish Sea'), a region where they were historically abundant prior to whaling (Calambokidis *et al*. 2017).

Three other humpback whale stocks are recognized in the Pacific region stock assessment reports: (1) Central North Pacific Stock (with feeding areas from Southeast Alaska to the Alaska Peninsula), (2) Western North Pacific Stock (with feeding areas from the Aleutian Islands, the Bering Sea, and Russia), and (3) American Samoa Stock in the South Pacific (with largely undocumented feeding areas as far south as the Antarctic Peninsula). ~~The relationship of MMPA stocks to ESA distinct population segments (DPSs) is complex. Whales from three different DPSs (Central America, Mexico, and Hawaii) are included in the MMPA stock identified in this report as the "California/Oregon/Washington Stock". Most humpbacks that feed in California and Oregon waters in summer originate from the threatened Mexico DPS, while a much smaller fraction originate from the endangered Central American DPS (Wade *et al*. 2016, Wade 2017). In Washington and southern British Columbia waters, all three DPSs (Hawaii, Mexico, and Central America) occur.~~

~~Appropriate risk assessment for both MMPA stocks and ESA DPSs requires a combination of appropriate stock delineation and PBR calculation under the MMPA, which is currently pending. Also, assignment of observed and estimated anthropogenic impacts to the appropriate DPS and MMPA stocks will be required where possible (NMFS 2016).~~

**POPULATION SIZE**

Calambokidis and Barlow (2020) estimated humpback whale abundance for the U.S. West Coast based on updated survey data through 2018, using mark-recapture methods consistent with past estimates (Calambokidis and Barlow 2004, 2013, 2017). Data from a 2018 line-transect and photo-ID survey represents the most comprehensive effort with respect to spatial coverage and sample sizes to date along the U.S. West Coast (Calambokidis and Barlow 2020, Henry *et al*. 2020). The best estimate of current abundance is based the most-recent 4 years (2015-2018) of mark-recapture data and a Chao model that accounts for heterogeneity of capture probabilities, resulting in an estimate of 4,973 (CV=0.048) whales (Calambokidis and Barlow 2020). This estimate is calculated using identifications from California and Oregon waters, but the authors note that it likely includes a smaller number of whales from Washington state waters since there is interchange with that area. Becker *et al*. (2020) also estimated humpback whale abundance for this region based on habitat models and 1991-2018 line-transect data and the most-recent estimate for 2018 is 4,784 whales (CV=0.31). However, the mark-recapture estimate is considered the best estimate because 1) it has better precision and 2) the line-transect estimate reflects only whale densities within the study area during summer and autumn when surveys were conducted. ~~Based on whaling data, the pre-1905 population of humpback whales in the North Pacific was estimated at 15,000 (Rice 1978), but whaling reduced this population to approximately 1,200 whales by 1966 (Johnson and Wolman 1984). A 2004 to 2006 photo-identification study estimated humpback whale abundance in the entire Pacific Basin at 21,808 (CV=0.04) (Barlow *et al*. 2011). Barlow (2016) estimated 3,064 (CV=0.82) humpback whales from a 2014 summer/fall ship line-transect survey of California, Oregon, and Washington waters. Line-transect estimates of humpback whales in this region have less precision than corresponding estimates from mark-recapture studies, and for that reason, estimates of population size for this stock are based on mark-recapture estimates detailed below.~~

~~Abundance estimates from photographic mark-recapture surveys in California and Oregon waters every year~~

~~from 1991 through 2014 represent the most precise estimates (Calambokidis *et al*. 2017). These estimates include only whales photographed in California and Oregon waters and exclude whales from the Washington state and southern British Columbia feeding group (Calambokidis *et al*. 2009, 2017a). California and Oregon estimates range from 1,400 to 2,400 animals, depending on the choice of mark-recapture model and sampling period (Fig. 2). The best estimate of abundance for California and Oregon waters is the 2011 to 2014 Chao estimate of 2,374 (CV=0.03) whales. This estimate is considered the best of those mark-recapture estimates reported because it accounts for individual capture heterogeneity (Calambokidis *et al*. 2017). This estimate includes virtually the entire Central American DPS, which, depending on choice of mark-recapture model, was estimated to include between 431 (CV=0.34) and 783 (CV=0.17) whales, based on 2004 to 2006 photographic mark-recapture data (Wade *et al*. 2016, Wade 2017). However, abundance estimates for the Central American DPS are ≥ 8 years old and are not considered reliable estimates of current abundance (NOAA 2016b).~~

~~Calambokidis *et al*. (2017) estimated the northern Washington and southern British Columbia feeding group population size to be 526 (CV=0.23) animals based on 2013 and 2014 mark-recapture data.~~

~~Combining abundance estimates from both the California/Oregon and Washington/southern British Columbia feeding groups (2,374 + 526) yields an estimate of 2,900 animals for the California/Oregon/Washington stock. A coefficient of variation for both feeding groups combined can be calculated as a weighted-mean CV of the 2 estimates, or $CV_{N1+N2}$ = sqrt(($CV_1$*N1)$^2$ + ($CV_2$*N2)$^2$ ) / (N1+N2) or CV = 0.048.~~

**Minimum Population Estimate**

The minimum population estimate for humpback whales in the California/Oregon/Washington stock is taken as the lower 20th percentile of the ~~log-normal distribution of the combined mark-recapture estimate for both feeding groups given above, or 2,784~~ mark-recapture estimate based on 2015-2018 data, or 4,776 whales (Calambokidis and Barlow 2020).

**Current Population Trend**

~~Ship surveys indicate that humpback whale abundance probably increased in California waters between 1979/80 and 1991 (Barlow 1994) and between 1991 and 2014 (Barlow 2016), but this increase was not linear, and short-term declines were apparent in 2001 and 2008. Mark-recapture population estimates had shown a long-term increase of approximately 8% per year (Calambokidis *et al*. 2009, Fig. 2), but more recent estimates suggest a possible leveling-off of the population size (Fig. 2), depending on the choice of model and time frame used (Calambokidis and Barlow 2013, Calambokidis *et al*. 2017). Population estimates for the entire North Pacific have also increased substantially from 1,200 in 1966 to approximately 18,000 to 20,000 whales in 2004 to 2006 (Calambokidis *et al*. 2008). Although these estimates are based on different methods and the earlier estimate is extremely uncertain, the growth rate implied by these estimates (6-7%) is consistent with the growth rate of the California/Oregon/Washington stock.~~ Calambokidis and Barlow (2020) report that humpback whale abundance appears to have increased within the California Current at approximately 7.5% annually since the late 1980s (Figure 2). 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). Calambokidis and Barlow (2020) note that the apparent increase in abundance from 2014 to 2018 is too great to represent real population growth and may reflect negatively-biased estimates during 2009 to 2014 due to less representative sampling compared with 2018.

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

The proportion of calves in the California/Oregon/Washington stock from 1986 to 1994 appeared much lower than previously measured for humpback whales in other areas (Calambokidis and Steiger 1994), but between 1995 and 1997 a greater proportion of calves were identified, and the 1997 reproductive rates for this population were closer to those reported for other humpback whale populations (Calambokidis *et al*. 1998). Despite the apparently low proportion of calves, two independent lines of evidence indicate that this stock was growing in the 1980s and early 1990s (Barlow 1994; Calambokidis *et al*. 2003) with a best estimate of 8% growth per year (Calambokidis *et al*. 1999), which is taken as the maximum net productivity rate for this stock. The current net productivity rate is unknown.

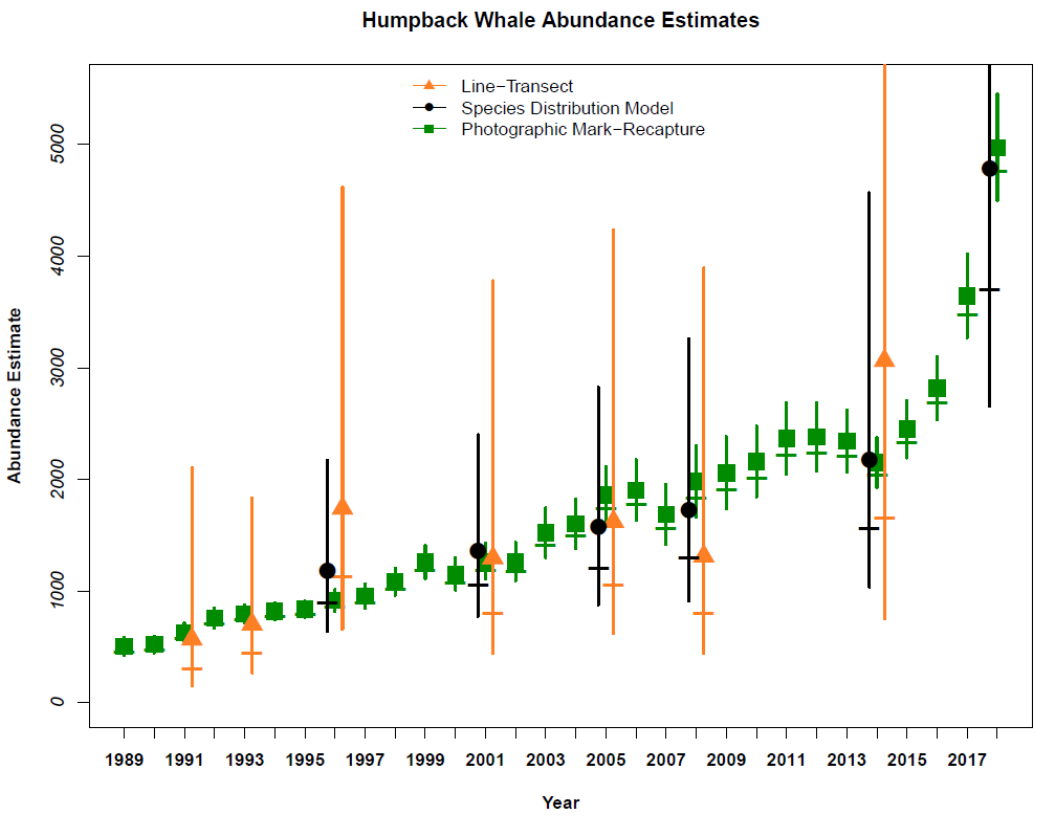


**Figure 2.** Estimated abundance of humpback whales off California and Oregon based on line-transect surveys (Barlow 2016), habitat-based species distribution models (Becker *et al.* 2020) and photographic mark-recapture surveys (Calambokidis and Barlow 2020). Mark-recapture estimates are based on a Chao model that uses rolling 4-year periods and accounts for heterogeneity of capture probability. Horizontal hatch marks represent minimum population size estimates based on 20th 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 provide the best display in variability in mean estimates between methods. Upper 95% confidence limits for line-transect surveys in 2014 and species distribution models in 2018 not visible in this plot were approximately 12,000 and 8,600 whales respectively.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (~~2,784~~ 4,776) times one half the estimated population growth rate for this stock of humpback whales (½ of 8%) times a recovery factor of 0.3 (for an endangered species~~;~~ with $N_{min}$ > 1,500 and CV($N_{min}$) < 0.50, Taylor *et al.* 2003), resulting in a PBR of ~~33.4~~ 57.3. Because this stock spends approximately half its time outside the U.S. Exclusive Economic Zone (EEZ), the PBR in U.S. waters is ~~16.7~~ 28.7 whales per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

A total of ~~138~~ 172 human-related interactions involving humpback whales are summarized for ~~the five-year period of 2013 to 2017~~ 2015 to 2019 by Carretta *et al.* (~~2019a~~ 2021). These records include serious injuries, non-serious injuries, and mortality involving pot/trap fisheries (n=~~62~~ 81), unidentified fishery interactions (~~56~~ 68), gillnet fisheries (~~4~~ 11), vessel strikes (~~14~~ 10), hook and line fisheries (1) and marine ~~moorings~~ debris (1). The number of serious injuries and mortalities in each category are summarized below. In addition to interactions with humpback whales, ~~20~~ 16 entanglements involving 'unidentified whales' ~~(totaling 15.5 serious injuries and mortalities)~~ occurred from ~~2013 to 2017~~ 2015 to 2019, some of which were certainly humpback whales. These 16 unidentified whale injuries are prorated to species using the method in Carretta (2018) and result in an additional ≥ 2.0 humpback whale serious injuries annually (Table 1). ~~(Carretta 2018, Carretta *et al.* 2019a). The number of human-related deaths and injuries for each humpback whale feeding group are unknown, but based on the proportion of the overall abundance (2,900~~

~~whales) belonging to the California-Oregon (82%) and Washington and southern British Columbia (18%) feeding groups, a majority of cases likely involve whales from the California-Oregon feeding group that includes nearly all of the Central American DPS (Calambokidis *et al*. 2017).~~

**Fishery Information**

Pot and Trap Fisheries

Pot and trap fishery entanglements are the most frequently-documented source of serious injury and mortality of humpback whales in U.S. west coast waters and reported entanglements increased considerably in 2014 (Carretta *et al*. 2013, 2019a). From ~~2013 to 2017~~ 2015 to 2019, ~~62~~ 81 observed interactions with pot and trap fisheries were observed (Carretta *et al.* ~~2019a~~ 2021). ~~Two~~ Four pot/trap fishery records ~~include serious injuries totaling 1.75 whales in recreational Dungeness crab pot gear~~ involved recreational fisheries and one record ~~includes a serious injury in~~ involved Washington tribal crab pot gear, which are excluded from Table 1 commercial fishery totals, but are summarized in the 'Other Mortality' section of this report. Commercial fishery serious injuries and mortalities in the remaining 76 cases involving pot/trap gear total 52.75 whales for 2015 to 2019 (Table 1 and Carretta *et al*. 2021). ~~Nineteen records involved non-serious injuries resulting from human intervention to remove gear, or cases where animals were able to free themselves. Three records involved dead whales, including one humpback recovered in sablefish pot gear in Oregon, one case where severed humpback flukes were found entangled in California Dungeness crab gear (whale presumed dead) and a third case of an entangled humpback in California Dungeness crab gear in declining health that was detected over several months, before eventually being found dead without attached gear. This was a well-marked individual that was readily identifiable by the whale watching community (Carretta *et al.* 2019a). The remaining 37 pot/trap fishery injury cases, once evaluated per the NMFS serious injury policy, resulted in a total of 30.75 serious injuries / 5 years, or 6.15 humpback whales annually (Table 1). Documented five year mortality, serious injury, plus prorated injury totals (*i.e.* entangled humpback whales with an injury score < 1) for pot/trap fisheries, in order of frequency are: California Dungeness crab pot (19.25), unidentified pot/trap fishery (7.0), Washington/Oregon/California sablefish pot fishery (2.5), California spot prawn (2.5), Washington Dungeness crab pot (1.75), and Oregon Dungeness crab pot (0.75) (Table 1).~~ The totals ~~above~~ in Table 1 represent minimum observed cases from opportunistic at-sea sightings or stranding records, except for bycatch estimates based on systematic observer program data. It is recognized that entanglement totals do not represent all cases due to incomplete detection of incidents and no method is currently available to correct for undetected entanglements. ~~An effort is made where possible to account for this negative bias. For example, total entanglement mortality and serious injury in the WA/OR/CA sablefish pot fishery is parsed out into statistical estimates from observer program data and opportunistically detected records derived from strandings and at-sea sightings linked to the same fishery. In this case, the annual statistical bycatch estimates and at-sea and stranding observations are nearly-equal, despite the fact that the latter category is uncorrected for undetected cases. In commercial pot and trap fisheries, the mean annual mortality and serious injury between 2013 and 2017 is the sum of observer program derived estimates (9.5), plus opportunistically detected animals (32.75) = 42.25 whales / 5 years = 8.45 whales.~~

Gillnet and Unidentified Fisheries

Gillnet (n= ~~4~~ 11) and unidentified fisheries (n= ~~56~~ 68) accounted for ~~60~~ 79 humpback whale interactions ~~between 2013 and 2017~~ from 2015 to 2019 (Carretta *et al.* ~~2019a~~ 2021). Based on the proportion of humpback whale records where the type of fishing gear is positively identified, it is likely that most cases involving ~~'~~unidentified fisheries~~'~~ represent pot and/or trap gear (Carretta *et al*. ~~2019a~~ 2021). Of the 11 gillnet-related interactions, three were attributed to tribal fisheries (see Other Mortality section). Commercial fishery serious injuries and mortalities due to gillnet (3.5) plus unidentified fishery interactions (52.25) total 55.75 whales for 2015 to 2019 (Table 1). ~~Three records involved dead whales. The remaining 57 records, once evaluated per the NMFS serious injury policy, resulted in six non-serious injuries and 41.25 serious injuries. The total annual mortality and serious injury due to unidentified and gillnet fisheries is the sum of observed deaths (3) and serious injuries (41.25), or 44.25 whales. The five-year annual mean serious injury and mortality due to gillnet and unidentified fisheries during 2013 to 2017 is therefore 44.25 / 5 = 8.85 whales.~~

Three humpback whale entanglements (all released alive) were observed in the CA swordfish drift gillnet fishery from ~~8,956~~ 9,158 fishing sets monitored between 1990 and ~~2017~~ 2019 (Carretta *et al.* ~~2019b~~ 2021). Some ~~opportunistic~~ at-sea sightings of free-swimming humpback whales entangled in gillnets may originate from this fishery. The most recent ~~model-based~~ estimate of humpback whale ~~bycatch~~ serious injury and mortality in this fishery for ~~2013 to 2017~~ 2015 to 2019 is ~~0.2~~ 0.1 whales (CV=~~1.2~~ 4.6)~~, but due to three of four observations resulting in non-~~

~~serious injury releases from gear, it is estimated that only one-quarter of these entanglements represent serious injuries~~ (Carretta *et al*. ~~2019b~~ 2021). ~~The corresponding ratio estimate of bycatch for the same time period is zero (Carretta *et al*. 2019b). The model-based estimate is considered superior because it utilizes all 28 years of data for estimation, in contrast to the ratio estimate that uses only 2013 to 2017 data.~~ The average annual estimated serious injury and mortality in the CA swordfish drift gillnet fishery is <0.1 whales.

**Table 1.** Observed and estimated incidental mortality and serious injury of humpback whales (California/Oregon/Washington stock) in commercial fisheries that are likely to take this species (Carretta 2021, Carretta *et al*., ~~2019a~~ 2021, ~~2019b,~~ Jannot *et al*. 2018). Mean annual takes are based on ~~2013 to 2017~~ 2015 to 2019 data unless noted otherwise. Serious injuries may include prorated serious injuries with values less than one (NOAA 2012), thus the sum of serious injury and mortality may not be a whole number.

| Fishery Name | Year(s) | Data Type | Percent Observer Coverage | Observed Mortality + Serious Injury | Estimated mortality and serious injury (CV) | Mean Annual mortality and serious injury (CV) |
|---|---|---|---|---|---|---|
| **WA/OR/CA Sablefish Pot** | 2012<br>2013<br>2014<br>2015<br>2016 | observer | 35%<br>14%<br>31%<br>61%<br>71% | 0<br>0<br>1<br>0<br>0 | 0.12 (n/a)<br>0.19 (n/a)<br>1.15 (n/a)<br>0.08 (n/a)<br>0.06 (n/a) | 0.32 (n/a) |
| ~~**WA/OR/CA Sablefish Pot**~~ | ~~2013-2017~~ | ~~Strandings / sightings~~ | ~~n/a~~ | ~~0 + 1.5~~ | ~~n/a[1]~~ | ~~≥ 0.30 (n/a)~~ |
| **Open Access Fixed Gear Pot** | 2012<br>2013<br>2014<br>2015<br>2016 | observer | 7%<br>9%<br>8%<br>6%<br>7% | 0<br>0<br>0<br>0<br>0 + 0.75 | 1.12 (n/a)<br>0.67 (n/a)<br>1.3 (n/a)<br>2.03 (n/a)<br>2.76 (n/a) | 1.58 (n/a) |
| **CA swordfish and thresher shark drift gillnet fishery** | ~~2013-2017~~<br>2015-2019 | observer | ~~23%~~<br>21% | 0[2] | ~~<0.1 (1.9)~~<br>0.1 (4.6) | ~~<0.1 (1.9)~~<br>< 0.1 (4.6) |
| **CA halibut/white seabass and other species large mesh (≥3.5") set gillnet fishery** | ~~2013-2017~~<br>2017 | observer | ~~<~~~10% | 0 | 0 | 0 (n/a) |
| **CA spot prawn pot** | ~~2013-2017~~<br>2015-2019 | Strandings / sightings | n/a | 0 + 2.5 | n/a | ≥ 0.50 (n/a) |
| **Unspecified pot or trap fisheries (includes generic 'Dungeness' crab gear not attributed to a specific state fishery)** | ~~2013-2017~~<br>2015-2019 | Strandings / sightings | n/a | ~~0 + 7.0~~<br>1 + 11 | n/a | ≥ ~~1.4~~<br>2.4 (n/a) |
| **CA Dungeness crab pot** | ~~2013-2017~~<br>2015-2019 | Strandings / sightings | n/a | ~~2 + 17.25~~<br>5 + 22.75 | n/a | ≥ ~~3.85~~<br>5.55 (n/a) |
| **OR Dungeness crab pot[3]** | ~~2013-2017~~<br>2015-2019 | Strandings / sightings | n/a | ~~0 + 0.75~~<br>0 + 1.75 | n/a | ≥ ~~0.15~~<br>0.35 (n/a) |
| **WA Dungeness crab pot** | ~~2013-2017~~<br>2015-2019 | Strandings / sightings | n/a | ~~0 + 1.75~~<br>1 + 5.25 | n/a | ≥ ~~0.35~~<br>1.25 (n/a) |
| **unidentified fisheries (includes 'unidentified gillnet') involving identified humpback whales** | ~~2013-2017~~<br>2015-2019 | Strandings / sightings | n/a | ~~3 + 41.25~~<br>2 + 53.75 | n/a | ≥ ~~8.85~~<br>11.15 (n/a) |
| **Unidentified fishery interactions involving *unidentified whales* prorated to humpback whale** | 2015-2019 | Strandings / sightings | n/a | 0 + 10.0 | n/a | ≥ 2.0 |
| **Total Annual Takes** | | | | | | ~~≥ 17.3~~<br>≥ 25.2 (n/a) |

Unidentified whales represent approximately 15% of entanglement cases along the U.S. West Coast,

~~[1] No estimate of total bycatch has been generated for this fishery.~~

~~[2] There were no observations of humpback whales in this fishery during 2013-2017, but the model-based estimate of bycatch for this period results in a positive estimate of bycatch (Carretta *et al*. 2019b).~~

~~[3] There were 3 additional non-serious injuries involving humpback whales that were successfully disentangled from OR Dungeness crab pot gear between 2013 and 2017; these would have been serious injuries without the disentanglement response.~~

(Carretta 2018). Observed entanglements may lack species identifications (IDs) due to rough seas, distance from whales, or a lack of cetacean identification expertise. In older stock assessments, these unidentified entanglements were not assigned to species, which results in underestimation of entanglement risk, especially for commonly-entangled species. To remedy this negative bias, a cross-validated species identification model was developed from known-species entanglements ~~('model data')~~. The model is based on several variables (location + depth + season + gear type + sea surface temperature) found to be statistically-significant predictors of known-species entanglement cases (Carretta 2018). The species model was used to assign species ID probabilities for ~~20~~ 16 unidentified whale entanglement cases ~~('novel data')~~ from ~~2013 to 2017~~ 2015 to 2019 (Carretta 2018). The sum of species assignment probabilities for ~~this five-year period~~ 2015 to 2019 result in an additional ~~13.7~~ 13.3 humpback whale entanglements ~~for 2013 to 2017~~. 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 ~~13.7~~ 13.3 x 0.75 = ~~10.3~~ 10 additional humpback serious injuries are represented from the ~~20~~ 16 unidentified whale entanglement cases, or ~~2.1~~ 2.0 humpback whales annually.

Total commercial fishery serious injury and mortality of humpback whales from ~~2013 to 2017~~ 2015 to 2019 is the sum of pot/trap fishery records and estimates from Table 1 (~~42.25~~ 59.7), plus gillnet and unidentified fishery records (~~44.25~~ 55.85), plus prorated unidentified whale entanglements (~~10.3~~ 10.0), or ~~96.8~~ 125.6 total whales. The mean annual serious injury and mortality (observed and estimated) from commercial fisheries is ~~96.8~~ 125.6 whales / 5 years = ~~19.4~~ 25.2 whales from ~~2013 to 2017~~ 2015 to 2019 (Table 1). Most serious injury and mortality records from commercial fisheries reflect opportunistic stranding and at-sea sighting data and thus, represent minimum counts of impacts, for which no correction factor is currently available.

Despite an ~~overall~~ increase in the number of reported entanglements in recent years, increasing efforts to disentangle humpback whales ~~from fisheries~~ has led to an increase in the fraction of ~~cases~~ injuries reported as non-serious ~~injuries~~, due to the removal of gear from humpback whales that otherwise appear healthy. In the absence of human intervention, these records would have represented at least ~~11.5~~ 10 additional serious injuries ~~over the five-year period 2013 to 2017~~ from 2015 to 2019, or an additional ~~2.3~~ 2.0 humpback whales annually (Carretta *et al.* ~~2019a~~ 2021).

**Vessel Strikes**

~~Fourteen~~ Ten humpback whales (totaling ~~eight~~ 7 deaths, ~~2.8~~ and 1.8 serious injuries~~, and two non-serious injuries~~) were reported struck by vessels between ~~2013 and 2017~~ 2015 and 2019 (Carretta *et al.* ~~2019a~~ 2021). ~~The observed a~~ Average annual serious injury and mortality of humpback whales due to observed vessel strikes is ~~2.2~~ 1.76 whales per year (~~eight~~ 7 deaths, plus ~~2.8~~ 1.8 serious injuries = ~~10.8~~ 8.8 / 5 years). Vessel strike mortality was estimated for humpback 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), and whale movement patterns obtained from satellite-tagged animals ~~in the region~~ to estimate whale/vessel interactions that would result in mortality. 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 ~~time of year~~ season that overlaps with cetacean habitat models generated from line-transect surveys (Becker *et al*. 2016, Rockwood *et al*. 2017). This estimate was based on an assumption of a moderate level of vessel avoidance (55%) by humpback whales, as measured by the behavior of satellite-tagged whales in the presence of vessels (McKenna *et al*. 2015). The estimated mortality of 22 humpback whales annually due to vessel strikes represents approximately ~~0.7~~ 0.4% of the estimated population size of the stock (22 deaths / ~~2,900~~ 4,973 whales). The results of Rockwood *et al*. (2017) also include a no-avoidance encounter model that results in a worst-case estimate of 48 annual humpback whale vessel strike deaths, representing ~~1.6~~ 0.9% of the estimated population size. The authors also note that 82% of humpback 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. The number of vessel strikes attributable to each ~~breeding ground~~ DPS (Central America, Hawaii, Mexico) is unknown. Using the moderate level of avoidance model from Rockwood *et al.* (2017), estimated vessel strike deaths of humpback whales are 22 per year. A comparison of average annual vessel strikes observed ~~over the period~~ during ~~2013 to 2017 (2.2/yr)~~ 2015 to 2019 (1.76/yr) *versus* estimated vessel strikes (22/yr) indicates that the detection rate of ~~reporting for~~ humpback whale vessel strikes is ~~approximately~~ ≤10%.

Vessel strikes within the U.S. West Coast EEZ continues to be a threat to all large whale populations (Redfern *et al*. 2013; 2019; Moore *et al*. 2018). However, ~~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 ~~their~~ speed when ~~they transited~~ transiting longer routes to avoid the ECAs. Further research is ~~necessary~~ 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.

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

~~A humpback whale was entangled in a research marine mooring buoy in 2014. The whale is estimated to have been entangled for three weeks and had substantial necrotic tissue around the caudal peduncle. Although the whale was fully disentangled, this animal was categorized as a serious injury because of the necrotic condition of the caudal peduncle and the possibility that the whale would lose its flukes due to the severity of the entanglement (NOAA 2012, Carretta *et al*. 2019a). Additionally, two humpback whales were entangled in recreational Dungeness crab pot gear, resulting in a total of 1.75 serious injuries (Carretta *et al*. 2019a). One humpback whale was entangled and seriously injured in Washington tribal crab pot gear in 2017 (Carretta *et al*. 2019a). The total number of serious injuries from marine moorings (1), recreational fisheries (1.75), and tribal fisheries (1) from 2013 to 2017 is 3.75 whales, or 0.75 whales annually.~~ Carretta *et al*. (2021) summarize non-commercial serious injury and mortality totals during 2015-2019 for the following sources: recreational Dungeness crab pot (1.75), hook and line fisheries (0.75), marine debris (1.00), tribal gillnet (2.50), and Washington state tribal Dungeness crab pot (1.00). The sum of these non-commercial sources is 7 whales, or 1.4 whales annually for 2015 to 2019.

**Habitat Concerns**

Increasing levels of anthropogenic sound in the world's oceans (Andrew *et al*. 2002), such as those produced by shipping traffic, or LFA (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).

**STATUS OF STOCK**

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), presumably from this stock. 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. As a result of commercial whaling, humpback 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 (ESA) in 1973. The humpback whale ESA listing final rule (81 FR 62259, September 8, 2016) established 14 distinct population segments (DPSs) with different listing statuses. The CA/OR/WA humpback whale stock primarily includes whales from the endangered Central American DPS and the threatened Mexico DPS, plus a small number of whales from the non-listed Hawaii DPS. Humpback whale stock delineation under the MMPA is currently under review, and until this review is complete, the CA/OR/WA stock will continue to be considered endangered and depleted for MMPA management purposes (e.g., selection of a recovery factor, stock status). Consequently, the California/Oregon/Washington stock is automatically considered as a "strategic" stock under the MMPA. ~~The observed annual mortality and serious injury due to commercial fishery entanglements in 2013 to 2017 (17.3/yr) (Table 1), non-fishery entanglements (0.2/yr), recreational crab pot fisheries (0.35/yr), tribal fisheries (0.2/yr), serious injuries assigned to unidentified whale entanglements (2.1/yr), plus observed ship strikes (2.2/yr), equals 22.35 animals, which exceeds the PBR in U.S. waters of 16.7 animals. *Estimated* vessel strike deaths are 22 humpback whales annually (Rockwood *et al*. 2017), but this does not include vessel strikes that occur outside of the U.S. West Coast EEZ. Using this estimate of vessel strike deaths instead of the observed 2.2/yr observed value noted above, the total~~ Total annual human-caused serious injury and mortality of humpback whales is the sum of commercial fishery (~~17.3~~ 25.2/yr) + ~~recreational fishery (0.35) + tribal fishery (0.2/yr) + non-fishery entanglements (0.2/yr) + serious injuries assigned to unidentified whale entanglements (2.1/yr)~~ non-commercial sources (1.4/yr) + estimated vessel strikes (22/yr), or ~~42.1~~ 48.6 humpback whales annually. This exceeds the ~~range-wide~~ PBR estimate of ~~33.4~~ 28.7 humpback whales. Other than the vessel strike estimates, most data on human-caused serious injury and mortality for this population is based on opportunistic stranding and at-sea sighting data and

represents a minimum count of total impacts. There is currently 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. Based on strandings and at-sea observations, annual humpback whale mortality and serious injury in commercial fisheries (~~17.3~~ 25.2/yr) ~~exceeds~~ is less than the PBR of 28.7; ~~therefore,~~ however, if methods were available to correct for undetected serious injury and mortality, total fishery mortality and serious injury would likely exceed PBR. ~~is not approaching zero mortality and serious injury rate.~~ Observed and assigned levels of serious injury and mortality due to commercial fisheries (≥ 25.2) exceed 10% of the stock's PBR (28.7), thus, commercial fishery take levels are not approaching zero mortality and serious injury rate. Despite impacts of anthropogenic-related serious injury and mortality of humpback whales along the U.S. West Coast, Calambokidis and Barlow (2020) suggest that the number of humpback whales in the region has been increasing at roughly 7.5% annually since the late 1980s. ~~The California/Oregon/Washington stock showed a long-term increase in abundance from 1990 through approximately 2008 (Figure 2), but more recent estimates through 2014 indicate a leveling-off of the population size (Calambokidis *et al.* 2017).~~

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Revised 8/6/2021 ~~04/15/2020~~

# 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.

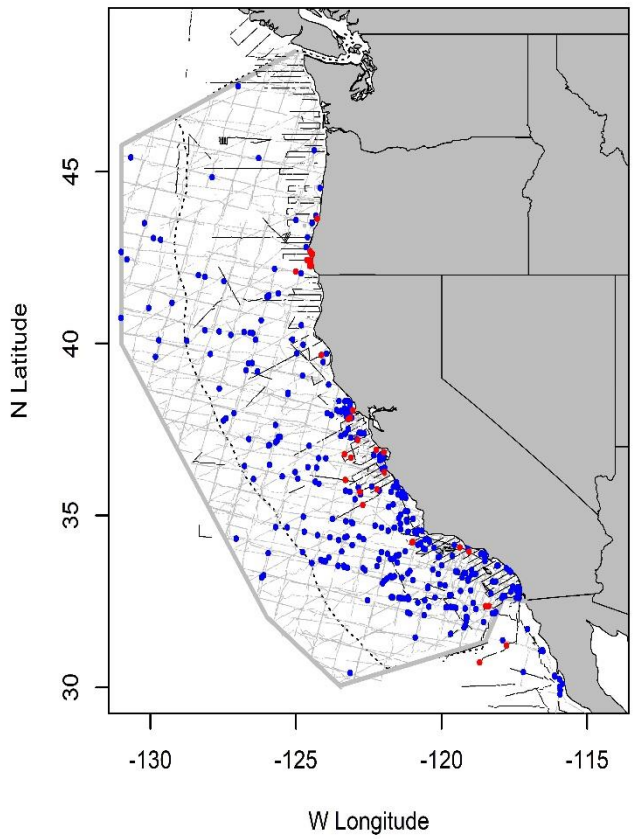


**Figure 1**. Blue whale sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018. Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

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 ~~'biologically~~ important areas~~'~~ for blue whale feeding ~~are~~ 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 ~~remained stable~~ (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. ~~An analysis of line-transect survey data from 1996-2014 provided a range of blue whale estimates from a high of approximately 2,900 whales in 1996 to a low of 900 whales in 2008 (Barlow 2016). Photographic mark-recapture estimates of abundance from 2005 to 2011 range from 1,000 to 2,300 whales, with the most consistent estimates represented by a four-year sampling period Chao model that incorporates individual capture heterogeneity (Calambokidis and Barlow 2013). The Chao model consistently yielded estimates of approximately 1,500 whales (Fig. 2), with 1,647 (CV=0.07) whales estimated for the 2008-2011 period (Calambokidis and Barlow 2013). This estimate is now over 8 years old and is considered outdated (NMFS 2016). The most recent abundance estimate is 1,496 (CV=0.44) whales, based on the 2014 line-transect survey within the California Current (Barlow 2016).~~

**Minimum Population Estimate**

The minimum population estimate of blue whales is ~~taken~~ calculated as the lower 20th percentile of the ~~log-normal distribution of the 2014 line-transect abundance~~ 2018 mark-recapture estimate, or ~~1,050~~ 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 is ~~no~~ may be evidence of a population size increase ~~in this blue whale population~~ since the ~~early~~ 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, ~~explains~~ 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.

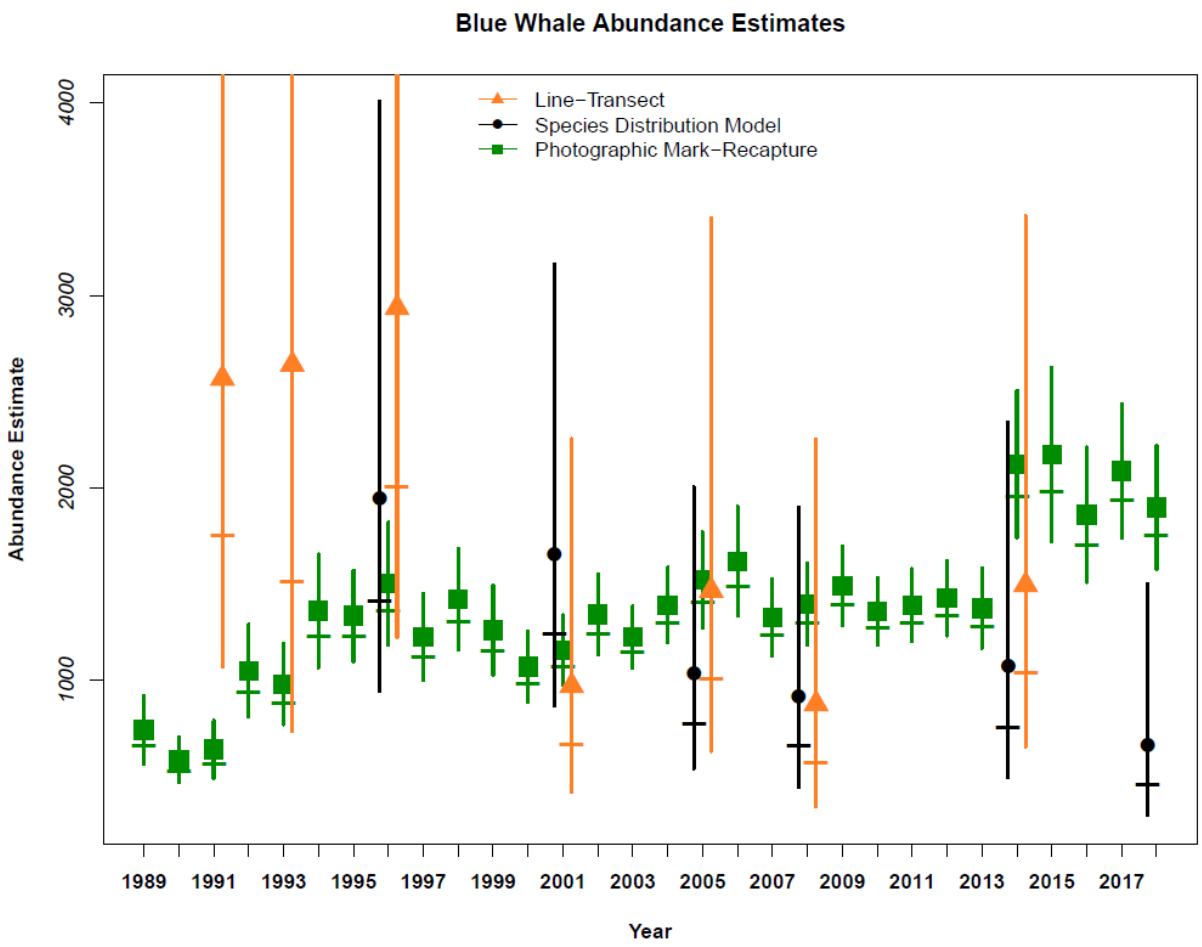


**Figure 2**. Estimated abundance of blue whales based on three methods (standard vessel-based line transect surveys, habitat-based species distribution models, and a photographic mark-recapture model). The line-transect estimates are based on surveys reported by Barlow (2016). Species distribution model 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 (~~1,050~~ 1,767) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of ~~0.1~~ 0.2 (for an endangered species with a minimum abundance ~~less~~ greater than 1,500 and unknown population trend), resulting in a PBR of ~~2.1~~ 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 ~~1.23~~ 4.1 whales per year. ~~NMFS guidelines for preparing marine mammal stock assessments note that "In transboundary situations where a stock's range spans international boundaries or the boundary of the U.S. Exclusive Economic Zone (EEZ), the best approach is to establish an international management agreement for the species and to evaluate all sources of human-caused mortality and serious injury (U.S. and non-U.S.) relative to the PBR for the entire stock range. In the interim, if a transboundary stock is migratory and it is reasonable to do so, the fraction of time the stock spends in U.S. waters should be noted, and the PBR for U.S. fisheries should be apportioned from the total PBR based on this fraction." (NMFS 2016). The latter approach is taken here, as data on serious injury and mortality for this stock in international waters is unavailable.~~

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fisheries Information

~~Two blue whales were seriously injured in California Dungeness crab pot gear and a third whale was seriously injured in an unidentified pot/trap fishery during the most recent 5-year period of 2013 to 2017 (Carretta *et al*. 2019a). Five additional prorated serious injuries were observed during the same period, including one in the California Dungeness crab fishery and four in unidentified fishing gear (Table 1). There have been no observed entanglements of blue whales in the California swordfish drift gillnet fishery during a 28-year observer program that includes 8,956 observed fishing sets from 1990 to 2017 (Carretta *et al*. 2019b). However, some gillnet mortality of large whales may go unobserved because whales swim away with a portion of the net. The total observed serious injury and mortality due to commercial fisheries from 2013 to 2017 is 6.75 whales, or 1.35 whales annually.~~

Blue whales are occasionally been 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 ~~2013-2017~~ 2015-2019 is the sum of observed annual entanglements (1.35/yr), plus species probability assignments (Carretta 2018) from 16 unidentified whale entanglements (~~0.09~~ 0.04/yr), totaling ~~1.44~~ 1.39 blue whales annually (Table 1).~~This represents~~ 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.

~~Unidentified whales represent approximately 15% of entanglement cases along the U.S. West Coast, (Carretta 2018). Observed entanglements may lack species identification (IDs) due to rough seas, distance from whales, or a lack of cetacean identification expertise. In older stock assessments, these unidentified entanglements were not assigned to species, resulting in underestimation of entanglement risk, especially for commonly entangled species. To remedy this negative bias, a cross-validated species identification model was developed from known-species entanglements ('model data'). The model is based on several variables (location + depth + season + gear type + sea surface temperature) found to be statistically significant predictors of known species entanglement cases (Carretta 2018). The species model was used to assign species ID probabilities for 20 unidentified whale entanglement cases ('novel data') from 2013-2017. Species probability assignments resulted in an additional 0.46 additional blue whale entanglements, or 0.09 blue whales annually.~~

**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*. ~~2019a~~ 2021, ~~2019b~~ Carretta 2021). 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 | Percent 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** | ~~2013-2017~~ 2015-2019 | Strandings + sightings | n/a | ~~0 (2.75)~~ 0 + 2 | n/a | ≥ ~~0.55~~ 0.4 (n/a) |
| **Unidentified pot/trap fishery** | ~~2013-2017~~ 2015-2019 | Strandings + sightings | n/a | ~~0 (2.5)~~ | n/a | ≥ ~~0.50~~ 0.2 (n/a) |

| | | | | | | |
|---|---|---|---|---|---|---|
| | | | | 0 + 1 | | |
| **Unidentified fishery interactions involving identified blue whales** | ~~2013-2017~~ 2015-2019 | Strandings + sightings | n/a | ~~0 (1.5)~~ 0 + 3.75 | n/a | ≥ ~~0.3~~ 0.75 (n/a) |
| **Unidentified fishery interactions involving *unidentified whales* prorated to blue whale** | 2015-2019 | Strandings + Sightings | n/a | n/a | 0.2 | ≥ 0.04 |
| **CA/OR thresher shark/swordfish drift gillnet fishery** | ~~2013-2017~~ 2015-2019 | observer | ~~23%~~ 21% | 0 | 0 | 0 (n/a) |
| **Total Annual Takes** | | | | | | ≥ ~~1.35~~ 1.39 (n/a) |

**Vessel Strikes**

~~Two~~ Four blue whale vessel strike deaths were observed during ~~2013-2017~~ 2015-2019 (Carretta *et al*. ~~2019a~~ 2021), resulting in an observed annual average of ~~0.4~~ 0.8 vessel strike deaths. ~~There were no reported ship-strike related serious injuries during this period (Carretta *et al.* 2019a).~~ 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 ~~12~~ 14 blue whales and ~~4~~ 7 unidentified whales (Carretta *et al*. 2013, ~~2019a~~ 2021). ~~No methods have been developed~~ 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 the seasonal distribution of blue whales is in close proximity 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 17%), highlighting that observed numbers are unrepresentative of true impacts (Kraus *et al*. 2005, 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 ~~1,496~~ 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 ~~2.4~~ 2.1% of the estimated population size. The authors also 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 ~~2013-2017~~ 2015-2019 (~~0.4~~ 0.8/yr) versus estimated vessel strikes (18/yr) indicates that the rate of detection for blue whale vessel strikes is approximately ~~2~~ 4%. 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 20th 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 continue to be a threat to all large whale populations (Redfern *et al*. 2013; 2019; Moore *et al*. 2018). However, ~~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 their speed when they transited longer routes to avoid the ECAs. Further research is ~~necessary~~ 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.

**Habitat Issues**

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.

**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 a current analysis suggesting that the Eastern North Pacific population is at 97% of carrying capacity (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 ~~rate from ship strikes (0.4/yr) and~~ commercial fisheries (≥ ~~1.44~~ 1.39 /yr), plus estimated vessel strikes (18/yr), ~~totals 1.84~~ is 19.4 whales annually ~~from 2013-2017~~ for 2015-2019. This exceeds the calculated PBR of ~~1.23~~ 4.1 for this stock ~~of blue whales~~. ~~Furthermore, observations alone are not representative of impacts due to incomplete detection of vessel strikes and fishery entanglements, and the *estimated* vessel strike mortality (18/yr) exceeds the PBR for this stock of blue whales and does not include vessel strikes outside of the U.S. EEZ~~. Monnahan *et al*. (2015) proposed that estimated ~~ship~~ 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 ~~ship~~ vessel strikes: 97% of K with 10 annual strikes and 91% of K with 35 annual strikes. The highest estimates of blue whale ~~ship~~ vessel strike mortality (35/yr; Monnahan *et al*. (2015) and 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 (≥ ~~1.44~~ 1.39) exceed 10% of the stock's PBR (~~1.23~~ 4.1), thus, commercial fishery take levels are not approaching zero mortality and serious injury rate.

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Revised 8/6/2021 ~~8/20/2020~~

# 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 20º to 30º 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). ~~Non~~A non-migratory population~~s~~ ~~exist~~ 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)~~ and the East China Sea (Fujino 1960)~~. 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~~. The whale~~ 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.

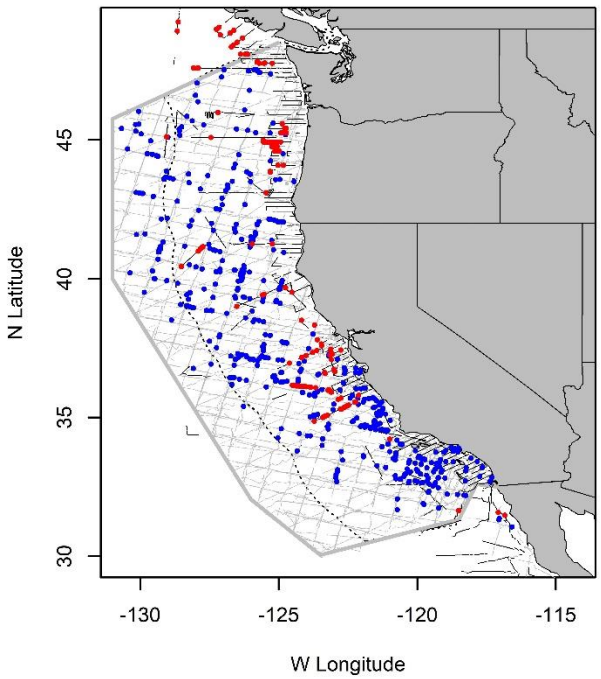


**Figure 1.** Fin whale sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018 (Barlow 2016, Henry *et al*. 2020). Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown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

~~The pre-whaling population of fin whales in the North Pacific was estimated to be 42,000-45,000 (Ohsumi and Wada 1974). In 1973, the North Pacific population was estimated to have been reduced to 13,620-18,680 (Ohsumi and Wada 1974), of which 8,520-10,970 were estimated to belong to the eastern Pacific stock.~~ 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). ~~The best estimate of fin whale abundance in California, Oregon, and Washington waters out to 300 nmi is 9,029 (CV=0.12) whales, based on a trend analysis of 1991-2014 line-transect data (Nadeem *et al*. 2016; Fig. 2).~~ This estimate is ~~based on similar methods applied to this population~~ 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). ~~However, the new abundance estimate is significantly higher than earlier estimates because the new analysis incorporates lower estimates of *g(0)*, the trackline detection probability (Barlow 2015). The trend model analysis incorporates information from the entire 1991-2014 time series for each annual estimate of abundance, and given the strong evidence of an increasing abundance trend over that time (Moore and Barlow~~

~~2011, Nadeem *et al.* 2016), the best estimate of abundance is represented by the estimate for the most recent year, or 2014. This is probably an underestimate because it excludes some fin whales that could not be identified in the field and were recorded as "unidentified rorqual" or "unidentified large whale".~~

**Minimum Population Estimate**

The minimum population estimate for fin whales is taken as the lower 20th percentile of the posterior distribution of ~~2014~~ 2018 abundance estimate, or ~~8,127~~ 7,970 whales (Becker *et al*. 2020b).

**Current Population Trend**

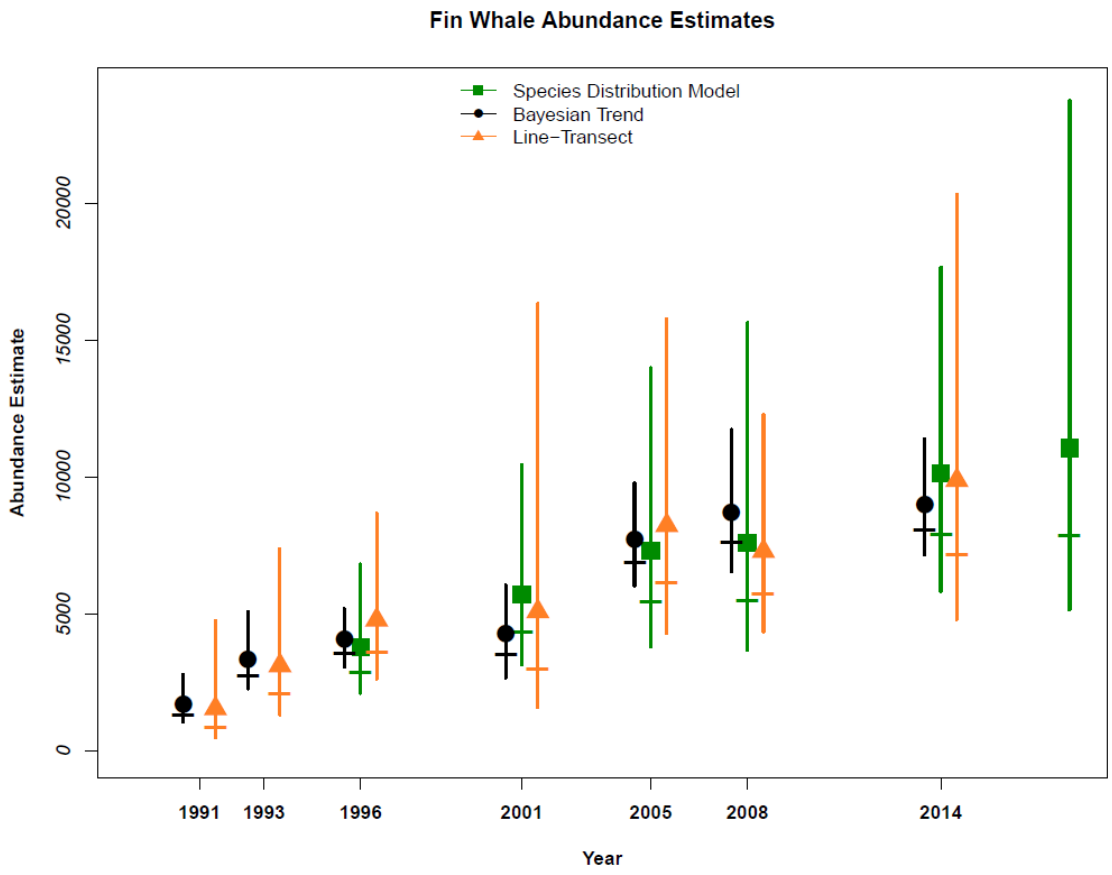


**Figure 2.** Fin whale abundance estimated from three methods (standard vessel-based line transect surveys (Barlow 2016), habitat-based species distribution models (Becker *et al*. 2020), and a Bayesian trend analysis (Nadeem *et al*. 2016). Vertical bars indicate approximate 95% log-normal confidence limits for line-transect estimates, 95% confidence limits reported from species distribution model estimates, and 95% prediction intervals from Nadeem *et al*. (2016). Line-transect surveys in 1991 and 1993 exclude Oregon and Washington waters. Horizontal hatch marks represent minimum population size estimates based on 20th 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 ~~2008~~ 2018 based on analysis of line transect surveys ~~conducted in the California Current between 1991 and 2014~~ (Moore and Barlow 2011, Nadeem *et al.* 2016, Becker *et al.* 2020a, Fig. 2). ~~Abundance in waters out to 300 nmi off the coast of California approximately doubled between 1991 and 1993, from approximately 1,744 (CV = 0.25) to 3,369 (CV= 0.21), suggesting probable immigration of animals into the area. Across the entire study area (waters off California, Oregon, and Washington), the~~ Nadeem *et al.* (2016) reported mean annual

abundance increased ~~was~~ 7.5% annually during 1991 to 2014. ~~, although abundance appeared stable between 2008 and 2014. In all, there has been a roughly 5-fold abundance increase between 1991 and 2014. Since 2005, the abundance increase has been driven by increases off northern California, Oregon and Washington, while numbers off Central and Southern California have been stable (Nadeem *et al.* 2016). Zerbini *et al.* (2006) found similar evidence of increasing fin whale abundance in Alaskan waters at a rate of 4.8% annually between 2001 and 2003.~~

## 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 (~~8,127~~ 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 ~~81~~ 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 ~~9,085~~ 9,158 observed fishing sets during 1990 - ~~2018~~ 2019 (Carretta ~~2020~~ 2021). The estimated bycatch of fin whales in this fishery for the most recent 5-year period is zero whales (Carretta ~~2020~~ 2021).

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 assigned 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.

Three fin whale serious injuries were documented in unidentified fishing gear during ~~2014-2018~~ 2015-2019, or 0.6 whales annually (Carretta *et al*. ~~2020~~ 2021). Additionally, there were ~~21~~ 16 *unidentfied whale* entanglements during this period, of which, ~~0.46~~ 0.28 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 ~~0.46~~ 0.28 x 0.75 = ~~0.35~~ 0.21 fin whale serious injuries occurred from the ~~21~~ 16 unidentified whale entanglement cases during ~~2014-2018~~ 2015-2019 (Table 1). This represents a negligible annual estimate of ~~0.07~~ 0.04 prorated fin whales derived from sightings of unidentified entangled whales. Total mean annual fishery-related serious injury and mortality is the sum of observed (0.6) and prorated (~~0.07~~ 0.04) mean annual deaths and serious injuries, or ~~0.67~~ 0.64 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) | Percent 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** | ~~2014-2018~~ 2015-2019 | observer | 21% | 0 | 0 (n/a) | 0 (n/a) |

| Fishery Name | Data Type | Year(s) | Percent Observer Coverage | Observed (or self-reported) | Estimated Mortality (and serious injury) | Mean Annual Mortality and Serious Injury (CV in parentheses) |
|---|---|---|---|---|---|---|
| **Unidentified fishery interactions involving *fin whales*** | ~~2014-2018~~ 2015-2019 | at-sea sightings | n/a | 3 | 0 (3) | ≥ 0.6 |
| **Unidentified fishery interactions involving *unidentified whales* prorated to fin whale** | ~~2014-2018~~ 2015-2019 | at-sea sightings | n/a | n/a | 0 (~~0.35~~ 0.21) | ~~0.07~~ ≥ 0.04 |
| **Minimum total annual takes** | | | | | | ≥ ~~0.67~~ 0.64 (n/a) |

**Vessel Strikes**

Vessel strikes were implicated in the deaths of ~~8~~ 7 fin whales from ~~2014-2018~~ 2015-2019 (Carretta *et al*. ~~2020~~ 2021). 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~~ 1.4 fin whales per year during ~~2014-2018~~ 2015-2019. 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 33%), 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). 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 ~~<~~ ~~0.5~~ 0.4% of the estimated population size (43 deaths / ~~9,029~~ 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 ~~1~~ 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 can be effective if applied over relatively small regions. ~~The authors of~~ 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 ~~12~~ 13-year period 2007-~~2018~~ 2019 (Carretta *et al.* 2013, 2018, 2020, 2021). During this ~~12~~ 13-year period, there were 20 observations of fin whale vessel strike deaths and 1 serious injury assumed to result in ~~the~~ death ~~of the whale~~, or ~~1.8~~ 1.6 fin whales annually. ~~The most conservative estimate of vessel strike deaths from Rockwood *et al.* (2017) is 43 whales annually.~~ The ratio of documented vessel strike deaths (~~1.8~~ 1.6/yr) to estimated annual deaths from the moderate avoidance model (43) implies a carcass recovery/documentation rate of ~~4.1~~ 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 ~~quite~~ 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 ~~that~~ primary vessel travel routes 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 ~~necessary~~ 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 in the North Pacific were given protected status by the IWC in 1976. 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. The total observed incidental mortality and serious injury (~~2.5~~ 2.2/yr), due to fisheries (~~0.67~~ 0.64/yr, including identified and prorated fin whales), and vessel strikes (~~1.8~~ 1.6/yr), is less than the calculated PBR (~~81~~ 80). However, observations alone underestimate true impacts due to incomplete detection of vessel strikes and fishery entanglements. 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 ~~0.5~~ 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 ~~1~~ 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.

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Revised 8/7/2021 ~~8/16/2016~~

# 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 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). 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 ~~considered~~ addressed in a separate stock assessment report.

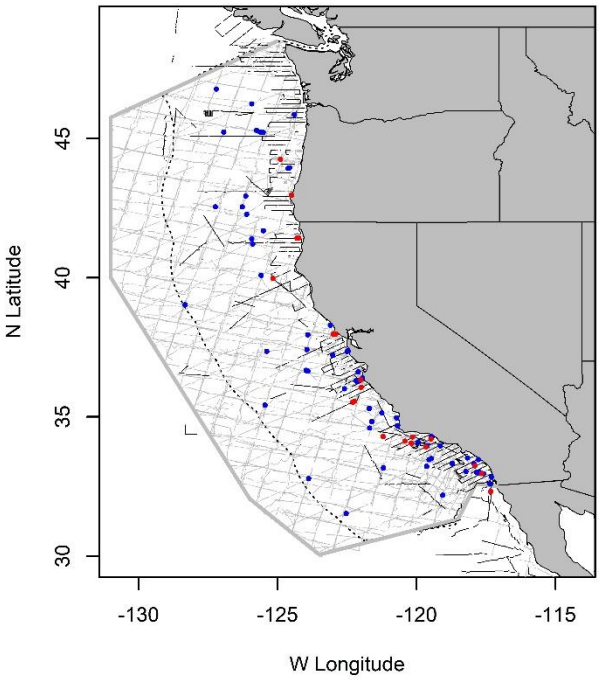


**Figure 1.** Minke whale sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018. Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

## POPULATION SIZE

~~No estimates have been made for the number of minke whales in the entire North Pacific. The most recent abundance estimate for this stock is based on the geometric mean of estimates obtained from ship line transect surveys in summer and autumn in 2008 and 2014, or 636 (CV=0.72) whales (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*. 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), ~~estimated from 2008 and 2014 summer/fall ship surveys in California, Oregon, and Washington waters (Barlow 2016)~~ or ~~approximately 369~~ 509 whales.

**Current Population Trend**

~~There are no data on trends in minke whale abundance in waters of California, Oregon and/or Washington.~~ 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).

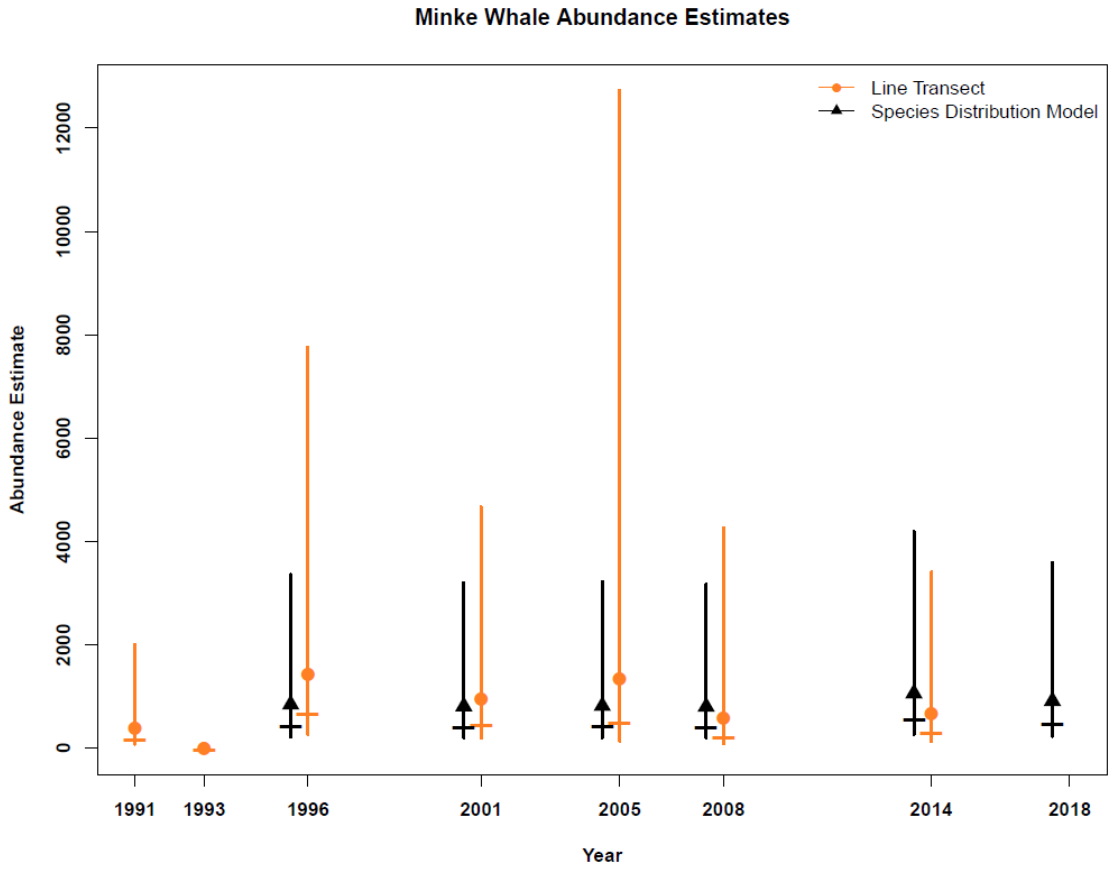


**Figure 2.** Minke whale abundance estimated from vessel-based line transect surveys (Barlow 2016) and habitat-based species distribution models based on 1991-2018 line-transect surveys (Becker *et al*. 2020). Vertical bars indicate approximate 95% log-normal confidence limits for line-transect estimates 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 (~~369~~ 509) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of ~~0.48~~ 0.40 (for a stock of unknown status with a mortality estimate CV > 0.80 ~~0.30 and < 0.60~~), resulting in a PBR of ~~3.5~~ 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 2020, Carretta *et al.* ~~2016a~~ 2021). Mean annual takes are based on ~~2010-2014~~ 2015-2019 data.

| Fishery Name | Years | Data Type | Percent Observer Coverage | Observed mortality (and serious injury) | Estimated Mortality (CV) | Mean Annual Takes (CV) |
|---|---|---|---|---|---|---|
| **CA/OR thresher shark/swordfish gillnet fishery** | ~~2010-2014~~ 2015-2019 | Observer | ~~22%~~ 21% | ~~1~~ 0 | ~~4.5 (0.58)~~ 1.2 (0.99) | ~~0.9 (0.58)~~ 0.24 (0.99) |
| **CA halibut and other species large mesh (>3.5") set gillnet fishery** | ~~2010-2014~~ 2017 | Observer | ~10% ~~9%~~ | 0 | 0 | 0 (n/a) ~~n/a~~ |
| **Unidentified fisheries** | ~~2010-2014~~ 2015-2019 | Sightings and strandings | n/a | 1 (0.75) | 1.75 (n/a) | ≥ 0.35 (n/a) |
| **Total annual takes** | | | | | | ≥ ~~1.3 (0.58)~~ 0.59 (0.99) |

### 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 1.2 (CV=0.99) whales for the 5-year period 2015-2019, or 0.24 whales annually (Carretta 2021, Table 1). This is a model-based estimate based on a total of ~~Four~~ four minke whales ~~were~~ observed entangled (2 dead, 2 released alive) between 1990-~~2014~~ 2019 ~~in the California swordfish drift gillnet fishery~~ from ~~over 8,600 monitored~~ 9,158 observed fishing sets (Carretta ~~*et al.* 2016a~~ 2021). ~~One animal 'released alive' in 1999 occurred in a set with a large hole in the net from which a skin sample was collected and positively identified as a minke whale with genetic sequencing. It is unknown whether or not gear remained on the whale. The estimate for the drift gillnet fishery in Table 1 (4.5 whales / 5 years = 0.9 annually) currently reflects total bycatch, regardless of animal condition (Carretta *et al.* 2016a).~~ Two additional unidentified fishery interactions with minke whales ~~fishery interactions~~ were recorded during ~~2010-2014~~ 2015-2019, totaling 1.75 serious injuries/deaths (Carretta *et al.* 2021). ~~: an entangled whale sighted at sea with rope and net material (=0.75 serious injury) and a live stranding of an animal that later died and appeared to have been previously entangled in unknown cable material (Carretta *et al.* 2016b).~~ The mean annual mortality and serious injury of minke whales from this stock during ~~2010-2014~~ 2015-2019 is ~~1.3~~ 0.59 animals (Table 1).

### Vessel Strikes

No vessel strikes of minke whales were reported during the most recent 5-years, 2015 to 2019, 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). ~~period of 2010-2014. Additional mortality from ship strikes probably goes unreported because the whales do not strand or, if they do, they do not always have obvious signs of trauma.~~

## STATUS OF STOCK

Minke whales are not listed as "endangered" under the Endangered Species Act and are not considered "depleted" under the MMPA. ~~The greatest uncertainty in their status is whether entanglement in commercial gillnets and ship strikes could have reduced this relatively small population. Because of this, the status of the west coast stock is considered "unknown".~~ The annual mortality and serious injury due to fisheries (~~1.3~~ 0.59/yr) and vessel strikes (0.0/yr) is less than the calculated PBR for this stock (~~3.5~~ 4.1), so

they are not considered a "strategic" stock under the MMPA. ~~Fishery~~ Estimated fishery mortality is not less than 10% of the PBR; therefore, total fishery mortality is not approaching zero mortality and serious injury rate. ~~There is no information on trends~~ 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 ~~items~~ (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 ~~5/5/2021~~8/5/2021

# FALSE KILLER WHALE (*Pseudorca crassidens*): Hawaiian Islands Stock Complex – Main Hawaiian Islands Insular, Northwestern Hawaiian Islands, and Hawaii 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, Hawaii, and the eastern tropical Pacific. False killer whales were encountered during three shipboard line-transect surveys of the U.S. Exclusive Economic Zone (EEZ) around the Hawaiian Islands in 2002, 2010, and 2017 (Figure 1; Barlow 2006, Bradford *et al.* 2014, Yano *et al.* 2018) and focused studies near the main and Northwestern Hawaiian Islands indicate that false killer whales occur in near shore waters throughout the Hawaiian archipelago (Baird *et al.* 2008, 2013). This species also occurs in U.S. EEZ waters around Palmyra and Johnston Atolls (e.g., Barlow *et al.* 2008) and American Samoa (Johnston *et al.* 2008, Oleson 2009).

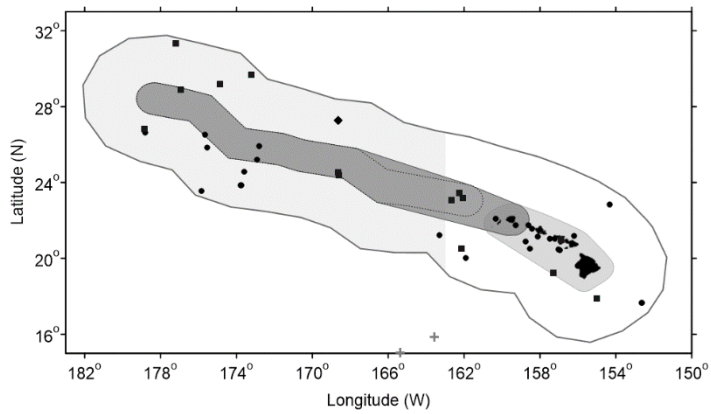


**Figure 1.** False killer whale sighting locations during the 2002 (diamond), 2010 (circle), and 2017 (square) shipboard surveys of U.S. EEZ waters surrounding the Hawaiian Islands (Barlow 2006, Bradford *et al.* 2014, Yano *et al.* 2018). Medium gray shaded area is the main Hawaiian Islands insular false killer whale stock area., including overlap zone between MHI insular and pelagic false killer whale stocks; Dark shaded gray area is the Northwestern Hawaiian Islands stock area, which overlaps the pelagic false killer whale stock area and part of the MHI insular false killer whale stock area. Outer line represents approximate boundary of survey area and U.S. EEZ. Dotted line represents the original boundary of the Papahanaumokuakea Marine National Monument and the light gray shaded area is the 2016 Expansion area. Detail of stock boundaries shown in Figure 2.

Genetic, photo-identification, and telemetry studies indicate there are three demographically-independent populations of false killer whales in Hawaiian waters. 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.* 2010; Martien 2014). Martien *et al.* (2014) analyzed mitochondrial DNA (mtDNA) control region sequences and genotypes from 16 nuclear DNA (nuDNA) microsatellite loci from 206 individuals from the MHI, NWHI, and offshore waters of the CNP and ENP and showed highly significant differentiation between populations confirming limited gene flow in both sexes. 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 three separate social clusters (Baird *et al.* 2012, Martien *et al.* 2019). 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 ~~scientific support~~ evidence for the separation of false killer whales in Hawaiian waters into three separate stocks is summarized by Oleson *et al.* (2010, 2012).

Fishery observers have collected tissue samples for genetic analysis from cetaceans incidentally caught in the Hawaii-based longline fishery since 2003. Between 2003 and 2010, eight false killer whale samples, four collected outside the Hawaiian EEZ and four collected within the EEZ, but more than 100 nautical miles (185km) from the main Hawaiian Islands were determined to have Pacific pelagic haplotypes (Chivers *et al.* 2010). At the broadest scale, significant differences in both mtDNA and nuDNA are evident between pelagic false killer whales in the ENP and CNP strata (Chivers *et al.* 2010). Sample distribution east and west of Hawaiian waters is insufficient to determine

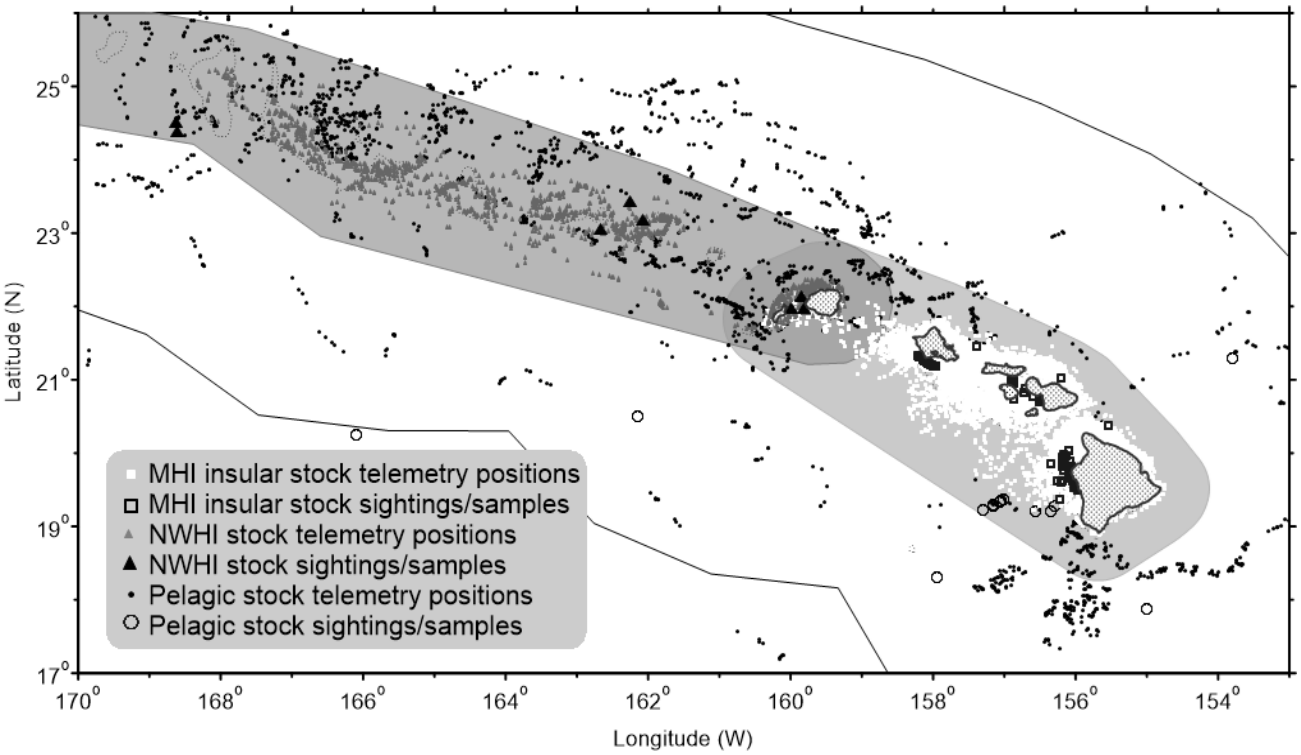


**Figure 2.** Sighting, biopsy sample, and telemetry record locations of false killer whale identified as being part of the MHI insular (squares ~~symbols~~), NWHI (triangles ~~symbols~~), or pelagic (circles ~~symbols~~) stocks. The MHI stock area is shown in light gray; the NWHI stock area is shown in dark gray; the pelagic stock area includes the entire EEZ (reproduced from Bradford *et al.* 2015, with pelagic stock boundary revision described in Bradford *et al*. 2020). The MHI insular, pelagic, and NWHI stocks overlap around Kauai and Niihau.

whether the sampled strata represent one or more stocks, and where pelagic stock boundaries may occur.

The stock range and boundaries of the three Hawaiian stocks of false killer whales are reviewed ~~in detail~~ in Bradford *et al.* (2015), and ~~further~~ revised for the pelagic stock in Bradford *et al* (2020) (Figure 2). The stocks have partially overlapping ranges. MHI insular false killer whales have been satellite tracked as far as 115 km from the main Hawaiian Islands, while pelagic stock animals have been tracked to within 5.6 km of the main Hawaiian Islands and throughout the NWHI. NWHI false killer whales have been seen up to 93 km from the NWHI and near-shore around Kauai and Oahu (Baird *et al.* 2012, Bradford *et al.* 2015). 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 Kauai and Niihau. The NWHI boundary is latitudinally expanded at the eastern end of the NWHI to encompass animal movements observed outside of the 93-km radius (~~see~~ Figure 2). The pelagic stock has no inner or outer boundary within the EEZ. The 2015 boundary revision placed an inner boundary at 11km from shore around each of the MHI, though this boundary was removed, given new telemetry data indicating use of waters within 5.6 km the MHI (Bradford *et al*. 2020) 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 pelagic stocks. The entirety of the NWHI stock range is an overlap zone between NWHI and pelagic false killer whales. All three stocks overlap out to the MHI insular stock boundary between Kauai and Nihoa and to the NWHI stock boundary between Kauai and Oahu (see Figure 2).

The pelagic stock includes animals found within the U.S EEZ around Hawaii and in adjacent international waters. New model-based abundance estimates for the central Pacific enable examination of the status of the broader population of false killer whales relative to human-caused impacts ~~resulting~~ from U.S. fisheries operating in international waters. The Palmyra Atoll stock of false killer whales is still considered to be a separate stock because comparisons amongst false killer whales sampled at Palmyra Atoll and those sampled from the MHI insular stock and the pelagic ENP reveal restricted gene flow, although the sample size remains too low for robust comparisons (Chivers

*et al.* 2010). The status of Hawaii pelagic stock is evaluated based on data from U.S. EEZ waters of the Hawaiian Islands (NMFS 2005), and abundance estimates for the broader central Pacific (including Palmyra Atoll) are provided for comparison to U.S. fisheries impacts on the high-seas.

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 main Hawaiian Islands, 2) the Northwestern Hawaiian Islands stock, which includes animals inhabiting waters within a 93 km radius around the NWHI and Kauai, with a latitudinal expansion of this area at the eastern end of the range, 3) the Hawaii pelagic stock, which includes false killer whales inhabiting waters of the U.S. EEZ around Hawaii and adjacent high seas waters, 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.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

Interactions with false killer whales, including depredation of pelagic fish catch, ~~have been~~ are identified in logbooks and NMFS observer records from Hawaii 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~~ing~~ lines of commercial and recreational fishermen (Shallenberger 1981). There are anecdotal reports of marine mammal interactions in the commercial Hawaii shortline fishery which sets gear at Cross Seamount and possibly around the main Hawaiian Islands. The commercial shortline fishery is licensed to sell catch through the State of Hawaii 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. 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 *et al*. 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 ~~have come~~originate from a variety of near-shore fisheries. No estimates of human-caused mortality or serious injury are currently available for near-shore hook and line or other fisheries because these fisheries are not monitored for protected species bycatch.

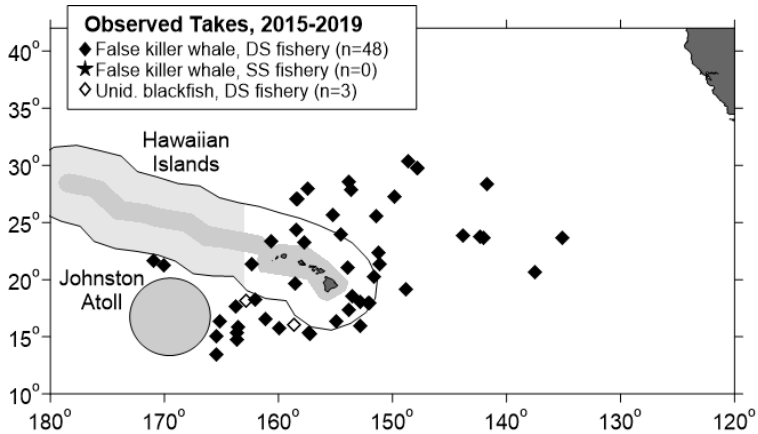


**Figure 3.** Locations of observed false killer whale takes (black symbols) and possible takes (blackfish) of this species (open symbols) in the Hawaii-based longline fisheries, 2015~~2014~~-2019~~2018~~. Some take locations overlap. Stock boundaries for false killer whales are not shown. One take occurred inside the NWHI-pelagic stock overlap area ~~and all observed takes during this period occurred in the pelagic stock area~~. Solid lines represent the U. S. EEZ. Gray shading notes areas closed to commercial fishing, with the PMNM Expansion area closed since August 2016~~. Takes within the PMNM closed area occurred prior to the closure in 2016.~~

Because of high rates of false killer whale mortality and serious injury in Hawaii-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 Hawaii pelagic, MHI insular and Palmyra stocks of false killer whales in Hawaii-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 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 LLEZ 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. As of August, 2016, the PMNM area was expanded to extend to the 200 nmi EEZ boundary west of 163º W. Between 20152014 and 20192018, noone false killer whales were observed hooked or entangled in the SSLL fishery (100% observer coverage) within the U.S. EEZ of the Hawaiian Islands, and 4844 false killer whales were observed taken in the DSLL fishery (18-21% observer coverage) within Hawaiian waters or adjacent high-seas waters (Bradford 2018a, 2018b, 2020, 2021Bradford and Forney 2017) (Figure 3). The severity of injuries resulting from interactions with longline gear is determined based on an evaluation of the observer's description of each interaction and following follows the most recently developed criteria for assessing serious injury in marine mammals (NMFS 2012). The one animal taken in the SSLL fishery was considered not seriously injured. In the DSLL fishery, 139 false killer whales were taken within the Hawaiian EEZ, including one within the overlap area of the pelagic and NWHI stocks. Stock identitiy is not known for any of the whales taken within the EEZ, though those outside of the stock overlap area are assumed to be pelagic stock animals. all within the range of the pelagic stock, withEleven of the 13 whales were 7 considered seriously injured, and two one non-seriously injured, and one could not be determined based on the information provided by the observer. Outside of the Hawaii EEZ five three were observed dead, 2124 were considered seriously injured, seven six were considered not seriously injured, and two could not be determined based on the information provided by the observer. Three One additional unidentified "blackfish" (unidentified cetaceans known to be either false killer whales or short-finned pilot whales) were also taken within in the DSLL fishery outside of the Hawaii EEZ, with one and was considered seriously injured, one not seriously injured, and one that could not be determined based on the information provided by the observer. The SEZ, a large triggered closure area south of the MHI implemented under the TRP, was closed following 2 serious injuries within the Hawaii 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 two serious injuries within the Hawaii EEZ. Following the closure there were three additional serious injuries within the Hawaii EEZ in 2019. The SEZ remained closed until August 2020.

The total estimated number of dead or seriously injured whales 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). 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).

The pelagic stock interacts with longline fisheries based on two genetic samples obtained by fishery observers (Chivers *et al.* 2010). 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). 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, Hawaii 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 20202019). 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 Hawaii EEZ are apportioned to each stock area by first allocating take to each areas 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.~~Two mortality and serious injury estimates are provided: a 5-yr average for the period prior to TRP-implementation (2008-2012), and a 5-yr average for the most recent 5 years following the TRP (2014-2018). The bycatch rate (per 1000 sets) and the proportion of non-serious injuries prior to and following TRP implementation are examined for all stocks as part of the FKW TRT monitoring strategy.~~

**Table 1.** Summary of available information on incidental mortality and serious injury (MSI) of false killer whales and unidentified blackfish (false killer whale or short-finned pilot whale) in commercial longline fisheries, by stock and EEZ area, as applicable (McCracken 2020~~2019~~). 5-yr mean annual takes are presented for 2015-2019~~2008-2012 (prior to the implementation of the TRP) and for 2014-2018~~. Information on observed takes (T) and combined mortality and serious injury is included. Unidentified blackfish are pro-rated as either false killer whales or short-finned pilot whales based on distance from shore (McCracken 2010). CVs are estimated based on the combined variances of annual false killer whale and blackfish 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 M&SI (CV)** | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | | | **FKW T/MSI<br>UB T/MSI** | | **Pelagic Stock** | | | |
| **Fishery Name** | **Year** | **Data Type** | **Percent Observer Coverage** | **Outside U.S EEZ** | **Within Hawaii EEZ** | **Outside U.S EEZ** | **Within Hawaii EEZ** | **MHI insular Stock** | **NWHI Stock** |
| Hawaii-based deep-set longline fishery | ~~2014~~ | Observer data | ~~21%~~ | ~~9/8<br>0~~ | ~~2/1‡<br>0~~ | ~~35.8 (0.5)~~ | ~~8.4 (0.7)~~ | ~~0.0 (0.8)~~ | ~~0.0 (1.5)~~ |
| | 2015 | | 21% | 5/4<br>1/1† | 0<br>0 | 22.3 (0.4) | 0 (-) | 0 (-) | 0 (-) |
| | 2016 | | 20% | 9/8†<br>0 | 1/1<br>0 | 27.9 (0.3) | 4.0 (0.8) | 0 (0.8) | 0 (2.1) |
| | 2017 | | 20% | 4/4†<br>0 | 2/1<br>0 | 28.5 (0.4) | 8.1 (0.6) | 0.1 (0.6) | 0 (2.0) |
| | 2018 | | 18% | 8/5<br>0 | 4/4<br>0 | 29.7 (0.4) | 11.9 (0.4) | 0.1 (0.5) | 0 (2.0) |
| | 2019 | | | 9/7<br>1/0 | 6/5<br>0 | 37.2 (0.3) | 25.0 (0.4) | 0.0 (0.5) | 0.6 (2.0) |
| ~~**Pre-TRP Mean Estimated Annual Take (CV) 2008-2012**~~ | | | | | | ~~**10.0 (0.4)**~~ | ~~**13.3 (0.2)**~~ | ~~**0.2 (0.4)**~~ | ~~**0.6 (0.8)**~~ |
| **Mean Estimated Annual Take (CV) 2015-2019~~2014-2018~~** | | | | | | **28.8 (0.2)** | **9.8 (0.3)**<br>~~**6.5 (0.3)**~~ | **0.0 (0.4)**<br>~~**0.03 (0.3)**~~ | **0.1 (1.8)**<br>~~**0.01 (1.1)**~~ |
| Hawaii-based shallow-set longline fishery | ~~2014~~ | Observer data | ~~100%~~ | ~~0<br>0~~ | ~~1/0<br>0~~ | ~~0.2~~ | ~~0~~ | ~~0~~ | ~~0~~ |
| | 2015 | | 100% | 0<br>0 | 0<br>0 | 0 | 0 | 0 | 0 |
| | 2016 | | 100% | 0<br>0 | 0<br>0 | 0 | 0 | 0 | 0 |
| | 2017 | | 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 |
| ~~Pre TRT Mean Annual Takes (100% coverage) 2008-2012~~ | | | | | | ~~0.3~~ | ~~0.3~~ | ~~0~~ | ~~0~~ |
| **Mean Annual Takes (100% coverage) 2015-2019~~2014-2018~~** | | | | | | 0<br>~~0.2~~ | **0** | **0** | **0** |
| ~~Pre TRP Minimum total annual takes within U.S. EEZ (2008-2012)~~ | | | | | | | ~~13.6 (0.2)~~ | ~~0.2 (0.4)~~ | ~~0.6 (0.8)~~ |
| | | | | | | | | | |
| **Minimum total annual takes within U.S EEZ (2015-2019~~2014-2018~~)** | | | | | | | 9.8 (0.3)<br>~~6.5 (0.3)~~ | 0.03 (0.4)<br>~~0.03 (0.3)~~ | 0.1 (1.8)<br>~~0.01 (1.1)~~ |

† Injury status could not be determined based on information collected by the observer. Injury status is prorated (see text).

~~Proration of false killer whale and unidentified blackfish takes within overlap zones introduces unquantified uncertainty into the bycatch estimates. Until methods of determining stock identity and/or species (e.g. photos, tissue samples) for animals observed taken within the overlap zone are available, proration approaches are needed ensure that potential impacts to all stocks are assessed in the overlap zones.~~

## 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 reflect the abundance of MHI insular false killer whales within the surveyed area in that year, and therefore should 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) times one half the default maximum net growth rate for cetaceans (½ of 4%) times 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 MHI insular stock 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). The Status Review report produced by the Biological Review Team (BRT) (Oleson *et al.* 2010, amended in Oleson *et al.* 2012) found that Hawaiian insular false killer whales are a Distinct Population Segment (DPS) of the global false killer whale taxon. Of the 29 identified threats to the population, the BRT considered 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 & Ito, 1993, Reeves *et al.* 2009), and hooking, entanglement, or intentional harm by fishermen to be the most substantial threats to the population. 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. 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. Prior to that time, a seasonal contraction to the LLEZ potentially exposed a significant portion of the offshore range of the stock to longline fishing. For the most recent 5-yr period, the estimate of mortality and serious injury (0.03) 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. Effects of other threats have yet to be assessed, e.g., nearshore hook and line fishing and environmental contamination. There is significant geographic overlap between various nearshore fisheries and evidence of interactions with hook-and-line gear (~~e.g.~~ Baird *et al.* 2015), such that these fisheries may pose a threat to the stock. Five MHI insular false killer whales stranded between 2010-2016, including four from cluster 3 (PIRO MMRN), a high rate for a single social cluster. ~~Recent research has indicated that~~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)~~ed proposed threshold levels for health effects in 84% of sampled MHI insular false killer whales (Foltz *et al.* 2014)~~.

**HAWAII PELAGIC STOCK**

**POPULATION SIZE**

Encounter data from shipboard line-transect surveys conducted throughout the central Pacific were used to estimate the abundance of false killer whales across the central Pacific, including within the Hawaii EEZ (Bradford *et al.* 2020; Table 2).

Table 2. Model-based line-transect abundance estimates for false killer whales derived from NMFS surveys in the central Pacific since 1997 (Bradford *et al.* 2020).

| | Hawaii EEZ | | | Central Pacific | | |
|---|---|---|---|---|---|---|
| Year | Model-based abundance | CV | 95% Confidence Limits | Model-based abundance | CV | 95% Confidence Limits |
| 2017 | 2,086 | 0.35 | 1,079-4,031 | 34,536 | 0.35 | 17,782-54,363 |
| 2010 | 2,144 | 0.32 | 1,159-3,965 | 25,212 | 0.33 | 13,449-47,262 |
| 2002 | 2,122 | 0.33 | 1,136-3,964 | 25,723 | 0.30 | 14,397-45,958 |

Bradford *et al.* (2020) also produced design-based abundance estimates for false killer whales within each survey year and these can be used as a point of comparison to ~~the~~ model-based estimates. 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 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 available estimate for each survey year. Current model-based estimates for the central Pacific include animals that are considered part of the Palmyra Atoll stock, as well as animals that may be part of an eastern Pacific stock on the eastern edge of the modeled area, and therefore are likely an overestimate of the full Hawaii pelagic stock abundance. Previous abundance estimates from the Hawaii EEZ and central Pacific using subsets of the full dataset and different line-transect parameters have been published previously. The estimate of 2,086 (CV=0.35) from the 2017 survey is considered the best available current estimate for false killer whales in the Hawaii EEZ (Bradford *et al*. 2020).

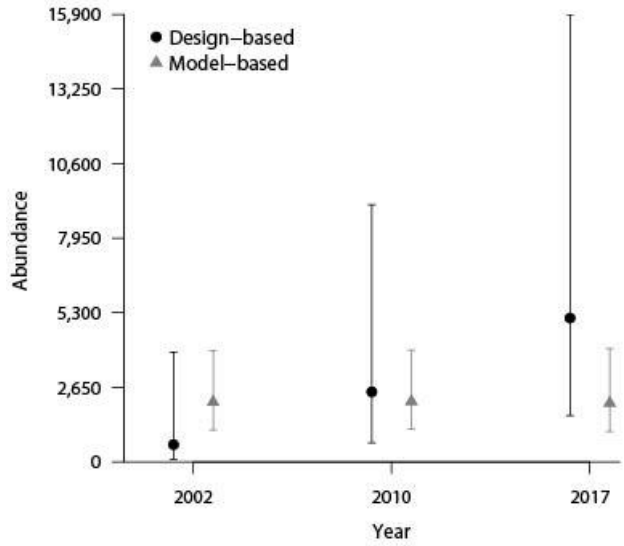


**Figure 4.** Comparison of design-based (circles) and model-based (triangles) (Bradford *et al*. 2020) estimates of abundance for false killer whales for each survey year (2002, 2010, 2017).

The reanalysis may still be subject to potential bias due to vessel attraction as described by Bradford *et al.* (2014)~~. There the authors~~ who reported that most (64%) false killer whale groups seen during the 2010 survey were seen 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 data 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 size is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2017 abundance estimate for the Hawaiian Islands EEZ (Bradford *et al.* 2020) or 1,567 false killer whales. For the entire central Pacific study area, the minimum population size for 2017 is estimated to be 25,940 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 Hawaii EEZ, however, these changes can be largely explained by random variability in encounter rate common for species with low density and patchy distribution. Examination of population trend for false killer whales requires additional data inside and outside of the Hawaii 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 for the Hawaii pelagic stock of false killer whales is calculated as the minimum population estimate for the U.S. EEZ of the Hawaiian Islands (1,567) 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 a Hawaiian Islands EEZ mortality and serious injury rate CV <= 0.30; Wade and Angliss 1997), resulting in a PBR of 16 false killer whales per year. For the entire central Pacific, based on the minimum population size of 25,940 false killer whales, and using the same recovery factor and maximum net growth rate as for the Hawaii pelagic stock, would yield a PBR of 259 false killer whales per year.

**STATUS OF STOCK**

The status of the Hawaii pelagic stock of false killer whales relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. The mean concentration of polychlorinated biphenyls (PCBs) in all Hawaii 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).~~Concentrations of polychlorinated biphenyls (PCBs) exceeded proposed threshold levels for health effects in 84% of sampled MHI insular false killer whales (Foltz *et al.* 2014), and elevated concentrations are also expected in pelagic false killer whales given the amplification of these contaminants through the food chain and likely similarity in false killer whale diet across the region.~~ 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 2005), the status of this transboundary stock of false killer whales is assessed based on the estimated abundance and mortality and serious injury within the U.S. EEZ of the Hawaiian Islands because estimates of human-caused mortality and serious injury from all U.S. and non-U.S. sources in high seas waters are not available. The estimated mortality and serious injury within the Hawaii EEZ in 2018 was the highest recorded since before the TRP was ~~put into place~~ implemented, with the estimated take in 2019 more than double that in 2018. ~~Although take estimates for 2019 are not reported here, 5 observed takes within the EEZ were reported in that year, resulting in closure of the SEZ in February 2019.~~ Take rates of false killer whales by the deep-set longline fishery outside of the EEZ continue to remain significantly higher since the TRP. Model-based estimates of abundance and PBR for the central Pacific should be considered when evaluating stock status across the fishery area. Total 5-year mortality and serious injury for 2015-2019~~2014-2018~~ (9.8~~6.5~~) is less than PBR (16), therefore this stock is not considered a "strategic stock" under the MMPA. ~~Additional monitoring of bycatch rates of this stock are required before assessing whether TRP measures have reduced rangewide fishery takes below PBR.~~ Total fishery mortality and serious injury for the Hawaii pelagic stock of false killer whales cannot be considered to be insignificant and approaching zero (i.e. less than 10% of PBR, or 1.6 animals per year).

**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 Northwestern Hawaiian Islands false killer whales (Bradford *et al*. 2020; Table 3).

Table 3. Line-transect abundance estimates for Northwestern Hawaiian Islands false killer whales derived from surveys of the entire Hawaii EEZ in 2002, 2010, and 2017 (Bradford *et al.* 2020).

| Year | Abundance | CV | 95% Confidence Limits |
|---|---|---|---|
| 2017 | 477 | 1.71 | 48-4,712 |
| 2010 | 878 | 1.15 | 145-5,329 |
| 2002 | N/A | | |

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 ~~following~~ 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*. (2020), uses 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)~~. Those authors~~, who reported that most (64%) false killer whale groups seen during the 2010 HICEAS survey were seen 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 Northwestern Hawaiian Islands stock (Bradford *et al.* 2020) or 178 false killer whales. This estimate has not been corrected for vessel attraction and may be positively-biased.

**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 in the waters surrounding the Northwestern Hawaiian Islands.

**POTENTIAL BIOLOGICAL REMOVAL**

The potential biological removal (PBR) level for the Northwestern Hawaiian Islands false killer whale stock is calculated as the minimum population estimate (178) 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 Hawaiian Islands EEZ mortality and serious injury rate CV > 0.8; Wade and Angliss 1997), resulting in a PBR of 1.4 false killer whales per year.

**STATUS OF STOCK**

The status of false killer whales in Northwestern Hawaiian Islands waters relative to OSP is unknown, and insufficient data exists to evaluate abundance trends. The mean concentration of polychlorinated biphenyls (PCBs) in all Hawaii 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).~~Concentrations of polychlorinated biphenyls (PCBs) exceeded proposed threshold levels for health effects in 84% of sampled MHI insular false killer whales (Foltz *et al.* 2014), and elevated concentrations are expected in NWHI false killer whales given amplification of these contaminants through the food chain and likely similarity in false killer whale diet across the region.~~ Biomass of some false killer whale prey species may have declined around the Northwestern Hawaiian Islands (Oleson *et al.* 2010, Boggs & Ito 1993, Reeves *et al.* 2009), though waters within the original PMNM~~Papahānaumokuākea Marine National Monument~~ 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 (0.1~~0.01~~) is less than the PBR (1.4 animals per year), and can be considered to be insignificant and approaching zero. 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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Appendix 2. Pacific reports revised in 2021 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 | Total Annual Mortality + Serious Injury | Annual Fishery Mortality + Serious Injury | Strategic Status | Recent Abundance Surveys | | | SAR Last Revised |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| California Sea Lion | United States | 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 | ~~179,000~~ | n/a | ~~81,368~~ | 0.12 | 1 | ~~4,882~~ | ~~8.8~~ | 4 | N | ~~2002~~ | ~~2005~~ | ~~2010~~ | ~~2014~~ |
| | | 187,386 | | 85,369 | | | 5,122 | 5.3 | 5.3 | | 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 | Hawaii | ~~1,437~~ | 0.03 | ~~1,374~~ | 0.07 | 0.1 | 4.8 | ~~≥4.8~~ | ≥2.0 | S | ~~2016~~ | ~~2017~~ | ~~2018~~ | ~~2020~~ |
| | | 1,437 | | 1,376 | | | | ≥4.6 | | | 2017 | 2018 | 2019 | 2021 |
| Harbor porpoise | Morro Bay | ~~4,255~~ | ~~0.56~~ | ~~2,737~~ | ~~0.097~~ | 0.5 | ~~66~~ | ~~≥0.4~~ | ~~≥0.2~~ | N | 2008 | 2011 | 2012 | ~~2019~~ |
| | | 4,191 | 0.56 | 2,698 | 0.096 | | 65 | 0 | 0 | | | | | 2021 |
| Harbor porpoise | Monterey Bay | ~~3,455~~ | ~~0.58~~ | ~~2,197~~ | ~~0.042~~ | 0.5 | ~~23~~ | ≥0.2 | ≥0.2 | N | 2011 | 2012 | 2013 | ~~2019~~ |
| | | 3,760 | 0.561 | 2,421 | 0.058 | | 35 | | | | | | | 2021 |
| Harbor porpoise | San Francisco - Russian River | ~~7,524~~ | ~~0.57~~ | ~~4,801~~ | ~~0.04~~ | 0.5 | ~~48~~ | ~~≥0.6~~ | ~~≥0.6~~ | N | 2014 | 2016 | 2017 | ~~2019~~ |
| | | 7,777 | 0.62 | 4,811 | 0.061 | | 73 | ≥0.4 | ≥0.4 | | | | | 2021 |
| Harbor porpoise | Northern CA/Southern OR | ~~24,195~~ | ~~0.4~~ | ~~17,447~~ | 0.04 | 1 | ~~349~~ | ~~≥0.2~~ | 0 | N | 2011 | 2014 | 2016 | ~~2019~~ |
| | | 24,685 | 0.41 | 17,713 | | | 354 | 0 | | | | | | 2021 |
| Harbor porpoise | Northern OR/Washington Coast | 21,487 | 0.44 | 15,123 | 0.04 | 0.5 | 151 | ≥3.0 | ≥3.0 | N | 2002 | 2010 | 2011 | 2013 |
| 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 | ~~25,750~~ | ~~0.45~~ | ~~17,954~~ | 0.04 | 0.48 | ~~172~~ | ~~0.3~~ | ~~0.3~~ | N | ~~2005~~ | ~~2008~~ | ~~2014~~ | ~~2016~~ |
| | | 16,498 | 0.61 | 10,286 | | | 99 | ≥0.66 | ≥0.66 | | 2008 | 2014 | 2018 | 2021 |
| Pacific white-sided dolphin | California/Oregon/Washington | ~~26,814~~ | ~~0.28~~ | ~~21,195~~ | 0.04 | ~~0.45~~ | ~~191~~ | ~~7.5~~ | ~~1.1~~ | N | ~~2005~~ | ~~2008~~ | ~~2014~~ | ~~2016~~ |
| | | 34,999 | 0.222 | 29,090 | | 0.48 | 279 | 7 | 4 | | 2008 | 2014 | 2018 | 2021 |
| 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 | ~~1,924~~ | ~~0.54~~ | ~~1,255~~ | 0.04 | ~~0.45~~ | ~~11~~ | ~~≥1.6~~ | ~~≥1.6~~ | N | ~~2005~~ | ~~2008~~ | ~~2014~~ | ~~2016~~ |
| | | 3,477 | 0.696 | 2,048 | | 0.48 | 20 | ≥0.8 | ≥0.8 | | 2008 | 2014 | 2018 | 2021 |
| Striped dolphin | California/Oregon/Washington | ~~29,211~~ | ~~0.20~~ | ~~24,782~~ | 0.04 | 0.48 | ~~238~~ | ~~≥0.8~~ | ~~≥0.8~~ | N | ~~2005~~ | ~~2008~~ | ~~2014~~ | ~~2016~~ |
| | | 29,988 | 0.30 | 23,448 | | | 225 | ≥4.0 | ≥4.0 | | 2008 | 2014 | 2018 | 2021 |
| Common dolphin, short-beaked | California/Oregon/Washington | ~~969,861~~ | ~~0.17~~ | ~~839,325~~ | 0.04 | 0.5 | ~~8,393~~ | ~~≥40~~ | ~~≥40~~ | N | ~~2005~~ | ~~2008~~ | ~~2014~~ | ~~2016~~ |
| | | 1,056,308 | 0.21 | 888,971 | | | 8,889 | ≥30.5 | ≥30.5 | | 2008 | 2014 | 2018 | 2021 |
| Common dolphin, long-beaked | California | ~~101,305~~ | ~~0.49~~ | ~~68,432~~ | 0.04 | 0.48 | ~~657~~ | ~~≥35.4~~ | ~~≥32.0~~ | N | ~~2005~~ | ~~2008~~ | ~~2014~~ | ~~2016~~ |
| | | 83,379 | 0.216 | 69,636 | | | 668 | ≥29.7 | ≥26.5 | | 2008 | 2014 | 2018 | 2021 |
| Northern right whale dolphin | California/Oregon/Washington | ~~26,556~~ | ~~0.44~~ | ~~18,608~~ | 0.04 | 0.48 | ~~179~~ | ~~3.8~~ | ~~3.8~~ | N | ~~2005~~ | ~~2008~~ | ~~2014~~ | ~~2016~~ |
| | | 29,285 | 0.72 | 17,024 | | | 163 | ≥6.6 | ≥6.6 | | 2008 | 2014 | 2018 | 2021 |

Appendix 2. Pacific reports revised in 2021 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 | Total Annual Mortality + Serious Injury | Annual Fishery Mortality + Serious Injury | Strategic Status | Recent Abundance Surveys | | | SAR Last Revised |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Killer whale | Eastern N Pacific Offshore | 300 | 0.1 | 276 | 0.04 | 0.5 | 2.8 | 0 | 0 | N | 2010 | 2011 | 2012 | 2018 |
| Killer whale | Eastern N Pacific Southern Resident | ~~73~~ | n/a | ~~73~~ | 0.035 | 0.1 | 0.13 | ≥0.4 | 0 | S | ~~2017~~ | ~~2018~~ | ~~2019~~ | ~~2020~~ |
| | | 72 | | 72 | | | | | | | 2018 | 2019 | 2020 | 2021 |
| 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 | ~~2,697~~ | ~~0.6~~ | ~~1,633~~ | 0.04 | 0.5 | ~~16.0~~ | ~~0~~ | ~~0~~ | N | ~~2005~~ | ~~2008~~ | ~~2014~~ | ~~2017~~ |
| | | 1,363 | 0.53 | 894 | | | 8.9 | ≥0.2 | 0 | | 2008 | 2014 | 2018 | 2021 |
| Mesoplodont beaked whales | California/Oregon/Washington | 3,044 | 0.54 | 1,967 | 0.04 | 0.5 | 20.0 | 0.1 | 0.1 | N | 2005 | 2008 | 2014 | 2017 |
| Cuvier's beaked whale | California/Oregon/Washington | 3,274 | 0.67 | 2,059 | 0.04 | 0.5 | 21 | <0.1 | <0.1 | N | 2005 | 2008 | 2014 | 2017 |
| Pygmy Sperm whale | California/Oregon/Washington | 4,111 | 1.12 | 1,924 | 0.04 | 0.5 | 19.2 | 0 | 0 | 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 |
| Sperm whale | California/Oregon/Washington | 1,997 | 0.57 | 1,270 | 0.04 | 0.1 | 2.5 | 0.6 | 0.64 | S | 2005 | 2008 | 2014 | 2019 |
| Gray whale | Eastern N Pacific | 26,960 | 0.05 | 25,849 | 0.062 | 1.0 | 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 | California/Oregon/Washington | ~~2,900~~ | ~~0.05~~ | ~~2,784~~ | 0.08 | 0.3 | ~~16.7~~ | ~~≥ 42.1~~ | ~~≥ 19.4~~ | S | ~~2011~~ | ~~2013~~ | ~~2014~~ | ~~2019~~ |
| | | 4,973 | 0.05 | 4,776 | | | 28.7 | ≥ 48.6 | ≥ 25.2 | | 2016 | 2017 | 2018 | 2021 |
| Blue whale | Eastern N Pacific | ~~1,496~~ | ~~0.44~~ | ~~1,050~~ | 0.04 | ~~0.1~~ | ~~1.2~~ | ~~≥ 19.4~~ | ~~≥ 1.44~~ | S | ~~2008~~ | ~~2011~~ | ~~2014~~ | ~~2019~~ |
| | | 1,898 | 0.085 | 1,767 | | 0.2 | 4.1 | ≥ 19.4 | ≥ 1.39 | | 2016 | 2017 | 2018 | 2021 |
| Fin whale | California/Oregon/Washington | ~~9,029~~ | ~~0.12~~ | ~~8,127~~ | 0.04 | 0.5 | ~~81~~ | ~~≥ 43.7~~ | ~~≥ 0.67~~ | S | ~~2005~~ | ~~2008~~ | ~~2014~~ | ~~2020~~ |
| | | 11,065 | 0.405 | 7,970 | | | 80 | | ≥ 2.2 | | 2008 | 2014 | 2018 | 2021 |
| Sei whale | Eastern N Pacific | 519 | 0.4 | 374 | 0.04 | 0.1 | 0.75 | ≥ 0.2 | ≥ 0.2 | S | 2005 | 2008 | 2014 | 2018 |
| Minke whale | California/Oregon/Washington | ~~636~~ | ~~0.72~~ | ~~369~~ | 0.04 | ~~0.48~~ | ~~3.5~~ | ~~≥ 1.3~~ | ~~≥ 1.3~~ | N | ~~2005~~ | ~~2008~~ | ~~2014~~ | ~~2016~~ |
| | | 915 | 0.792 | 509 | | 0.4 | 4.1 | ≥ 0.59 | ≥ 0.59 | | 2008 | 2014 | 2018 | 2021 |
| 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 | Hawaii | 76,375 | 0.41 | 54,804 | 0.04 | 0.5 | 548 | 0 | 0 | N | 2002 | 2010 | 2017 | 2020 |
| 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 | Hawaii | 7,385 | 0.22 | 6,150 | 0.04 | 0.5 | 61 | 0 | 0 | N | 2002 | 2010 | 2017 | 2020 |
| Common Bottlenose dolphin | Hawaii Pelagic | unk | unk | unk | 0.04 | 0.5 | undet | 0 | 0 | N | 2002 | 2010 | 2017 | 2017 |
| Common Bottlenose dolphin | Kaua'i and Ni'ihau | unk | unk | 97 | 0.04 | 0.5 | 1.0 | unk | unk | N | 2003 | 2012 | 2015 | 2017 |
| Common Bottlenose dolphin | O'ahu | unk | unk | n/a | 0.04 | 0.5 | undet | unk | unk | N | 2002 | 2003 | 2006 | 2017 |
| Common Bottlenose dolphin | 4 Islands Region | unk | unk | n/a | 0.04 | 0.5 | undet | unk | unk | N | 2002 | 2003 | 2006 | 2017 |
| Common Bottlenose dolphin | Hawaiian Island | unk | unk | 91 | 0.04 | 0.5 | 0.9 | unk | unk | N | 2002 | 2003 | 2006 | 2017 |
| Pantropical Spotted dolphin | Hawaii Pelagic | 39,798 | 0.51 | 26,548 | 0.04 | 0.5 | 265 | 0 | 0 | N | 2002 | 2010 | 2017 | 2020 |
| Pantropical Spotted dolphin | O'ahu | unk | unk | unk | 0.04 | 0.5 | undet | unk | unk | N | | | n/a | 2017 |
| Pantropical Spotted dolphin | 4 Islands Region | unk | unk | unk | 0.04 | 0.5 | undet | unk | unk | N | | | n/a | 2017 |
| Pantropical Spotted dolphin | Hawaii Island | unk | unk | unk | 0.04 | 0.5 | undet | ≥ 0.2 | ≥ 0.2 | N | | | n/a | 2017 |
| Spinner dolphin | Hawaii 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 |

Appendix 2. Pacific reports revised in 2021 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 | Total Annual Mortality + Serious Injury | Annual Fishery Mortality + Serious Injury | Strategic Status | Recent Abundance Surveys | | | SAR Last Revised |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Spinner dolphin | American Samoa | unk | unk | unk | 0.04 | 0.5 | undet | unk | unk | unk | | | n/a | 2010 |
| Striped dolphin | Hawaii Pelagic | 35,179 | 0.23 | 29,058 | 0.04 | 0.5 | 291 | 0 | 0 | N | 2002 | 2010 | 2017 | 2020 |
| Fraser's dolphin | Hawaii | 40,960 | 0.70 | 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 | 2002 | 2010 | 2017 | 2020 |
| Pygmy killer whale | Hawaii | 10,328 | 0.75 | 5,885 | 0.04 | 0.5 | 59 | 0 | 0 | N | 2002 | 2010 | 2017 | 2020 |
| False killer whale | NW Hawaiian Islands | 477 | 1.71 | 178 | 0.04 | 0.4 | 1.4 | ~~0.01~~ | ~~0.01~~ | N | 2002 | 2010 | 2017 | ~~2020~~ |
| | | | | | | | | 0.1 | 0.1 | | | | | 2021 |
| False killer whale | Hawaii Pelagic | 2,086 | 0.35 | 1,567 | 0.04 | 0.5 | 16 | ~~6.5~~ | ~~6.5~~ | N | 2002 | 2010 | 2017 | ~~2020~~ |
| | | | | | | | | 9.8 | 9.8 | | | | | 2021 |
| 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 | Main Hawaiian Islands Insular | 167 | 0.14 | 149 | 0.04 | 0.1 | 0.30 | ~~0.0~~ | ~~0.03~~ | S | 2013 | 2014 | 2015 | ~~2020~~ |
| | | | | | | | | 0.03 | 0.03 | | | | | 2021 |
| False killer whale | American Samoa | unk | unk | unk | 0.04 | 0.5 | undet | unk | unk | unk | n/a | n/a | n/a | 2010 |
| Killer whale | Hawaii | 161 | 1.06 | 78 | 0.04 | 0.5 | 0.8 | 0 | 0 | N | 2002 | 2010 | 2017 | 2020 |
| Pilot whale, short-finned | Hawaii | 12,607 | 0.18 | 10,847 | 0.04 | 0.4 | 87 | 0.9 | 0.9 | N | 2002 | 2010 | 2017 | 2020 |
| Blainville's beaked whale | Hawaii Pelagic | 1,132 | 0.99 | 564 | 0.04 | 0.5 | 5.6 | 0 | 0 | N | 2002 | 2010 | 2017 | 2020 |
| Longman's Beaked Whale | Hawaii | 2,550 | 0.67 | 1,527 | 0.04 | 0.5 | 15 | 0 | 0 | N | 2002 | 2010 | 2017 | 2020 |
| Cuvier's beaked whale | Hawaii Pelagic | 4,431 | 0.41 | 3,180 | 0.04 | 0.5 | 32 | 0 | 0 | N | 2002 | 2010 | 2017 | 2020 |
| Pygmy sperm whale | Hawaii | 42,083 | 0.64 | 25,695 | 0.04 | 0.5 | 257 | 0 | 0 | N | 2002 | 2010 | 2017 | 2020 |
| Dwarf sperm whale | Hawaii | unk | unk | unk | 0.04 | 0.5 | undet | 0 | 0 | N | 2002 | 2010 | 2017 | 2020 |
| Sperm whale | Hawaii | 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 | Hawaii | 203 | 0.99 | 101 | 0.04 | 0.1 | 0.2 | 0 | 0 | S | 2002 | 2010 | 2017 | 2020 |
| Bryde's whale | Hawaii | 602 | 0.22 | 501 | 0.04 | 0.5 | 5.0 | 0 | 0 | N | 2002 | 2010 | 2017 | 2020 |
| Sei whale | Hawaii | 391 | 0.90 | 204 | 0.04 | 0.1 | 0.4 | 0.2 | 0.2 | S | | 2002 | 2010 | 2017 |
| Minke whale | Hawaii | 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 | 0 | 0 | S | 2006 | 2007 | 2008 | 2009 |
| Sea Otter | Southern (California) | 3,272 | n/a | 3,272 | 0.06 | 0.1 | 10 | ≥5.2 | ≥0.4 | S | 2014 | 2015 | 2016 | 2017 |
| Sea Otter | Northern (Washington) | n/a | n/a | 1,806 | 0.2 | 0.1 | 18 | ≥1.0 | ≥0.4 | N | 2014 | 2015 | 2016 | 2018 |