



U.S. PACIFIC DRAFT MARINE MAMMAL STOCK ASSESSMENTS: 2018

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Stock assessment reports and appendices revised in 2018. Previous editions of stock assessments for other stocks and years can be found at the [NOAA Marine Mammal Stock Assessment](#) page.

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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 2018 Pacific marine mammal stock assessments include revised reports for 16 Pacific marine mammal stocks under NMFS jurisdiction, including 7 “strategic” stocks: Hawaiian monk seal, Eastern North Pacific blue whale, Western North Pacific gray whale, California/Oregon/Washington humpback whale, California/Oregon/Washington fin whale, Eastern North Pacific sei whale and Southern Resident killer whale. New abundance estimates are available for 8 stocks: California sea lions, Hawaiian monk seals, Eastern North Pacific Offshore killer whales, Southern Resident killer whales, Eastern North Pacific gray whales, Western North Pacific gray whales, California/Oregon/Washington humpback whales, and Hawaii Island spinner dolphins. A new population assessment for California sea lions estimates that the population size was approximately 258,000 animals in 2014 (Laake et al. 2018). The stock is estimated to be approximately 40% above its maximum net productivity level (MNPL = 183,481 animals), and it is therefore considered within the range of its optimum sustainable population (OSP) (Laake et al. 2018). The California sea lion population’s carrying capacity was estimated at approximately 275,000 animals in 2014 (Laake et al. 2018).

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. In particular, new information on serious injury and mortality resulting from estimated vessel strikes is included for the following stocks of large whales: California/Oregon/Washington humpback whale, California/Oregon/Washington fin whale, and the Eastern North Pacific blue whales, based on an analysis by Rockwood *et al.* 2017. Estimated levels of vessel strike mortality exceed PBR for both blue and humpback whale stocks, although estimated vessel strike levels represent a small fraction of the overall estimated population sizes

(0.5% to <2%). Estimated vessel strikes are also compared with recent detected levels of vessel strikes, which indicate that detection rates for vessel strike events are quite low, generally less than 15%.

For large whale stocks, previous cases of unidentified whale entanglements have been assigned to species based on an assignment model generated from historic known-species entanglements in the region (Carretta 2018). This has eliminated one negative bias in assessments that occurs when unidentified whale entanglements are not assigned to any species/stock.

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 Wade and Angliss (1997). The authors solicit any new information or comments which would improve future stock assessment reports.

Draft versions of the 2018 stock assessment reports were reviewed by the Pacific Scientific Review Group at the March 2018 meeting.

These Stock Assessment Reports summarize information from a wide range of original data sources and an extensive bibliography of all sources is given 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:

- [Carretta, J.V. 2018. A machine-learning approach to assign species to 'unidentified' entangled whales. *Endangered Species Research* Vol. 36: 89–98.](#)
- [Rockwood C.R., J. Calambokidis, and J. Jahncke. 2017. High mortality of blue, humpback and fin whales from modeling of vessel collisions on the U.S. West Coast suggests population impacts and insufficient protection. *PLoS ONE* 12\(8\): e0183052.](#)
- [Laake, J.L., M.S. Lowry, R.L. DeLong, S.R. Melin, and J.V. Carretta. 2018. Population growth and status of California sea lions. *The Journal of Wildlife Management*, DOI: 10.1002/jwmg.21405.](#)
- Wade, P.R. and R.P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS workshop April 3-5, 1996, Seattle, Washington. NOAA Technical Memorandum NMFS-OPR-12. Available from Office of Protected Resources, National Marine Fisheries Service, Silver Spring, MD. 93p.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

There are no international agreements between the U.S., Mexico, and Canada for joint management of California sea lions, and the number of sea lions at the Coronado Islands is not regularly monitored. Consequently, this stock assessment report considers only the U.S. Stock, i.e. sea lions at rookeries within the U.S. Pup production at the Coronado Islands is minimal (between 12 and 82 pups annually; Lowry and Maravilla-Chavez 2005) and does not represent a significant contribution to the overall size of the Pacific Temperate population.

POPULATION SIZE

The entire population cannot be counted because all age and sex classes are not ashore at the same time. In lieu of counting all sea lions, pups are counted during the breeding season (because this is the only age class that is ashore in its entirety), and the number of births is estimated from the pup count. Population size is then estimated from the number of births and the proportion of pups in the population. Surveys are conducted in July after all pups have been born. To estimate the number of pups born, the pup count for rookeries in southern California in 2008 (59,774) was adjusted for an estimated 15% pre-census mortality (Boveng 1988; Lowry *et al.* 1992), giving an estimated 68,740 live births in the population. The fraction of newborn pups in the population (23.2%) was estimated from a life table derived for the northern fur seal (*Callorhinus ursinus*) (Boveng 1988, Lowry *et al.* 1992) which was modified to account for the growth rate of this California sea lion population (5.4% yr⁻¹, see below). Multiplying the number of pups born by the inverse of this fraction (4.317) results in a population estimate of 296,750. More recent pup counts made in 2011 totaled 61,943 animals, the highest recorded to date (Figure 2). Estimates of total population size based on these counts are currently being developed, along with new estimates of the fraction of newborn pups in the population. [The size of the California sea lion population was estimated from a 1975-2014 time series of pup counts \(Lowry *et al.* 2017\), combined with mark-recapture estimates of survival rates \(DeLong *et al.* 2017, Laake *et al.* 2018\). Population size in 2014 was estimated at 257,606 animals, which corresponded with a pup count of 47,691 animals along the U.S. west coast \(Lowry *et al.* 2017, Laake *et al.* 2018\).](#)

Minimum Population Estimate

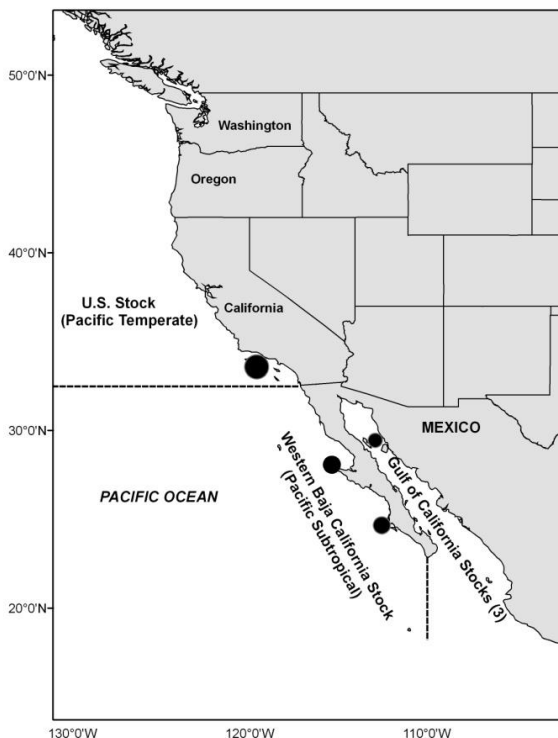


Figure 1. Geographic range of California sea lions showing stock boundaries and locations of major rookeries. The U.S. stock also ranges north into Canadian waters.

The minimum population size was determined from counts of all age and sex classes that were ashore at all the major rookeries and haulout sites in southern and central California during the 2007 breeding season. The minimum population size of the U.S. stock is 153,337 (NMFS unpubl. data). It includes all California sea lions counted during the July 2007 census at the Channel Islands in southern California and at haulout sites located between Point Conception and Point Reyes, California. An additional unknown number of California sea lions are at sea or hauled out at locations that were not surveyed. [The minimum population size for 2014 is taken as the lower 95% confidence interval of the 2014 population size estimate, or 233,515 animals \(Laake et al. \(2018\).](#)



Figure 2. U.S. pup count index for California sea lions (1975–2011). Trends in pup counts from 1975 through 2011 are shown for four rookeries in southern California and for haulouts in central and northern California.

[No estimate of the lower 20th percentile of the estimated population size available, which is typically used for \$N_{min}\$ in stock assessments and a coefficient of variation \(CV\) is unavailable from the estimated population size for use in calculating \$N_{min}\$. The lower 95% confidence limit is a more conservative estimate of minimum population size in this case and is superior to previous approaches that simply used 2x the annual pup count, which results in negatively-biased \$N_{min}\$ values because not all age classes are represented.](#)

Current Population Trend

Trends in [pup counts](#) [population size](#) from 1975 through 2011–2014 are shown in Figure 2–for four rookeries in southern California and for haulouts in central and northern California. [The time series of population](#)

[size estimates are derived from 3 primary data sources: 1\) annual pup counts \(Lowry et al. 2017\); 2\) estimates of annual survivorship from mark-recapture data \(DeLong et al. 2017\); and 3\) estimates of human-caused serious injuries, mortalities, and bycatch removals \(Carretta and Enriquez 2012a, 2012b, Carretta et al. 2016, Carretta et al. 2018a, 2018b\). These 3 data sources were combined to reconstruct the population size estimates shown in Figure 2 \(Laake et al. 2018\). The number of pups at rookeries that were not counted were estimated using multiple regression analyses derived from counts of two neighboring rookeries using data from 1975–2000 \(Lowry and Maravilla 2005\): \(1\) 1980 at Santa Barbara Is.; \(2\) 1978–1980 at San Clemente Is.; and \(3\) 1978 and 1979 at San Nicolas Is. The mean was used when more than one count was available for a given rookery. A regression of the natural logarithm of the pup counts by year indicates that pup counts increased at an annual rate of 5.4% between 1975 and 2008, when pup counts for El Niño years \(1983, 1984,](#)

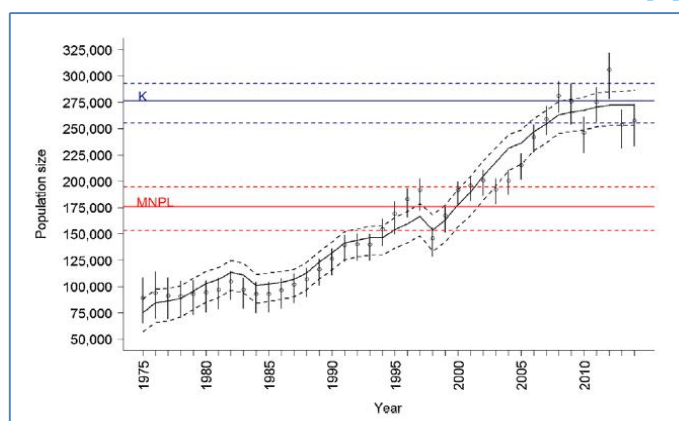


Figure 2. Fitted logistic growth curve (solid line) and 95% bootstrap intervals (dashed line) for reconstructed California sea lion annual population sizes in the United States, 1975–2014. Vertical lines indicate 95% confidence intervals for reconstructed annual population sizes. We also present estimated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

1992, 1993, 1998, and 2003) were removed from the 1975–2005 time series. Using 1975–2008 non-El Niño year data, the coefficient of variation for this average annual growth rate (CV=0.04) was computed via bootstrap sampling of the count data. The 1975–2008 time series of pup counts shows the effect of four El Niño events on the sea lion population (Figure 2). Pup production decreased by 35% in 1983, 27% in 1992, 64% in 1998, and 20% in 2003. After the 1992–93, 1997–98 and 2003 El Niños, pup production rebounded to pre-El Niño levels within two years. In contrast, however, the 1983–1984 El Niño affected adult female survivorship (DeLong *et al.* 1991), which prevented an immediate rebound in pup production because there were fewer adult females available in the population to produce pups (it took five years for pup production to return to the 1982 level). Other characteristics of El Niños are higher pup and juvenile mortality rates (DeLong *et al.* 1991, Lowry and Maravilla-Chavez, 2005) which affect future recruitment into the adult population for the affected cohorts. The 2002 and 2003 decline can be attributed to (1) reduced number of reproductive adult females being incorporated into the population as a result of the 1992–93 and 1997–98 El Niños, (2) domoic acid poisoning (Scholin *et al.* 2000, Lefebvre *et al.* 2000), (3) lower survivorship of pups due to hookworm infestations (Lyons *et al.* 2001), and (4) the 2003 El Niño. Large numbers of emaciated sea lion pups stranded in early 2013 in California and pup weight indices at the San Miguel Island rookery were significantly lower in 2012 compared with previous years (Wells *et al.* 2013). As a result of the large numbers of sea lion strandings in 2013, NOAA declared an unusual mortality event (UME). Although the exact causes of this UME are unknown, two hypotheses meriting further study include nutritional stress of pups resulting from a lack of forage fish available to lactating mothers and unknown disease agents during that time period. [Age- and sex-specific survival rates of California sea lions were estimated by DeLong *et al.* \(2017\), who report that female survivorship exceeds that of males. Annual pup survival was 0.600 and 0.574 for females and males, respectively. Maximum annual survival rates corresponded to animals 5 years of age \(0.952 and 0.931 for females and males, respectively\). Survival of pups and yearlings declined with increasing sea surface temperatures \(SST\). For every 1°C increase in SST, the authors found a corresponding 50% decline in survival rates. Such declines in survival are related to warm oceanographic conditions \(e.g. El Niño\) that limit prey availability to pregnant and lactating females \(DeLong *et al.* 2017\).](#)

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (453,337 [233,515](#)) times one half the default maximum net growth rate for pinnipeds (½ of 12%) times a recovery factor of 1.0 (for a stock of unknown status that is growing [within OSP](#), Laake *et al.* 2018, Wade and Angliss 1997); resulting in a PBR of 9,200 [or 14,011](#) sea lions per year.

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Serious Injury Guidelines

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

Historical Depletion

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

Fisheries Information

California sea lions are killed in a variety of trawl, purse seine, and gillnet fisheries along the U.S. west coast (Barlow *et al.* 1994, Carretta and Barlow 2011, Carretta *et al.* 2013^{2018a, 2018b}, Julian and Beeson 1998, Jannot *et al.* 2011, Stewart and Yochem 1987). Those for which recent observations or estimates of bycatch mortality exist are summarized in Table 1. In addition to bycatch estimates from fishery observer programs, information on fishery-related sea lion deaths and serious injuries comes largely from stranding data (Carretta *et al.* 2013^{2018b}). Stranding data represent a minimum number of animals killed or injured, as many entanglements are likely unreported or undetected.

California sea lions are also incidentally killed and injured by hooks from recreational and commercial fisheries. Sea lion deaths due to hook-and-line fisheries are often the result of complications resulting from ingestion of hooks, perforation of body cavities leading to infections, or the inability of the animal to feed. Many of the animals die post-stranding during rehabilitation or are euthanized as a result of their injuries. Between 2008 and 2012 and 2016, there were 124-146 California sea lion deaths / serious injuries attributed to hook and line fisheries, or an annual average of 25-29 animals (Carretta *et al.* 2014b^{2018b}). One sea lion death was reported in a tribal salmon gillnet in 2009 along the U.S. west coast.

Table 1. Summary of available information on the mortality and serious injury of California sea lions in commercial fisheries that might take this species (Carretta *et al.* 2014a, 2009, 2010, 2012a, 2012b, 2014a, 2018a, 2018b; Heery *et al.* 2010; Jannot *et al.* 2011; Appendix 1). Mean annual takes are based on 2008-2012 2012-2016 data unless noted otherwise. Bycatch estimates for 2 additional years, 2010 and 2011, have been included for the CA halibut and white seabass set gillnet fishery because this fishery has not been observed in recent years.

Fishery Name	Year(s)	Data Type	Percent Observer Coverage	Observed Mortality	Estimated Mortality (CV in parentheses)	Mean Annual Takes (CV in parentheses)
CA/OR thresher shark/swordfish large mesh drift gillnet fishery	2008	observer	13.5%	7	51 (0.52)	42 (0.50) <u>12.5 (0.24)</u>
	2009		13.3%	5	37 (0.83)	
	2010		11.9%	0	0 (n/a)	
	2011		19.5%	18	92 (0.79)	
	2012		18.6%	6	32 (0.60)	
	<u>2012</u>		<u>19%</u>	<u>6</u>	<u>16.1 (0.58)</u>	
	<u>2013</u>		<u>37%</u>	<u>3</u>	<u>11.6 (0.35)</u>	
	<u>2014</u>		<u>24%</u>	<u>3</u>	<u>10.9 (0.59)</u>	
	<u>2015</u>		<u>20%</u>	<u>0</u>	<u>6.2 (0.92)</u>	
	<u>2016</u>		<u>18%</u>	<u>0</u>	<u>17 (0.67)</u>	
	<u>2012-2016</u>		<u>23%</u>	<u>12</u>	<u>62.3 (0.24)</u>	
CA halibut and white seabass set gillnet fishery	2008	observer	0%	n/a	n/a	200 (0.21) <u>150 (0.28)</u>
	2009		0%	n/a	n/a	
	2010		12.5%	25	199 (0.30)	
	2011		8.0%	6	74 (0.39)	
	2012		5.5%	18	326 (0.33)	
	<u>2013</u>		<u>n/a</u>	<u>0</u>	<u>0 (n/a)</u>	
	<u>2014</u>		<u>0%</u>	<u>n/a</u>	<u>n/a</u>	
	<u>2015</u>		<u>0%</u>	<u>n/a</u>	<u>n/a</u>	
<u>2016</u>	<u>0%</u>	<u>n/a</u>	<u>n/a</u>			
CA small-mesh drift gillnet fishery for white seabass, yellowtail, barracuda, and tuna	2010	observer	0.7%	0	0 (n/a)	0 (n/a)
	2011		3.3%	0	0 (n/a)	
	2012		4.6%	0	0 (n/a)	
CA anchovy, mackerel, sardine, and tuna purse-seine fishery	2004-2008	observer	~5%	2	n/a	≥2 (n/a)

Fishery Name	Year(s)	Data Type	Percent Observer Coverage	Observed Mortality	Estimated Mortality (CV in parentheses)	Mean Annual Takes (CV in parentheses)
WA, OR, CA domestic groundfish trawl fishery (includes at-sea hake and other limited-entry groundfish sectors)	2005 2006 2007 2008 2009	observer	98% to 100% of tows in at-sea hake fishery	14 21 8 7 4	21 (n/a) 95 (n/a) 31 (n/a) 13 (n/a) 10 (n/a)	34 (n/a)
	2012-2016		Generally less than 30% of landings observed in other groundfish sectors	95	n/a	<u>≥ 19 (n/a)</u>
Unknown entangling net fishery	2008-2012 2012-2016	stranding	n/a	55	n/a	<u>≥ 53 (n/a)</u> <u>≥ 11 (n/a)</u>
Unknown trawl fishery and bait barge-net entanglement <u>Unidentified fishery interactions</u>	2008-2012	stranding	n/a	2 11	n/a	<u>≥ 2 (n/a)</u> <u>≥ 2.2</u>
Minimum total annual takes						<u>≥ 331 (0.14)</u> <u>≥ 197 (0.23)</u>

Other Mortality

[California sea lions strand with evidence of human-caused mortality and serious injury from a variety of non-commercial fishery sources, including shootings, hook and line fisheries, power plant entrapment, marine debris entanglement, oil exposure, vessel strikes, and dog attacks \(Carretta *et al.* 2018b\). Between 2012 and 2016, there were 485 mortality and serious injury cases documented from these sources \(Carretta *et al.* 2018b\), or an annual average of 97 sea lions killed and/or seriously injured. The three largest sources of such mortalities and serious injuries over this period were shootings \(n=155\), hook and line fisheries \(n=146\), entanglements in marine debris \(n=65\), and oil exposure \(n=58\) which accounted for 87% of all cases. These values represents a minimum accounting of impacts, because an unknown number of dead or injured animals are never detected. Live strandings and dead beach-cast California sea lions are regularly observed with gunshot wounds in California \(Lowry and Folk 1987, Goldstein *et al.* 1999, Carretta *et al.* 2013\). A summary of stranding records for 2008 to 2012 from California, Oregon, and Washington shows the following non-fishery related human-caused mortality and serious injuries: boat collisions \(13\), car collisions \(3\), entrapment in power plants \(59\), shootings \(151\), marine debris entanglement or ingestion \(37\), research-related \(18\), and other sources, including dog attacks, harassment, seal bombs, stabbings, and, blunt force trauma \(10\). Stranding records are a gross under-estimate of mortality and serious injury because many animals and carcasses are never recovered. The minimum number of non-fishery related deaths and serious injuries during 2008-2012 was 291 sea lions, or an annual average of 58 animals.](#)

Under authorization of MMPA Section 120, individually identifiable California sea lions have been killed or relocated since 2008 in response to their predation on endangered salmon and steelhead stocks in the Columbia River. Relocated animals were transferred to aquaria and/or zoos. Between 2009-2012 and 2013-2016, a total of 47 122 California sea lions were removed from this stock (40-115 lethal removals and 7-7 relocations to aquaria and/or zoos). The average annual mortality due to direct removals for the 2009-2013-2012-2016 period is 9.4 24.4 animals per year (relocations to aquaria/zoos are treated the same as mortality because animals are effectively removed from the stock).

Between ~~2008 and 2012~~ and 2016, ~~18-1~~ California sea lions were was incidentally killed, ~~2-3~~ seriously injured, and ~~8~~ there were 2 non-serious injuries along the U.S. west coast during scientific trawl and longline operations conducted by NMFS (Carretta *et al.*, 2018b-2014b). The average annual research-related mortality and serious injury of California sea lions from ~~2008 to 2012~~ and 2016 is ~~4.0~~ 0.8 animals.

[Mortality and serious injury may occasionally occur incidental to marine mammal research activities authorized under NMFS protected species permits issued to a variety of government, academic, and other research organizations. Between 2012-2016, there were nine reported mortalities during research activities, resulting in a mean annual mortality and serious injury rate of 1.8 sea lions.](#)

Habitat Concerns

Sea lion mortality linked to the algal-produced neurotoxin domoic acid has been documented sporadically since 1998 (Scholin *et al.* 2000, Brodie *et al.* 2006, Ramsdell and Zabka 2008). Future mortality may be expected to occur, due to the repeated occurrence of such harmful algal blooms.

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

Expanding pinniped populations in general have resulted in increased human-caused serious injury and mortality, due to shootings, entrapment 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, 2018b).

[Increasing sea-surface temperatures in the California Current negatively impact prey availability and reduce survival rates of California sea lions \(DeLong *et al.* 2017, Laake *et al.* 2018, Lowry *et al.* 1991, Melin *et al.* 2008, 2010\). For every 1°C increase in sea surface temperature, annual survivorship of sea lion pups and yearlings was found to decline by 50% \(DeLong *et al.* 2017\). Increasing ocean temperatures may continue to limit the population size of the California sea lion stock within the California Current \(Cavole *et al.* 2016, DeLong *et al.* 2017, Laake *et al.* 2018\).](#)

STATUS OF STOCK

California sea lions in the U.S. are not listed as "endangered" or "threatened" under the Endangered Species Act or as "depleted" under the MMPA. ~~The optimum sustainable population (OSP) status of this population has not been formally determined. The stock is estimated to be approximately 40% above its maximum net productivity level (MNPL = 183,481 animals), and it is therefore considered within the range of its optimum sustainable population (OSP) (Laake *et al.* 2018). The carrying capacity of the population was estimated at 275,298 animals in 2014 (Laake *et al.* 2018).~~ The average annual commercial fishery mortality is ~~331~~ 197 animals per year (Table 1). Other sources of human-caused mortality (shootings, direct removals, recreational hook, [research-related](#) and line fisheries, ~~tribal takes~~, entrapment in power plant intakes, etc.) average ~~97~~ 58 animals per year. Total human-caused mortality, [serious injury](#) of this stock is at least ~~389~~ 321 animals per year, [which does not include undetected and unreported mortalities and serious injuries](#). California sea lions are not considered "strategic" under the MMPA because total human-caused mortality is less than the PBR (9,200 [14,011](#)). The total fishery mortality and serious injury rate (~~389~~ 197 animals/year) for this stock is less than 10% of the calculated PBR and, therefore, is considered to be insignificant and approaching a zero mortality and serious injury rate.

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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 ~~1,324~~**1,415** (95% confidence interval ~~1,263~~**1,348**-~~1,430~~**1,525**; CV = 0.03), (Table 1, Baker *et al.* 2016, Johanos 2017a**2018a**, b, c). In 2015, NWHI field camp durations were longest in duration since 2011, with the exception of Midway Atoll. This allowed for more thorough demographic studies. 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.* ~~in review~~**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.* (~~in review~~**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 2015~~**2016**, total enumeration was achieved only at ~~Kure Atoll~~**Lisianski Island**, and a capture-recapture estimates ~~was~~**were** obtained for ~~Midway Atoll and Pearl and Hermes Reef~~**French Frigate Shoals**. At ~~French Frigate Shoals~~, Laysan Island, Lisianski Island, Pearl and Hermes Reef, and Midway ~~Kure~~ Atoll abundance estimates were obtained using discovery curve analysis. 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 2016**, ~~There were no a single counts count was~~ conducted at Necker Island in 2014 or 2015, so ~~two beach counts conducted in 2013 were used to estimate abundance (no change in abundance since 2013 assumed)~~**and four** ~~Three counts were conducted at Nihoa Island in 2015.~~

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 small number of surveys of Ni'ihau and nearby Lehua Islands have been 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 in ~~2015~~**2016** 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.*, ~~in review~~**2017**) found that MHI monk seals (N=23) spent a greater proportion of time ashore than Harting *et al.* (~~in review~~**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.*, in review 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 2014-2016. The estimation method is indicated for each site. Methods used include DC: discovery curve analysis, EnumEN: total enumeration; CR: capture-recapture; CC: counts corrected for the proportion of seals at sea; Min: minimum tally. Median values are presented with (in parentheses) 95% confidence intervals of 30,000 random draws from abundance distributions where estimates of error are available. 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).

Location	Total			Minimum N_{min}			Method
	Non-pups	Pups	Total	Non-pups	Pups	Total	
French Frigate Shoals	148 ₁₆₄	45 ₃₅	193 ₁₉₉	143 ₁₆₂	45 ₃₅	188 ₁₉₇	CRDC
Laysan	209 ₂₀₈	35 ₃₁	244 ₂₃₉	208	35 ₃₁	243 ₂₃₉	DC
Lisianski	142 ₁₃₃	18 ₂₃	151 ₁₆₅	133 ₁₄₂	18 ₂₃	151 ₁₆₅	DCEN
Pearl and Hermes Reef	118 ₁₃₅	27 ₂₉	145 ₁₆₄	118 ₁₃₃	27 ₂₉	145 ₁₆₂	DCCR
Midway	53 ₆₁	11 ₁₂	64 ₇₃	48 ₅₉	11 ₁₂	59 ₇₁	DCCR
Kure	78	12 ₂₀	90 ₉₈	78	12 ₂₀	90 ₉₈	EnumDC
Necker	59 ₆₃	5 ₇	64 ₇₀	49 ₅₃	5 ₇	54 ₆₀	CC
Nihoa	108 ₁₀₄	9 ₇	117 ₁₁₁	91 ₈₈	9 ₇	100 ₉₅	CC
MHI (without Ni'ihau/Lehua)	130 ₁₂₄	15 ₁₆	145 ₁₄₀	130 ₁₂₄	15 ₁₆	145 ₁₄₀	Min
Ni'ihau/Lehua	81 ₁₂₂	21	102 ₁₄₃	103 ₆₅	21	86 ₁₂₄	CC
Total Range-wide	1214 1214 ₁₁₂₆	201 201 ₁₉₈	1415 1415 ₁₃₂₄	1183 1183 ₁₀₆₃	201 201 ₁₉₈	1384 1384 ₁₂₆₁	

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 (1,261-1,384) are presented in Table 1.

Current Population Trend

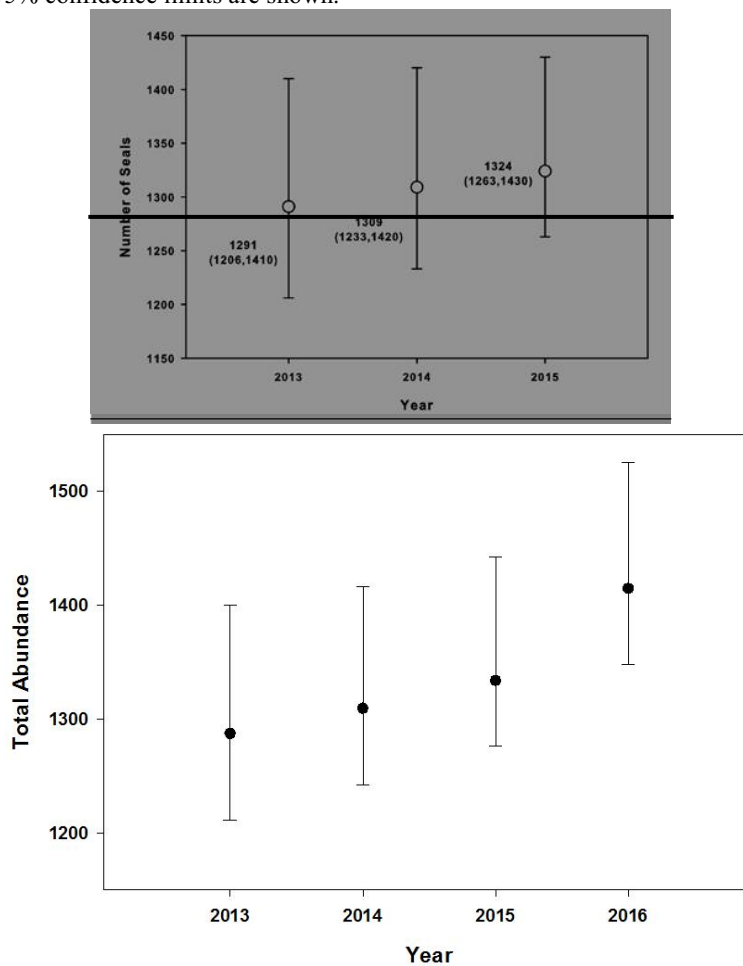
Range-wide abundance estimates are available only from 2013 to 2015-2016 (Figure 1). While these estimates remain somewhat negatively-biased for reasons explained in Baker et al. (2016), they provided a much more comprehensive assessment of status and trends than has been previously available. [A Monte Carlo approximation of the annual multiplicative rate of realized population growth during 2013-2016 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 1.04 \(1.01, 1.08\). Thus, the best estimate is that the population grew at an average rate of about 4% per year from 2013 to 2016. Only 1% of the distribution was below 1, indicating that there is a 99% chance that the monk seal population increased during 2013-2016.](#)

The abundance estimates from 2013 to 2015 are encouraging—the point estimate for 2014 is higher than for 2013, and 2015 is even higher. The confidence intervals for all years largely overlap one another. Thus, it is not currently possible to unequivocally conclude whether the current trend is declining, stable, or increasing. A reliable conclusion regarding population trend will only be apparent after more annual range-wide abundance estimates have accrued.

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 2017a). 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 (new figure for 2017). Range-wide abundance of Hawaiian monk seals, 2013-2016 (from Baker *et al.* 2016). Medians and 95% confidence limits are shown.



POTENTIAL BIOLOGICAL REMOVAL

Past reports have concluded that Hawaiian monk seal stock dynamics did not conform to the underlying model for calculating PBR such that PBR for the Hawaiian monk seal has been undetermined. That conclusion was based on the fact that the stock was declining despite being well below OSP (Optimum Sustainable Population level). The trend since 2013 (Figure 1) does not indicate the stock has continued to decline, recently been increasing so that PBR may be determined. Using current minimum population size (1,261, 1,384), R_{max} (0.07) and a recovery factor (F_r) for ESA endangered stocks (0.1), yields a Potential Biological Removal (PBR) of 4.4-4.8.

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 ~~since 2011~~ during 2012-2016 (Johanos 2015 ~~and~~ 2018d). There were no confirmed cases in 2016; an adult female died due to trauma which may or may not have been anthropogenic.

Year	Age/sex	Island	Cause of Death	Comments
2011	Adult male	Molokai	Skull fracture, blunt force trauma	Intent unconfirmed
2011	Juvenile female	Molokai	Skull fracture, blunt force trauma	Intent unconfirmed
2012	Juvenile male	Kauai	Gunshot wound	
2012	Subadult male	Kauai	Skull fracture	Intent unconfirmed
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
	Juvenile male	Kauai	Probable boat strike	
2015	Adult male	Laysan	Research handling	Accidental, specific cause undetermined

It is extremely unlikely that all carcasses of intentionally killed monk seals are discovered and reported. 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 ~~2015~~ 2016, 11 seal hookings were documented (Henderson 2017a). Among these were two serious injuries and one mortality. The latter was a weaned female pup who ingested a hook. The hook was surgically removed but the pup succumbed to post-surgical complications. ~~The remaining 8 hookings, and all~~ were classified as non-serious injuries, although 2-6 of these would have been deemed serious had they not been mitigated (Henderson 2017a, Mercer 2018). Several incidents involved hooks used to catch ulua (jacks, *Caranx* spp.). Monk seals also interact with n ~~Nearshore gillnets, and several confirmed deaths have resulted.~~ s became a more common source of mortality in the 2000s, with three seals confirmed dead in these gillnets (2006, 2007, and 2010), and one additional seal in 2010 may have also died in similar circumstances but the carcass was not recovered. No gillnet-related mortality or injuries have ~~were been documented~~ confirmed since ~~during 2012 to 2016, though two 2016 mortalities are considered suspect net mortalities (Mercer 2018)~~ 2010. Most reported hookings and gillnet entanglements have occurred since 2000 (Henderson 2017a). The MHI monk seal population appears to have been increasing in abundance during this period (Baker *et al.* 2011). 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 Type	% Obs. coverage	Observed/Reported Mortality/Serious Injury	Estimated Mortality/Serious Injury	Non-serious (Mitigated serious)	Mean Takes (CV)
Pelagic Longline	2011	observer	20.3% & 100%	0	0	0	0 (0)
	2012	observer	20.4% & 100%	0	0	0	
	2013	observer	20.4% & 100%	0	0	0	
	2014	observer	20.8% & 100%	0	0	0	
	2015	observer	20.6% & 100%	0	0	0	
	2016	observer	20.1% & 100%	0	0	0	
MHI Bottomfish	2011	Incidental observations of seals	none	0	n/a	0	n/a
	2012			0		0	
	2013			0		0	
	2014			0		0	
	2015			0		0	
	2016			0		0	
Nearshore	2011	Incidental observations of seals	none	0	n/a	9 (3)	≥1.6
	2012			4		12 (5)	
	2013			0		15 (6)	
	2014			1		13 (9)	
	2015			3		8 (2)	
	2016			0		11 (6)	
Minimum total annual takes							≥ 1.6

Fishery Mortality Rate

Total fishery mortality and serious injury is not considered to be insignificant and approaching a rate of zero. Monk seals are being hooked and entangled in the MHI at a rate that has not been reliably assessed but is certainly greater than zero. 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 9 documented mortalities](#). A total of 361 cases of seals entangled in fishing gear or other debris have been observed from 1982 to 2015 (Henderson 2001; Henderson 2017b, [2018b](#)). ~~Nine~~

~~documented deaths resulted from entanglement in marine debris (Henderson 1990, 2001; Henderson 2017b).~~—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.

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).

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). Infectious disease effects on monk seal demographic trends are low relative to other stressors. However, land-to-sea transfer of *Toxoplasma gondii*, a protozoal parasite shed in the feces of cats, is of growing concern. 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). Eight monk seal mortalities (and 1 suspect mortality) have been directly attributed to toxoplasmosis from 2001 to 2015 (2016). The number of mortalities from this pathogen are likely underrepresented, given that more seals disappear each year than are found dead and examined. 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. Furthermore, the consequences of a disease outbreak introduced from livestock, feral animals, pets or other carrier wildlife may 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 Littman 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 (<http://www.nmfs.noaa.gov/pr/permits/eis/hawaiianmonksealeis.htm>). A major habitat issue involves loss of terrestrial habitat at French Frigate Shoals, where some pupping and resting islets have shrunk or virtually disappeared (Antonelis *et al.* 2006). Projected increases in global average sea level may further significantly reduce terrestrial habitat for monk seals in the NWHI (Baker *et al.* 2006, Reynolds *et al.* 2012).

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.

Remains of the seawall at Tern Island, French Frigate Shoals, is an entrapment hazard for seals.—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. Strategies to mitigate these threats are currently under consideration and there are discussions of USFWS supporting the extension of monk seal field camps to allow for entrapment mitigation beyond the regular spring/summer field season.](#)

Vessel groundings pose a continuing threat to monk seals and their habitat, through potential physical damage to reefs, oil spills, and release of debris into habitats.

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. 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 (2011–2015–2016) was at least 3.4–3.0 animals, including fishery-related mortality in nearshore gillnets and hook-and-line gear ($\geq 1.6/\text{yr}$, Table 3), intentional killings and other human-caused mortalities (≥ 1.8 –1.4/yr, Table 2).

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

Killer whales have been observed in all oceans and seas (Leatherwood and Dahlheim 1978). Although reported from 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](#), [Green et al. 1992](#); [Barlow 1995, 1997](#); [Forney et al. 1995](#); [Barlow and Forney 2007](#)). Seasonal and year-round occurrence have been noted for killer whales throughout Alaska (Braham and Dahlheim 1982) and in the intracoastal waterways of British Columbia and Washington, where pods ~~are~~ [three ecotypes](#) have been labeled as [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](#)). Through examination of photographs of recognizable individuals and pods, movements of whales between geographical areas have been documented. For example, whales identified in Prince William Sound have been observed near Kodiak Island (Heise et al. 1991) and whales identified in Southeast Alaska have been observed in Prince William Sound, British Columbia, and Puget Sound (Leatherwood et al. 1990, Dahlheim et al. 1997). Movements of killer whales between the waters of Southeast Alaska and central California have also been documented (Goley and Straley 1994). Offshore killer whales have more recently also been identified off the coasts of [are known from southern](#) California waters north to the Aleutian Islands and they are considered to represent a single network of socially-connected individuals (Dahlheim et al. 2008, [Ford et al. 2014](#)). Photographic matches of individuals between Dutch Harbor, Alaska and southern California waters near Dana Point have been documented ([Dahlheim et al. 2008](#)), Oregon, and rarely, in Southeast Alaska (Ford et al. 1994, [Black et al. 1997](#), [Dahlheim et al. 1997](#)). They [Offshore killer whales](#) apparently do not mix with the transient and resident killer whale stocks found in these regions (Ford et al. 1994, [Black et al. 1997](#)). Studies indicate the 'offshore' type, although distinct from the other types ('resident' and 'transient'), appears to be more closely related genetically, morphologically, behaviorally, and vocally to the 'resident' type killer whales ([Black et al. 1997](#), [Hoelzel et al. 1998](#), [Morin et al. 2010](#)). [Genetic studies of killer whales globally suggest that residents and transient ecotypes warrant subspecies recognition \(Morin et al. 2010\) and are currently listed as unnamed subspecies of *Orcinus orca* \(Committee on Taxonomy 2018\).](#) At this time the offshore killer whale ecotype is included under *Orcinus orca* (Committee on Taxonomy 2018). ~~;~~ Ford, pers. comm.; L. Barrett-Lennard, pers. comm.). Based on data regarding association patterns, acoustics,

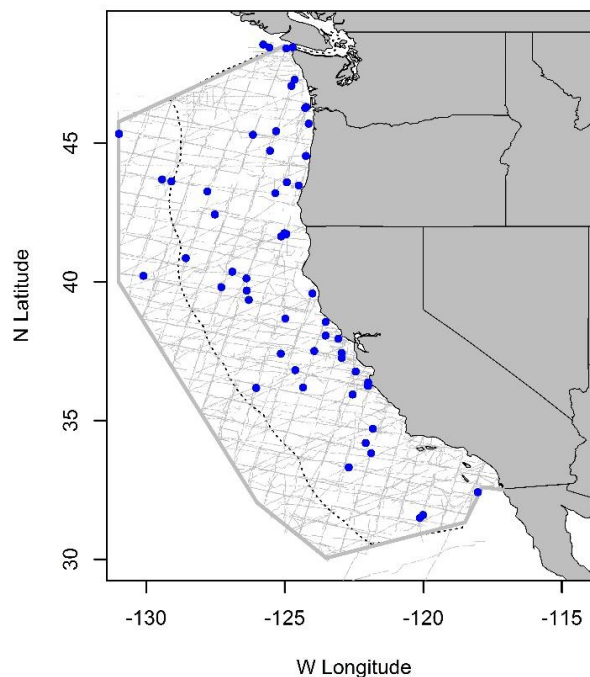


Figure 1. Sightings of killer whales (all ecotypes/stocks) encountered during Southwest Fisheries Science Center line-transect vessel surveys in the California Current ecosystem, 1991-2014.

For example, whales identified in Prince William Sound have been observed near Kodiak Island (Heise et al. 1991) and whales identified in Southeast Alaska have been observed in Prince William Sound, British Columbia, and Puget Sound (Leatherwood et al. 1990, Dahlheim et al. 1997). Movements of killer whales between the waters of Southeast Alaska and central California have also been documented (Goley and Straley 1994). Offshore killer whales have more recently also been identified off the coasts of [are known from southern](#) California waters north to the Aleutian Islands and they are considered to represent a single network of socially-connected individuals (Dahlheim et al. 2008, [Ford et al. 2014](#)). Photographic matches of individuals between Dutch Harbor, Alaska and southern California waters near Dana Point have been documented ([Dahlheim et al. 2008](#)), Oregon, and rarely, in Southeast Alaska (Ford et al. 1994, [Black et al. 1997](#), [Dahlheim et al. 1997](#)). They [Offshore killer whales](#) apparently do not mix with the transient and resident killer whale stocks found in these regions (Ford et al. 1994, [Black et al. 1997](#)). Studies indicate the 'offshore' type, although distinct from the other types ('resident' and 'transient'), appears to be more closely related genetically, morphologically, behaviorally, and vocally to the 'resident' type killer whales ([Black et al. 1997](#), [Hoelzel et al. 1998](#), [Morin et al. 2010](#)). [Genetic studies of killer whales globally suggest that residents and transient ecotypes warrant subspecies recognition \(Morin et al. 2010\) and are currently listed as unnamed subspecies of *Orcinus orca* \(Committee on Taxonomy 2018\).](#) At this time the offshore killer whale ecotype is included under *Orcinus orca* (Committee on Taxonomy 2018). ~~;~~ Ford, pers. comm.; L. Barrett-Lennard, pers. comm.). 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, 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 West Coast Transient stocks.

POPULATION SIZE

Off British Columbia, approximately 200 offshore killer whales were identified between 1989 and 1993 (Ford et al. 1994), and 20 of these individuals have also been seen off California (Black et al. 1997). Using only good quality photographs that clearly show characteristics of the dorsal fin and saddle patch region, an additional 11 offshore killer whales that were not previously known have been identified off the California coast, bringing the total number of known individuals in this population to 211. This is certainly an underestimate of the total population size, because not all animals in this population have been photographed. In the future, it may be possible to estimate the total abundance of this transboundary stock using mark-recapture analyses based on individual photographs. Based on summer/fall shipboard line transect surveys in 2005 (Forney 2007) and 2008 (Barlow 2010), the total number of killer whales within 300 nmi of the coasts of California, Oregon and Washington is estimated to be 691 animals (CV=0.49). There is currently no way to reliably distinguish the different stocks of killer whales from sightings at sea, but photographs of individual animals can provide a rough estimate of the proportion of whales in each stock. A total of 161 individual killer whales photographed off California and Oregon have been determined to belong to the transient (105 whales) and offshore (56 whales) stocks (Black et al. 1997). Using these proportions to prorate the line transect abundance estimate yields an estimate of $56/161 * 691 = 240$ offshore killer whales along the U.S. west coast. This is expected to be a conservative estimate of the number of offshore killer whales, because offshore whales apparently are less frequently seen near the coast (Black et al. 1997), and therefore photographic sampling may be biased towards transient whales. For stock assessment purposes, this combined value is currently the best available estimate of abundance for offshore killer whales off the coasts of California, Oregon and Washington. [Population size of the eastern North Pacific stock of offshore killer whales was estimated with photo-ID mark-recapture methods at 300 whales \(95% Highest Posterior Density Interval \(HPDI\) = 257–373, CV=0.10\), including marked and unmarked individuals encountered between 1988 and 2012 \(Ford et al. 2014\). This effort included 157 encounters of 355 distinct whales, over a broad geographic range from the Aleutian Islands to southern California. The cumulative number of unique animals reported by Ford et al. \(2014\) via a ‘discovery curve’ was not asymptotic, implying that additional numbers of unknown individuals were undocumented. Most encounters \(n=85\) during the photo-ID study were from the southeast Alaska and the Vancouver Island regions, where survey effort was most intense. The fraction of this population that utilizes U.S. waters at any one time is unknown and the number of animals that utilize areas outside of the currently known geographic range \(Aleutian Islands to southern California\) is also unknown.](#)

Minimum Population Estimate

The total number of known offshore killer whales along the U.S. West coast, Canada and Alaska is 211 animals, but it is not known what proportion of time this transboundary stock spends in U.S. waters, and therefore this number is difficult to work with for PBR calculations. A minimum abundance estimate for all killer whales along the coasts of California, Oregon and Washington can be estimated from the 2005-2008 line transect surveys as the 20th percentile of the geometric mean 2005-2008 abundance estimate, or 466 killer whales. Using the same prorating as above, a minimum of $56/161 * 466 = 162$ offshore killer whales are estimated to be in U.S. waters off California, Oregon and Washington. [The minimum population size is calculated as the lower 20th percentile of the estimate \(N=300, CV=0.1\) reported by Ford et al. \(2014\), or 276 animals.](#)

Current Population Trend

No information is available regarding trends in abundance of Eastern North Pacific offshore killer

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

~~No information on current or maximum net productivity rates is available for killer whales in this region.~~ [Annual recruitment rates of 2% \(95% HPDI = 0 – 7%\) were estimated by Ford *et al.* \(2014\) for offshore killer whales, based on a Bayesian mark-recapture model.](#)

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (~~162~~ [276](#)) ~~times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a species of unknown status with no known fishery mortality; Wade and Angliss 1997), resulting in a PBR of 4.6~~ [2.8](#) offshore killer whales per year.

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Fishery Information

~~A summary of information on fisheries that may take animals from this killer whale stock is shown in Table 1 (Carretta *et al.* 2005, Carretta and Enriquez 2006, 2007, 2009a, 2009b). More detailed information on these fisheries is provided in Appendix 1. In the California drift gillnet fishery, no offshore killer whales have been observed entangled (Julian 1997; Julian and Beeson 1998; Cameron and Forney 1999, 2000; Carretta and Chivers 2004, Carretta *et al.* 2005a, 2005b, Carretta and Enriquez 2006, 2007, 2009a, 2009b), but one killer whale from the Eastern North Pacific Transient Stock was observed taken in 1995, and offshore killer whales may also occasionally be entangled. Additional potential sources of killer whale mortality are set gillnets and longlines. In California, an observer program between July 1990 and December 1994 and additional observations between 2000 and 2008 monitored 5–15% of all sets in the large mesh (>3.5") set gillnet fishery for halibut, and no killer whales were observed taken. Based on observations for longline fisheries in other regions (i.e. Alaska; Yano and Dahlheim 1995), fishery interactions may also occur with U.S. West coast pelagic longline fisheries, but no such interactions have been documented to date. [Offshore killer whales have not been documented killed by anthropogenic sources in Alaska or U.S. west coast waters, but it is unlikely that such mortalities would be detected, given the offshore habits of this ecotype and the rarity of encounters. Ford *et al.* \(2014\) reports one offshore killer whale injury \(severed dorsal fin\) due to a vessel strike, but does not report a location or year. It is likely that offshore killer whales are vulnerable to the same anthropogenic threats \(fishery interactions, vessel strikes, sonar impacts\) as other killer whale stocks.](#)~~

Table 1. Summary of available information on the incidental mortality and injury of killer whales (Eastern North Pacific Offshore Stock) in commercial fisheries that might take this species. Mean annual takes are based on 2004–2008 [2012–2016](#) data unless noted otherwise. [No entanglements of killer whales have been observed in the CA swordfish drift gillnet fishery since 1995, when a single animal was killed \(Carretta *et al.* 2018a\). This animal was genetically identified as a transient whale and represents the only killer whale observed entangled in the gillnet fishery over a 27-year period \(Carretta *et al.* 2017, 2018\).](#) Estimates of bycatch for the CA thresher shark/swordfish drift gillnet fishery shown in Table 1 are based on a bycatch model that pools all years of observer data but do not include the observation of a transient killer whale in the fishery.

Fishery Name	Data Type	Year(s)	Percent Observer Coverage	Observed Mortality	Estimated Annual Mortality	Mean Annual Takes (CV in parentheses)
CA thresher shark/swordfish drift gillnet fishery	Observer	2004 2012	20.6 19%	0	0	0
		2005 2013	20.9 37%	0	0	
		2006 2014	18.5 24%	0	0	
		2007 2015	16.4 20%	0	0	
		2008 2016	13.5 18%	0	0	
Minimum total annual takes						0

Drift gillnet fisheries for swordfish and sharks exist along the entire Pacific coast of Baja California, Mexico and may take animals from this population. Quantitative data are available only for the Mexican swordfish drift gillnet fishery, which uses vessels, gear, and operational procedures similar to those in the U.S. drift gillnet fishery, although nets may be up to 4.5 km long (Holts and Sosa-Nishizaki 1998). The fleet increased from two vessels in 1986 to 31 vessels in 1993 (Holts and Sosa-Nishizaki 1998). The total number of sets in this fishery in 1992 can be estimated from data provided by these authors to be approximately 2700, with an observed rate of marine mammal bycatch of 0.13 animals per set (10 marine mammals in 77 observed sets; Sosa-Nishizaki et al. 1993). This overall mortality rate is similar to that observed in California driftnet fisheries during 1990-95 (0.14 marine mammals per set; Julian and Beeson, 1998), but species-specific information is not available for the Mexican fisheries. Previous efforts to convert the Mexican swordfish driftnet fishery to a longline fishery have resulted in a mixed fishery, with 20 vessels alternately using longlines or driftnets, 23 using driftnets only, 22 using longlines only, and seven with unknown gear type (Berdegué 2002).

STATUS OF STOCK

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

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

Killer whales have a cosmopolitan distribution, ranging from equatorial to polar waters, with highest densities found in coastal temperate waters (Forney and Wade 2006). Along the west coast of North America, killer whales occur along the entire Alaskan coast as far north as Barrow (George *et al.* 1994, Lowry *et al.* 1987, Clarke *et al.* 2013), in British Columbia and Washington inland waterways (Bigg *et al.* 1990), and along the outer coasts of Washington, Oregon, and California (Barlow and Forney 2007). Seasonal and year-round occurrence has been noted for killer whales throughout Alaska (Braham and Dahlheim 1982) and in the intra-coastal waterways of British Columbia and Washington State, where pods have been labeled as ‘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). Through examination of photographs of recognizable individuals and pods, movements of whales between Prince William Sound and Kodiak Island have been observed (Matkin *et al.* 1999) and whales identified in Southeast Alaska have been observed in Prince William Sound, British Columbia, and Puget Sound (Leatherwood *et al.* 1990, Dahlheim *et al.* 1997).

Genetic studies provide evidence that the ‘resident’ and ‘transient’ types are distinct (Stevens *et al.* 1989, Hoelzel 1991, Hoelzel and Dover 1991, Hoelzel *et al.* 1998, Morin *et al.* 2010). Analyses of complete mitochondrial genomes indicates that transient killer whales should be recognized as a separate species, and that, pending additional data, resident killer whales should be recognized as a separate subspecies (Morin *et al.* 2010). The genetic data results support previous lines of evidence for separation of the transient and resident ecotypes, including differences in 1) acoustic dialects; 2) skull features; 3) morphology; 4) feeding specializations; and 5) a lack of interbreeding between the two sympatric ecotypes (Krahn *et al.* 2004).

Most sightings of the Eastern North Pacific Southern Resident stock of killer whales have occurred in the summer in inland waters of Washington and southern British Columbia. However, pods belonging to this stock have also been sighted in coastal waters off southern Vancouver Island and Washington (Bigg *et al.* 1990, Ford *et al.* 2000, NWFSC unpubl. data). The complete winter range of this stock is uncertain. Of the three pods comprising this stock, one (J1) is commonly sighted in inshore waters in winter, while the other two (K1 and L1) apparently spend more time offshore (Ford *et al.* 2000). These latter two pods have been sighted as far south as Monterey Bay and central California in recent years. They sometimes have also been seen entering the inland waters of Vancouver Island through Johnstone Strait in the spring (Ford *et al.* 2000), suggesting that they may spend time along the outer coast of Vancouver Island during the winter. 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.). Passive autonomous acoustic recorders have recently 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 recently 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.

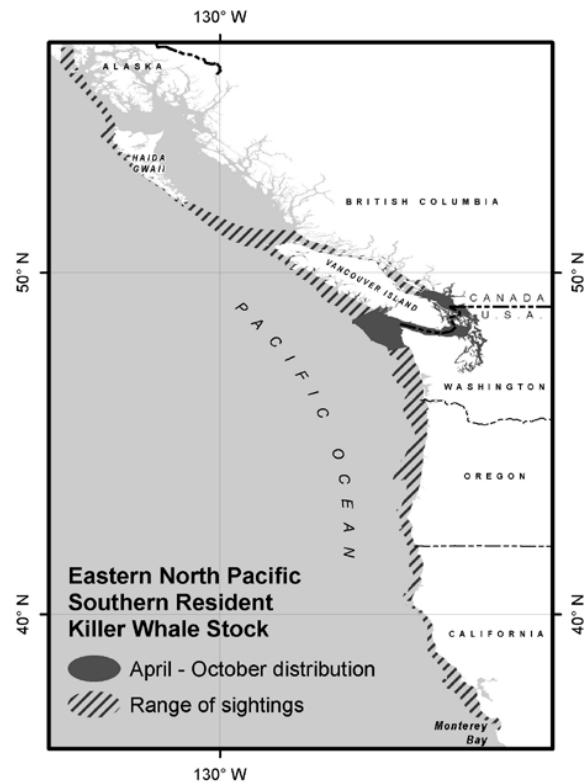


Figure 1. Approximate April - October distribution of the Eastern North Pacific Southern Resident killer whale stock (shaded area) and range of sightings (diagonal lines).

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. Photo-identification of individual whales through the years has advanced knowledge of this stock's structure, behaviors, and movements. 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 83–77 whales in 2016–2017 (Fig. 2; Ford *et al.* 2000; [Center for Whale Research 2016](#)[2017](#)). 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 2016 through 1 July 2017 includes no new calves and the deaths of three post-reproductive age females, a young adult male, and a young reproductive age female and her dependent calf.](#) ~~The most recent census spanning 1 July 2015 through 1 July 2016 includes five new calves (three male, one female, one sex unk.) and the deaths of one of the calves (sex unk.), a post-reproductive age female, and young adult male reproductive age adult female (that was pregnant with a female neonate), and a calf of unknown sex. This does not include the mortality of two post-reproductive females, a reproductive age female and her dependent male calf, or a young adult male. Nor does this include a stillborn fetus that was observed being pushed at the surface by its presumed mother (Durban *et al.* 2016).~~

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 83–77 animals.

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 annually. Between 1974 and the mid-1990s, the Southern Resident stock increased

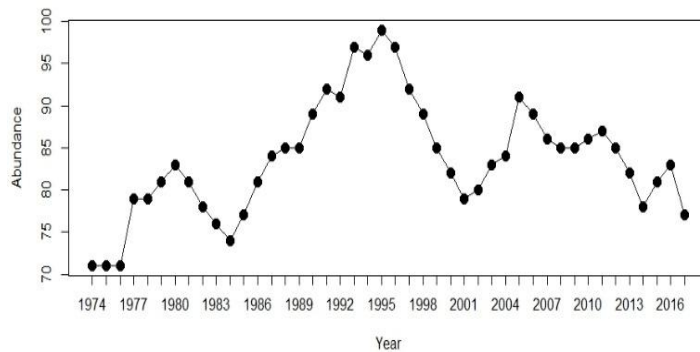


Figure 2. Population of Eastern North Pacific Southern Resident stock of killer whales, 1974- 2017. Each year's count includes animals first seen and first missed; a whale is considered first missed the year after it was last seen alive (Ford *et al.* 2000; [Center for Whale Research 2016](#)[2017](#)).

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 and currently stands at ~~77~~ 83 animals as of the ~~2016~~-2017 census (Ford *et al.* 2000; Center for Whale Research 2017).

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 (~~77~~ 83) times one-half the maximum net growth rate for Alaska resident killer whales ($\frac{1}{2}$ of 3.5%) times a recovery factor of 0.1 (for an endangered stock, Wade and Angliss 1997), resulting in a PBR of ~~0.14~~ 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 typically less than 10% (Erstad *et al.* 1996, Pierce *et al.* 1994, Pierce *et al.* 1996, NWIFC 1995). Fishing effort in the inland waters drift gillnet fishery has declined considerably since 1994 because far fewer vessels participate today (NOAA West Coast Region). 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).

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). 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 this stock is zero.

Other Mortality

No human-caused killer whale mortality or serious injuries of southern resident killer whales were reported from non-fisheries sources ~~in 2011-2015~~ during 2012-2016 (Carretta *et al.* 2017 2018). 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. The annual known level of non-fishery human-caused mortality for this stock over the past five years (~~2010-2014~~ 2012-2016) is zero animals per year. In spring 2016, a young adult male, L95, was found to have died of a fungal infection that may have been related to a satellite tag deployment approximately 5 weeks prior to its death. In fall 2016 another young adult male, J34, was found dead in the northern Georgia Strait. The necropsy indicated that the whale died of blunt force trauma to the head and the source of trauma is still under investigation.

STATUS OF STOCK

Total annual fishery mortality and serious injury for this stock (0) is not known to exceed 10% of the calculated PBR (0-14 [0.13](#)) and, therefore, appears to be insignificant and approaching zero mortality and serious injury rate. The estimated annual level of human-caused mortality and serious injury of zero animals per year does not exceed the PBR (0-14 [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).

Habitat Issues

Several potential risk factors identified for this population have habitat implications. 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). In 2011 vessel approach regulations were implemented to restrict vessels from approaching closer than 200m. This population appears to be Chinook salmon specialists (Ford and Ellis 2006, Hanson *et al.* 2010, Ford *et al.* 2016), although other species, such as chum, pink, and coho salmon also appear to be important elements of the diet (Ford *et al.* 1998, [Ford *et al.* 2016](#)). There is evidence that changes in Chinook abundance have affected this population (Ford *et al.* 2009, Ward *et al.* 2009). In addition, the high trophic level and longevity of the animals has predisposed them to accumulate levels of contaminants that are high enough to cause potential health impacts. In particular, there is recent evidence of extremely high levels of flame retardants in young animals (Krahn *et al.* 2007, 2009).

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

Once common throughout the Northern Hemisphere, the gray whale was extinct in the Atlantic by the early 1700s (Fraser 1970; Mead and Mitchell 1984), though one but anomalous sightings occurred in the Mediterranean Sea in 2010 (Scheinin *et al.* 2011) and another off Namibia in 2013 (Elwen and Gridley 2013). Gray whales are now only commonly found in the North Pacific. Genetic comparisons indicate there are distinct “Eastern North Pacific” (ENP) and “Western North Pacific” (WNP) population stocks, with differentiation in both mtDNA haplotype and microsatellite allele frequencies (LeDuc *et al.* 2002; Lang *et al.* 2011a; Weller *et al.* 2013).

During summer and fall, most whales in the ENP population feed in the Chukchi, Beaufort and northwestern Bering Seas (Fig. 1). An exception to this is the relatively small number of whales (are approximately 200) whales that summer and feed along the Pacific coast between Kodiak Island, Alaska and northern California (Darling 1984, Gosho *et al.* 2011, Calambokidis *et al.* 2012), referred to as the “Pacific Coast Feeding Group” (PCFG). Three primary wintering lagoons in Baja California, Mexico are utilized, and some females are known to make repeated returns to specific lagoons (Jones 1990). Genetic substructure on the wintering grounds is indicated by significant differences in mtDNA haplotype frequencies between females (mothers with calves) using two of the primary calving lagoons and females sampled in other areas (Goerlitz *et al.* 2003). Other research identified a small, but significant departure from panmixia between two of the lagoons using nuclear data, although no significant differences were identified using mtDNA (Alter *et al.* 2009).

Tagging, photo-identification and genetic studies show that some whales identified in the WNP off Russia have been observed in the ENP, including coastal waters of Canada, the U.S. and Mexico (Lang 2010; Mate *et al.* 2011; Weller *et al.* 2012; Urbán *et al.* 2013, Mate *et al.* 2015). In combination, these studies have recorded a total of 27 gray whales observed in both the WNP and ENP. Despite this overlap, significant mtDNA and nDNA differences are found between whales in the WNP and those summering in the ENP (Lang *et al.* 2011a).

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

Photo-identification studies between northern California and northern British Columbia provide data on the abundance and population structure of PCFG whales (Calambokidis *et al.* 2012). Gray whales using the study area in summer and autumn include two components: **1)** whales that frequently return to the area, display a high degree of intra-seasonal “fidelity” and account for a majority of the sightings between 1 June and 30 November. Despite movement and interchange among sub-regions of the study area, some whales are more likely to return to the same sub-region where they were observed in previous years; **2)** “visitors” from the northbound migration that are sighted only in one year, tend to be seen for shorter time periods in that year, and are encountered in more limited areas. Photo-



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

identification (Gosho *et al.* 2011; Calambokidis *et al.* 2012) and satellite tagging (Mate *et al.* 2010; Ford *et al.* 2012) studies have documented some PCFG whales off Kodiak Island, the Gulf of Alaska and Barrow, Alaska, well to the north of the pre-defined 41°N to 52°N boundaries used in some PCFG-related analyses (e.g. abundance estimation).

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

Lang *et al.* (2011b) assessed stock structure of ENP whales from different feeding grounds using both mtDNA and eight microsatellite markers. Significant mtDNA differentiation was found when samples from individuals (n=71) sighted over two or more years within the seasonal range of the PCFG were compared to samples from whales feeding north of the Aleutians (n=103), and when PCFG samples were compared to samples collected off Chukotka, Russia (n=71). No significant differences were found when these same comparisons were made using microsatellite data. The authors concluded that (1) the significant differences in mtDNA haplotype frequencies between the PCFG and whales sampled in northern areas indicates that use of some feeding areas is being influenced by internal recruitment (e.g., matrilineal fidelity), and (2) the lack of significance in nuclear comparisons suggests that individuals from different feeding grounds may interbreed. The level of mtDNA differentiation identified, while statistically significant, was low and the mtDNA haplotype diversity found within the PCFG was similar to that found in the northern strata. Lang *et al.* (2011b) suggested this could indicate recent colonization of the PCFG but could also be consistent with external recruitment into the PCFG. An additional comparison of whales sampled off Vancouver Island, British Columbia (representing the PCFG) and whales sampled at the calving lagoon at San Ignacio also found no significant differences in microsatellite allele frequencies, providing further support for interbreeding between the PCFG and the rest of the ENP stock (D'Intino *et al.* 2012). Lang and Martien (2012) investigated potential immigration levels into the PCFG using simulations and produced results consistent with the empirical (mtDNA) analyses of Lang *et al.* (2011b). Simulations indicated that immigration of >1 and <10 animals per year into the PCFG was plausible, and that annual immigration of 4 animals/year produced results most consistent with the empirical study.

While the PCFG is recognized as a distinct feeding aggregation (Calambokidis *et al.* 2012; Mate *et al.* 2010; Frasier *et al.* 2011; Lang *et al.* 2011b; IWC 2012), the status of the PCFG as a population stock remains unresolved (Weller *et al.* 2013). A NMFS gray whale stock identification workshop held in 2012 included a review of available photo-identification, genetic, and satellite tag data. The report of the workshop states “there remains a substantial level of uncertainty in the strength of the lines of evidence supporting demographic independence of the PCFG.” (Weller *et al.* 2013). The NMFS task force, charged with evaluating stock status of the PCFG, noted that “both the photo-identification and genetics data indicate that the levels of internal versus external recruitment are comparable, but these are not quantified well enough to determine if the population dynamics of the PCFG are more a consequence of births and deaths within the group (internal dynamics) rather than related to immigration and/or emigration (external dynamics).” Further, given the lack of significant differences found in nuclear DNA markers between PCFG whales and other ENP whales, the task force found no evidence to suggest that PCFG whales breed exclusively or primarily with each other, but interbreed with ENP whales, including potentially other PCFG whales. Additional research is needed to better identify recruitment levels into the PCFG and further assess the stock status of PCFG whales (Weller *et al.* 2013). In contrast, the task force noted that WNP gray whales should be recognized as a population stock under the MMPA, and NMFS prepared a separate report for WNP gray whales in 2014. Because the PCFG appears to be a distinct feeding aggregation and may warrant consideration as a distinct stock in the future, separate PBRs are calculated for the PCFG to assess whether levels of human-caused mortality are likely to cause local depletion.

POPULATION SIZE

Systematic counts of gray whales migrating south along the central California coast have been conducted by shore-based observers at Granite Canyon most years since 1967 (Fig. 2). The most recent estimate of abundance for the ENP population is from the ~~2010/2011~~ **2015/2016** southbound survey and is **26,960 (CV=0.05)** ~~20,990 (CV=0.05)~~ whales (Durban *et al.* ~~2013~~ **2017**) (Fig. 2).

Photographic mark-recapture abundance estimates for PCFG gray whales between 1998 and ~~2012~~ **2015**, including estimates for a number of smaller geographic areas within the IWC-defined PCFG region (41°N to 52°N), are reported in Calambokidis *et al.* (~~2014~~ **2017**). The ~~2012-2015~~ abundance estimate for the defined range of the PCFG between 41°N to 52°N is ~~209-243~~ **whales** (SE=~~15.4~~ **18.9**; CV= ~~0.07~~ **0.08**).

Eastern North Pacific gray whales experienced an unusual mortality event (UME) in 1999 and 2000, when large numbers of emaciated animals stranded along the west coast of North America (Moore *et al.*, 2001; Gulland *et al.*, 2005). Over 60% of the dead whales were adults, compared with previous years when calf strandings were more

common. Several factors following this UME suggest that the high mortality rate observed was a short-term, acute event and not a chronic situation or trend: 1) in 2001 and 2002, strandings decreased to levels below UME levels (Gulland *et al.*, 2005); 2) average calf production returned to levels seen before 1999; and 3) in 2001, living whales no longer appeared emaciated. Oceanographic factors that limited food availability for gray whales were identified as likely causes of the UME (LeBouef *et al.* 2000; Moore *et al.* 2001; Minobe 2002; Gulland *et al.* 2005), with resulting declines in survival rates of adults during this period (Punt and Wade 2012). The population has recovered to levels seen prior to the UME of 1999-2000 [and the current estimate of abundance is the highest that has been recorded in the 1967-2015 time series](#) (Fig. 2).

Gray whale calves have been counted from Piedras Blancas, a shore site in central California, in 1980-81 (Poole 1984a) and each year from 1994 to 2012 (Perryman *et al.* 2002; Perryman and Weller 2012). In 1980 and 1981, calves comprised 4.7% to 5.2% of the population (Poole 1984b). Calf production indices, as calculated by dividing northbound calf estimates by estimates of population abundance (Laake *et al.* 2012), ranged between 1.3–8.8% (mean=4.2%) during 1994-2012. Annual indices of calf production include impacts of early postnatal mortality but may overestimate recruitment because they exclude possibly significant levels of killer whale predation on gray whale calves north of the survey site (Barrett Lennard *et al.* 2011). The relatively low reproductive output reported is consistent with little or no population growth over the time period (Laake *et al.* 2012; Punt and Wade 2012).

Minimum Population Estimate

The minimum population estimate (N_{MIN}) for the ENP stock is calculated from Equation 1 from the PBR Guidelines (Wade and Angliss 1997): $N_{MIN} = N / \exp(0.842 \times [\ln(1 + [CV(N)]^2)]^{1/2})$. Using the 2010/11 [2015/2016](#) abundance estimate of ~~20,990~~ [26,960](#) and its associated CV of 0.05 (Durban *et al.* 2013), N_{MIN} for this stock is ~~20,125~~ [25,849](#).

The minimum population estimate for PCFG gray whales is calculated as the lower 20th percentile of the log-normal distribution of the ~~2012~~ [2015](#) mark-recapture estimate of ~~209~~ [243](#) (CV=0.070.08), or ~~197~~ [227](#) animals.

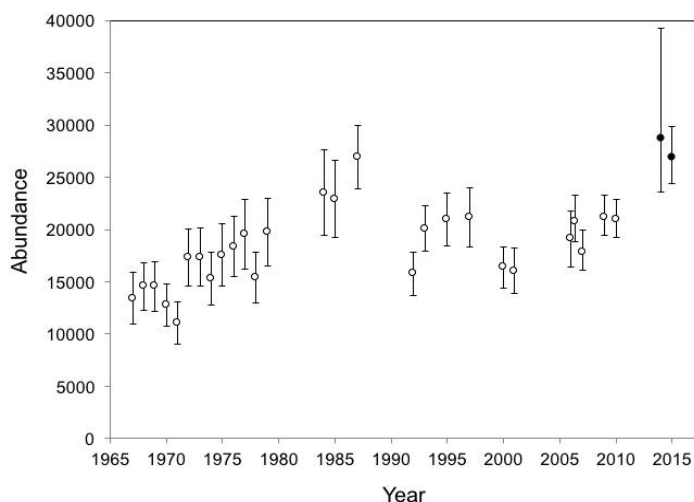


Figure 2. Estimated abundance of Eastern North Pacific gray whales from NMFS counts of migrating whales past Granite Canyon, California. Open circles represent abundance estimates and 95% confidence intervals reported by Laake *et al.* (2012) [and Durban *et al.* \(2015\)](#). Closed circles represent estimates and 95% posterior highest density intervals reported by Durban *et al.* (2013/2017) for the ~~2006/7, 2007/8, 2009/10, and 2010/11~~ [2014/2015 and 2015/2016](#) migration seasons.

Current Population Trend

The population size of the ENP gray whale stock has increased over several decades despite an UME in 1999 and 2000 ~~and has been relatively stable since the mid-1990s~~ (see Fig. 2). [Durban *et al.* \(2017\) noted that a recent 22% increase in ENP gray whale abundance over 2010/2011 levels is consistent with high observed and estimated calf production \(Perryman *et al.* 2017\). Recent increases in abundance also support hypotheses that gray whales may experience more favorable feeding conditions in arctic waters due to an increase in ice-free habitat that might result in increased primary productivity in the region \(Perryman *et al.* 2002, Moore 2016\). Abundance estimates of PCFG whales increased from 1998 through 2004, remained stable for the period 2005-2010, and have steadily increased during the 2011-2015 time period \(Calambokidis *et al.* 2017\). Abundance estimates of PCFG gray whales reported by Calambokidis *et al.* \(2014\) show a high rate of increase in the late 1990s and early 2000s, but have been relatively stable since 2003.](#)

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Using abundance data through 2006/07, an analysis of the ENP gray whale population led to an estimate of R_{max} of 0.062, with a 90% probability the value was between 0.032 and 0.088 (Punt and Wade 2012). This value of

Rmax is also applied to PCFG gray whales, as it is currently the best estimate of Rmax available for gray whales in the ENP.

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the ENP stock of gray whales is calculated as the minimum population size (20,125 25,849), times one-half of the maximum theoretical net population growth rate ($\frac{1}{2} \times 6.2\% = 3.1\%$), times a recovery factor of 1.0 for a stock above MNPL (Punt and Wade 2012), or ~~624~~ 801 animals per year.

The potential biological removal (PBR) level for PCFG gray whales is calculated as the minimum population size (~~497~~ 227 animals), times one half the maximum theoretical net population growth rate ($\frac{1}{2} \times 6.2\% = 3.1\%$), times a recovery factor of 0.5 (for a population of unknown status), resulting in a PBR of ~~31~~ 3.5 animals per year. Use of the recovery factor of 0.5 for PCFG gray whales, rather than 1.0 used for ENP gray whales, is based on uncertainty regarding stock structure (Weller *et al.* 2013) and guidelines for preparing marine mammal stock assessments which state that “Recovery factors of 1.0 for stocks of unknown status should be reserved for cases where there is assurance that N_{min} , R_{max} , and the kill are unbiased and where the stock structure is unequivocal” (NMFS 2005). Given uncertainties in the levels of external versus internal recruitment of PCFG whales described above, the equivocal nature of the stock structure, and the small estimated population size of the PCFG, NMFS will continue to use the default recovery factor of 0.5 for PCFG gray whales.

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Fisheries Information

~~No gray whales were observed entangled in California gillnet fisheries between 2008 and 2012 (Carretta and Enriquez 2009, 2010, 2012a, 2012b, Carretta et al., 2014a.), but previous mortality in the swordfish drift gillnet fishery has been observed (Carretta et al. 2004) and there have been recent sightings of free-swimming gray whales entangled in gillnets (Table 1). The California large-mesh drift gillnet fishery for swordfish and thresher shark includes 4 observed entanglement records of gray whales from 8,845 observed fishing sets over the 27-year period 1990-2016 (Carretta et al. 2018a). The estimated bycatch of gray whales in this fishery for the most recent 5-year period is 2.1 (CV=0.76) whales, or 0.4 whales annually (Carretta et al. 2018a). By comparison, the more coastal set gillnet fishery for halibut and white seabass has no observations of gray whale entanglements from over 10,000 observed sets for the same time period. This compares with 11 opportunistically documented gillnet entanglements of gray whales in U.S. west coast waters during the most recent 5 year period of 2012-2016, including one self-report from a set gillnet vessel operator (Carretta et al. 2018b). The origin of the gillnet gear for the remaining 10 entanglements is unknown. Alaska gillnet fisheries also interact with gray whales, but these fisheries largely lack observer programs, including those in Bristol Bay known to interact with gray whales. Some gillnet entanglements involving gray whales along the coasts of Washington, Oregon, and California may involve gear set in Alaska and/or Mexican waters and carried south and/or north during the annual migration.~~

Table 1. Entanglement mortality and serious injury of gray whales, 2012-2016 (Carretta *et al.* 2018a, 2018b). Fractional bycatch estimates in swordfish drift gillnets during 2014-2016 result from a model that incorporates all years of observer data for bycatch prediction, thus bycatch estimates can be positive even when no bycatch is observed. Entanglement in other fisheries is derived from strandings and at-sea sightings of entangled whales and thus represent minimum impacts because they are opportunistically documented (Carretta *et al.* 2018b). Mortality and injury information, where possible, is assigned to either the ENP gray whale stock or PCFG whales.

Fishery Name	Year(s)	Data Type	Percent Observer Coverage	Observed mortality (+ serious injury)	Estimated mortality (CV)	Mean annual takes 2012-2016 (CV)
CA/OR thresher shark/swordfish drift gillnet	2012	observer	19%	0 (0)	0 (n/a)	0.4 (0.76) (ENP stock)
	2013		37%	1 (0)	1 (n/a)	
	2014		24%	0 (0)	0.1 (5.9)	
	2015		20%	0 (0)	0.7 (2.1)	
	2016		18%	0 (0)	0.5 (2.4)	
	2012-2016		23%	ENP 1 (0)	2.1 (0.76)	
CA halibut and white seabass set gillnet		vessel self-report	n/a	ENP 0 (0.75)	n/a	ENP 0.15 (n/a)
CA Dungeness crab pot	2012-2016	strandings + sightings	n/a	ENP 1 (1.75)	n/a	ENP 0.55 (n/a)
OR Dungeness crab pot				PCFG 1 (0)		PCFG 0.2 (n/a)
				ENP 0 (0.75)		ENP 0.15 (n/a)

<u>Fishery Name</u>	<u>Year(s)</u>	<u>Data Type</u>	<u>Percent Observer Coverage</u>	<u>Observed mortality (+ serious injury)</u>	<u>Estimated mortality (CV)</u>	<u>Mean annual takes 2012-2016 (CV)</u>
<u>Cod pot fishery</u>				ENP 0 (0.75)		ENP 0.15 (n/a)
<u>Unidentified pot/trap fishery</u>				ENP 1 (7.25) PCFG 0 (1.5)		ENP 1.6 (n/a) PCFG 0.3 (n/a)
<u>Unidentified gillnet fishery</u>				ENP 3 (5.5)		ENP 1.7 (n/a)
<u>Unidentified fishery interactions</u>				ENP 2 (12) PCFG 0 (1)		ENP 2.8 (n/a) PCFG 0.2 (n/a)
<u>Marine debris entanglement</u>				ENP 1 (0.75)		ENP 0.35 (n/a)
<u>Tribal crab pot gear</u>	2012-2016	self-report	n/a	PCFG 0 (0.75)		PCFG 0.15 (n/a)
<u>Totals</u>				ENP 9.1 (29.5) PCFG 1 (3.25)		ENP 7.9 (n/a) PCFG 0.85 (n/a)

Entanglement in commercial pot and trap fisheries along the U.S. west coast is another source of gray whale mortality and serious injury (Carretta *et al.* 2018b). Most data on human-caused mortality and serious injury of gray whales are from strandings, including at-sea reports of entangled animals alive or dead (Carretta *et al.* 2013, 2014b 2018b). Strandings represent only a fraction of actual gray whale deaths (natural or human-caused), as reported by Punt and Wade (2012), who estimated that only 3.9% to 13.0% of gray whales that die in a given year end up stranding and being reported. This estimate of carcass detection, however, also included sparsely-populated coastlines of Baja California, Canada, and Alaska, for which the rate of carcass detection would be expected to be low. Since most U.S. cases of human-caused serious injury and mortality are documented from Washington, Oregon, and California waters, the Punt and Wade (2012) estimate of carcass recovery is not applicable to most documented cases. An appropriate correction factor for undetected anthropogenic mortality and serious injury of gray whales is currently not available.

A summary of human-caused mortality and serious injury resulting from unknown fishery and marine debris sources (mainly pot/trap or net fisheries) is given in Table 1 for the most recent 5-year period of 2012 to 2016 (Carretta *et al.* 2018b). 2008 to 2012. Total observed and estimated entanglement-related human-caused fishery mortality and serious injury for ENP gray whales is 7.9 whales annually (Table 1). The mean annual entanglement-related serious injury and mortality level for PCFG gray whales is 0.85 whales, based on one observed death in CA Dungeness crab pot gear and three serious injuries in other fishing gear (Table 1). 22.25 animals (8 serious injuries, 8.25 prorated serious injuries, and 6 deaths), or 4.45 whales per year (Table 1). Total observed human-caused fishery mortality and serious injury for gray whales observed in the PCFG range and season for the period 2008 to 2012 is 0.75 animals (0.75 prorated serious injuries), or 0.15 whales per year (Table 1). Three gray whales from Table 1 (one death and two serious injuries) were detected in California waters during the known PCFG season, but were south of the area recognized by the IWC as the PCFG management area. It is possible that some of these whales could be PCFG whales, but no photographic identifications were available to establish their identity. They are included in ENP gray whale serious injury and death totals. In addition to the mortality and serious injury totals listed above, there were 5 non-serious entanglement injuries of gray whales between 2012 and 2016 (Carretta *et al.* 2018b). Three non-serious injuries involved ENP gray whales, each with one record associated with the following sources: CA Dungeness crab pot fishery, unknown Dungeness crab pot fishery, and unidentified fishery interaction. During the same period, there were two non-serious injuries involving PCFG whales, one in tribal crab pot gear and the other in an unidentified gillnet fishery.

Unidentified whales represent approximately 15% of entanglement cases along the U.S. West Coast, (Carretta 2018). Observed entanglements may lack species IDs due to rough seas, distance from whales, or a lack of cetacean identification expertise. In previous 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) collectively found to be statistically-significant predictors of known-species entanglement cases (Carretta 2018). The species model was used to assign species ID probabilities for 21 unidentified whale entanglement cases ('novel data') during 2012-2016. The sum of species assignment probabilities for this 5-year period result in an additional 5.8 gray whale entanglements for 2012-2016. Of these 5.8 entanglements, only 0.8 occurred within the geographic range and seasonal limits considered to represent PCFG gray whales, while the remaining 5 are considered to be ENP gray whales. Unidentified whale entanglements typically involve whales seen at-sea with unknown gear configurations that are prorated to represent 0.75 serious injuries per entanglement case. Thus it is estimated that at least $5 \times 0.75 = 3.75$ additional ENP gray whale and $0.8 \times 0.75 = 0.6$ PCFG serious injuries are represented from the

21 unidentified whale entanglement cases during 2012-2016. This represents 0.75 ENP gray whales and 0.1 PCFG gray whales annually.

Table 1. Human-caused deaths and serious injuries (SI) of gray whales from fishery-related and marine debris sources for the period 2008 to 2012 as recorded by NMFS stranding networks and observer programs.

Date of observation	Location	PCFG range N41-N52 AND season?	Description	Determination (SI Prorate value)
13-Oct-2012	Fort Bragg, CA	No	Entangled animal report; animal reported with rope around the peduncle which wasn't seen in photographs but photos did show green gillnet with cuts to the head; animal disappeared and final status is unknown.	SI
31-Aug-2012	Los Angeles, CA	No	Animal first detected near San Diego. Subadult gray whale reported entangled with small gauge, dark-colored line deeply embedded around its tail stock. Little gear trails. Entanglement was once more involved as indicated by scars on the animal's body. Animal in very poor condition - emaciated, scarred and a heavy load of cyamid amphipods. Black line around peduncle, 20-ft trailing; observed off San Diego on 8/31, completely disentangled off L.A. 9/6, stranded dead 9/14/12.	Dead
22-Aug-2012	Prince William Sound, AK	No	Whale sighted by tour boat. Few details, other than part of a fishing net was observed being trailed from a gray whale's fin. Photos apparently available, but have not been located. Prince William Sound. Extent and severity of entanglement unknown.	SI (0.75)
16-Jun-2012	Prince William Sound, AK	No	30' gray whale in Prince William Sound entangled in gear. Thrashing at surface and moving at 4-5 knots. No wounds or chafing was observed. Gillnet, corkline (at least 12 floats), and leadline observed over animal's rostrum, body, and tailstock. Both pectoral flippers appeared pinned to body. Animal later appeared tired and was swimming at 2 knots. It was not relocated. Assigned serious injury because gear appears to be constricting movement of whale's flippers.	SI
13-May-2012	Monterey, CA	No	Animal entangled through mouth in at least two sets of suspected pot gear that hang below. Animal anchored with a short scope in 28 feet of water to suspected pots. Bundle of gear, including 4 buoys lie under animal. Animal having some difficulty getting to surface. Animal eventually disentangled, but results of entanglement may still be life-threatening.	SI
8-May-2012	Eureka, CA	No	Entangled animal report; deep cuts from rope around peduncle and lacerations at fluke notch and lateral edge of fluke; successfully disentangled but long-term survival noted as questionable. Gear was collected and identified as Dungeness crab pot gear. Animal entirely freed of gear. Animal in fair condition and slightly emaciated. Deep cuts (~2 inches) from the rope around the peduncle remained. Gear was recovered. Results of entanglement may still be life-threatening.	SI
5-May-2012	Monterey, CA	No	Whale watch vessel noticed from images taken of a 20-25 foot gray whale they had been observing earlier in the day, that animal was actually entangled. A small gauge line, likely from right side of mouth goes over the animal's back, and over blowholes, to left side of mouth. No buoys or trailing line were observed. Animal in fair condition. Animal sighted next day by whale watch vessel. Confirmed mouth entanglement, appears to be strapping material.	SI (0.75)
28-Apr-2012	Fort Bragg, CA	No	Small gray whale off Fort Bragg Fort Bragg, CA, in company of two other animals, trailing two buoys.	SI (0.75)
21-Apr-2012	San Simeon, CA	No	Rope like marks on caudal peduncle. Rope impression on pectoral fin. Photos taken.	Dead
17-Apr-2012	Laguna Beach, CA	No	40-foot gray whale reported entangled with approximately 150 feet of line trailing. Four sponges bullet buoys lie along the left side of the animal. Entanglement involves the mouth, a wrap over the head, and the left pectoral flipper. Entanglement appears recent. Partially disentangled on 5/3/12 by fishermen.	SI (0.75)
24-Mar-2012	San Diego, CA	No	Entangled animal report; gillnet gear around peduncle; response effort resulted in successful disentanglement with >100 ft of pink gillnet removed from animal, but animal subsequently observed dead on 03/27 (floating, skin sample taken, no necropsy). Net	Dead

			removed on 03/24 found to contain one dead ca sea lion and three dead sharks.	
28-Jan-2012	San Diego, CA	No	Entangled animal report; towing two orange buoys and at least 150 feet of line; unknown fishery, reported as possible gillnet; no response effort.	SI(0.75)
17-Jan-2012	Unimak Pass, AK	No	A 40' whale was caught in cod-pot gear near Unimak Pass. Lines were cut by boat crew and buoys were recovered, however, the pot and some line remained in the water. Any line possibly remaining on animal thought to be minimal. Gray whale species determination made following extensive questioning by local biologist. Determination: prorated serious injury because gear possibly remains on animal.	SI(0.75)
25-Aug-2011	San Mateo, CA	No	One white "crab-pot" buoy next to body by left pectoral fin; float stayed next to body and did not change position; animal remained in same position - possibly anchored; only observed for ~2 min; not resighted, no rescue, outcome unknown.	SI
12-Sep-2010	Central Bering Sea	No	Bering Sea / Aleutian Islands flatfish trawl fishery: 12 m animal caught in gear. Photos taken.	Dead
11-May-2010	Orange County CA	No	Free-swimming animal entangled in gillnet; animal first observed inside Dana Point Harbor on 5/11/10; animal successfully disentangled on 5/12/10 & swam out of harbor; animal observed alive in surf zone for several hours on 5/14/10 off Doheny State Beach before washing up dead on beach	Dead
7-May-2010	Cape Foulweather OR	No	Entangled in 3 crab pots, whale not relocated.	SI(0.75)
16-Apr-2010	Seaside OR	No	27-ft long gray whale stranded dead, entangled in crab-pot gear	Dead
8-Apr-2010	San Francisco CA	No	Rope wrapped around caudal peduncle; identified as gray whale from photo. Free-swimming, diving. No rescue effort, no resightings, final status unknown	SI
5-Mar-2010	San Diego	No	Free-swimming entangled whale reported by member of the public; no rescue effort initiated; no resightings reported; final status unknown.	SI(0.75)
21-Jul-2009	Trinidad Head CA	Yes	Free-swimming animal with green gillnet, rope & small black floats wrapped around caudal peduncle; report received via HSU researcher on scene during research cruise; animal resighted on 3 Aug; no rescue effort initiated. Photos show rope cutting into caudal peduncle. This whale was re-sighted in 2010 and 2011, still trailing gear. Whale was resighted in 2013 and had shed gear, and was apparently in good health (Jeff Jacobsen, pers. comm.).	-NSI
24-Jun-2009	Clallam County, WA	Yes	Whale found entangled in tribal set gillnet in morning. Net had been set 8 pm previous day. Whale able to breath, but not swim freely and was stationary in net. Right pectoral flipper and head were well wrapped in net webbing. In response to disentanglement attempts, whale reacted violently and swam away. The net was retrieved and found to be torn in two. No confirmation on whether whale was completely free of netting.	SI(0.75)
9-Apr-2009	Sitka, AK	No	Thick black line wrapped twice around whale's body posterior to the eyes was cut and pulled away by private citizen. Animal swam away and dove.	SI(0.75)
25-Mar-2009	Seal Beach CA	No	Free-swimming animal with pink gillnet wrapped around head, trailing 4 feet of visible netting; report received via naturalist on local whale watch vessel; no rescue effort initiated; final status unknown	SI(0.75)
31-Jan-2009	San Diego CA	No	Free-swimming animal towing unidentified pot/trap gear; report received via USCG on scene; USCG reported gear as 4 lobster pots; final status unknown	SI(0.75)
16-Apr-2008	Eel River CA	No	Observed 12 miles west of Eel River by Humboldt State University personnel. It was unknown sex, with an estimated length of 20 ft and in emaciated condition. The animal was described as towing 40-50 feet of line & 3 crab-pot-buoys from the caudal peduncle and moving very slowly. Vessel retrieved the buoys, pulled them and ~20 ft of line onto the deck and cut it loose from the whale. The whale swam away slowly with 20-30 feet of line still entangling the peduncle, outcome unknown. Identification numbers on buoy traced to crab-pot fishery gear that was last fished in Bering Sea in December 2007.	SI

Subsistence/Native Harvest Information

Subsistence hunters in Russia and the United States have traditionally harvested whales from the ENP stock in the Bering Sea, although only the Russian hunt has persisted in recent years (Huelsbeck 1988; Reeves 2002). In 2005, the Makah Indian Tribe requested authorization from NOAA/NMFS, under the MMPA and the Whaling Convention Act, to resume limited hunting of gray whales for ceremonial and subsistence purposes in the coastal portion of their usual and accustomed (U&A) fishing grounds off Washington State (NMFS 2008, 2015). The spatial overlap of the Makah U&A and the summer distribution of PCFG whales has management implications. The [hunt](#) proposal by the Makah Tribe includes time/area restrictions designed to reduce the probability of killing a PCFG whale and to focus the hunt on whales migrating to/from feeding areas to the north. The Makah proposal also includes catch limits for PCFG whales that result in the hunt being terminated if these limits are met. Also, observations of gray whales moving between the WNP and ENP highlight the need to estimate the probability of a gray whale observed in the WNP being taken during a [hunt by the Makah hunt Tribe](#) (Moore and Weller 2013). NMFS has ~~published a notice of intent to prepare an~~ [prepared a draft](#) environmental impact statement (DEIS) on the proposed hunt (NMFS 2012, 2015) and the IWC has evaluated the potential impacts of the proposed hunt and other sources of human-caused mortality on PCFG whales and concluded, with certain qualifications, that the proposed hunt meets the Commission's conservation objectives (IWC 2013). The Scientific Committee [has continued to](#) ~~has not scheduled an implementation review of the impacts of the Makah hunt on whales using summering feeding areas in the WNP, but is continuing to investigate stock structure of north Pacific gray whales and~~ [has convened five workshops on the subject between 2014 and 2018. The objective of the workshops has been to develop a series of range-wide stock structure hypotheses, using all available data sources \(e.g. photo-ID, genetics, tagging\), that can be tested within a modelling framework \(IWC 2017\). Completion of this work is scheduled for 2018-2019, may schedule such a review in the future \(IWC 2013\).](#) In 2012, the IWC approved a 6-year quota (2013-2018) of 744 gray whales, with an annual cap of 140, for Russian and U.S. (Makah Indian Tribe) aboriginals based on the joint request and needs statements submitted by the U.S. and [the Russian Federation](#). The U.S. and [the Russian Federation](#) have agreed that the quota will be shared with an average annual harvest of 120 whales by the Russian Chukotka people and 4 whales by the Makah Indian Tribe. Total takes by the Russian hunt during the past five years were: 130 in 2008, 116 in 2009, 118 in 2010, 128 in 2011, and 143 in 2012, [127 in 2013, 124 in 2014, 125 in 2015, and 120 in 2016 \(International Whaling Commission\).](#) ~~There were no whales taken by the Makah Indian Tribe during that period because their hunt request is still under review.~~ Based on this information, the annual subsistence take averaged ~~127~~ [128](#) whales during the 5-year period from 2008 ~~to 2012~~ [to 2016. The IWC reports a total of 3,787 gray whales harvested from annual aboriginal subsistence hunts for the 32-year period 1985 to 2016, which includes struck and lost whales.](#)

Other Mortality

Ship strikes are a source of mortality [and serious injury](#) for gray whales (Table 2). For the most recent five-year period, 2008-2012 ~~2012-2016~~, the total serious injury and mortality of ENP gray whales attributed to ship strikes is ~~9.8~~ [4](#) animals (including ~~7.4~~ [deaths and 2 non-serious injuries](#)) or ~~0.8~~ [whales annually](#); ~~2 serious injuries, and 0.8 prorated serious injuries, or 2.0 whales per year (Table 2, Carretta et al. 2013, Carretta et al. 2014b, Carretta et al. 2018b).~~ ~~The total~~ [Total](#) ship strike serious injury and mortality of gray whales observed in the PCFG range and season during this same period is ~~0.52~~ [2](#) animals, or ~~0.1~~ [0.4](#) whales per year (Table 2 [Carretta et al. 2018b](#)). ~~One gray whale ship strike in Table 2 was detected in California waters during the known PCFG season, but was south of the area recognized by the IWC as the PCFG management area. It is possible that this animal could be a PCFG whale, but no photographic identification was available to establish its identity. It is included in ENP gray whale serious injury and death totals. Additional mortality from ship strikes probably goes unreported because the whales either do not strand, are undetected, or do not have~~ [lack](#) obvious signs of trauma.

~~In February 2010, a gray whale stranded dead near Humboldt, CA with parts of two harpoons embedded in the body. Since this whale was likely harpooned during the aboriginal hunt in Russian waters, it would have been counted as "struck and lost" in the harvest data.~~

HABITAT CONCERNS

Near shore industrialization and shipping congestion throughout the migratory corridors of the ENP gray whale stock represent risks by increasing the likelihood of exposure to pollutants and ship strikes, as well as a general degradation of the habitat.

Evidence indicates that the Arctic climate is changing significantly, resulting in a reductions in sea ice cover (Johannessen *et al.* 2004, Comiso *et al.* 2008). These changes are likely to affect gray whales. For example, the summer range of gray whales has greatly expanded in the past decade (Rugh *et al.* 2001). Bluhm and Gradinger (2008) examined the availability of pelagic and benthic prey in the Arctic and concluded that pelagic prey is likely to

increase while benthic prey is likely to decrease in response to climate change. They noted that marine mammal species that exhibit trophic plasticity (such as gray whales which feed on both benthic and pelagic prey) will adapt better than trophic specialists.

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

Table 2. Summary of gray whale serious injuries (SI) and deaths attributed to vessel strikes for the five-year period 2008-2012. No vessel strikes were reported in 2012.

Date of observation	Location	PCFG-range N 41–N 52 AND season?	Description	Determination (SI prorated value)
6-Jun-2011	San Mateo CA	No	Massive hemorrhage into the thorax, blood clots around lungs. Lesions indicate massive trauma. Due to carcass position, the skeleton could not be completely examined (lying on back, top of skull in sand).	Dead
8-Apr-2011	San Francisco CA	No	Crushed mandible.	Dead
12-Feb-2011	Los Angeles CA	No	Private recreational vessel collided with free-swimming animal; animal breached just prior to contact, bouncing off side of vessel; dove immediately following contact & was not resighted; no blood observed in water; final status unknown; skin sample collected from vessel and genetically identified as a female gray whale. Vessel size assumed less than 65 ft and speed unknown.	SI (0.14)
22-Jan-2011	San Diego CA	No	Pleasure sailboat collided with free-swimming animal; animal dove immediately following contact & was not resighted; no blood observed in water; final status unknown. Vessel size assumed less than 65 ft. And speed unknown.	SI (0.14)
12-Mar-2010	Santa Barbara CA	No	21-meter sailboat underway at 13 kts collided with free-swimming animal; whale breached shortly after collision; no blood observed in water; minor damage to lower portion of boat's keel; final status unknown; DNA analysis of skin sample confirmed species.	SI
16-Feb-2010	San Diego CA	No	Free-swimming animal with propeller-like wounds to dorsum.	SI (0.52)
9-Sep-2009	Quileute River WA	Yes	USCG vessel reported to be traveling at 10 knots when they hit the gray whale at noon on 9/9/2009. The animal was hit with the prop and was reported alive after being hit, blood observed in water.	SI (0.52)
1-May-2009	Los Angeles CA	No	Catalina island transport vessel collided with free-swimming calf accompanied by adult animal; calf was submerged at time of collision; pieces of flesh & blood observed in water; calf never surfaced; presumed mortality.	SI
27-Apr-2009	Whidbey Is. WA	No	Large amount of blood in body cavity, bruising in some areas of blubber layer and in some internal organs. Findings suggestive of blunt force trauma likely caused by collision with a large ship.	Dead
5-Apr-2009	Sunset Beach CA	No	Dead stranding; 3 deep propeller-like cuts on right side, just anterior of genital opening; carcass towed out to sea	Dead
4-Apr-2009	Ihwaoo WA	No	Necropsied, broken bones in skull; extensive hemorrhage head and thorax; sub-adult male	Dead
1-Mar-2008	Mexico	No	Carcass brought into port on bow of cruise ship; collision occurred between ports of San Diego and Cabo San Lucas between 5:00 p.m. On 2/28 & 7:20 a.m. On 3/1	Dead
7-Feb-2008	Orange County CA	No	Carcass; propeller-like wounds to left dorsum from mid-body to caudal peduncle; deep external bruising on right side of head; field necropsy revealed multiple cranial fractures	Dead

STATUS OF STOCK

In 1994, the ENP stock of gray whales was removed from the List of Endangered and Threatened Wildlife (the List), as it was no longer considered endangered or threatened under the Endangered Species Act (NMFS 1994). Punt and Wade (2012) estimated the ENP population was at 85% of carrying capacity (K) and at 129% of the

maximum net productivity level (MNPL), with a probability of 0.884 that the population is above MNPL and therefore within the range of its optimum sustainable population (OSP).

Even though the stock is within OSP, abundance will fluctuate as the population adjusts to natural and human-caused factors affecting carrying capacity (Punt and Wade 2012). It is expected that a population close to or at carrying capacity will be more susceptible to environmental fluctuations (Moore *et al.* 2001). The correlation between gray whale calf production and environmental conditions in the Bering Sea may reflect this (Perryman *et al.* 2002; Perryman and Weller 2012). Overall, the population nearly doubled in size over the first 20 years of monitoring, and has fluctuated for the last 30 years, [with a recent increase to over 26,000 whales. Carrying capacity for this stock was estimated at 25,808 whales in 2009 \(Punt and Wade 2012\), however the authors noted that carrying capacity was likely to fluctuate along with environmental changes, especially those related to the productivity of arctic feeding grounds, around its average carrying capacity. This is consistent with a population approaching K.](#)

Based on ~~2008-2012~~ [2012-2016](#) data, the estimated annual level of human-caused mortality and serious injury for ENP gray whales includes Russian harvest (~~127~~ [128](#)), mortality and serious injury from commercial fisheries (~~4.45~~ [7.9](#)), [marine debris \(0.35\)](#), and ship strikes (~~2.0~~ [0.8](#)), [and unidentified whale entanglements assigned as gray whales \(0.75\)](#) totals ~~133~~ [138](#) whales per year, which does not exceed the PBR (~~624~~ [801](#)). Therefore, the ENP stock of gray whales is not classified as a strategic stock.

The IWC completed an implementation review for ENP gray whales (including the PCFG) in 2012 (IWC 2013) and concluded that harvest levels (including the proposed Makah hunt) and other human caused mortality are sustainable, given the ~~current~~ population abundance (Laake *et al.* 2012, Punt and Wade 2012).

PCFG gray whales do not currently have a formal status under the MMPA, [Abundance estimates of PCFG whales increased from 1998 through 2004, remained stable for the period 2005-2010, and have steadily increased during the 2011-2015 time period \(Calambokidis *et al.* 2017\)](#), though the population size appears to have been stable since 2003, based on photo-ID studies (Calambokidis *et al.* 2014, IWC 2012). Total annual human-caused mortality of PCFG gray whales during the period ~~2008 to 2012~~ [to 2016](#) includes ~~deaths~~ [mortality and serious injuries](#) due to commercial fisheries (~~0.15~~ [0.7](#)/yr), [tribal fisheries \(0.15/yr\)](#), and ship strikes (~~0.1~~ [0.4](#)/yr), [plus unidentified whale entanglements assigned as PCFG gray whales \(0.1\)](#), or ~~0.25~~ [1.35](#) whales annually. This does not exceed the PBR level of ~~3.1~~ [3.5](#) whales for this population. Levels of human-caused mortality and serious injury resulting from commercial fisheries and ship strikes for both ENP and PCFG whales represent minimum estimates as recorded by stranding networks or at-sea sightings.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

Some gray whales observed feeding off Sakhalin and Kamchatka migrate during the winter to the west coast of North America in the eastern North Pacific (Mate *et al.* 2011; Weller *et al.* 2012; Urbán *et al.* 2013), while others, including at least one whale first identified as a calf off Sakhalin, migrate to areas off Asia in the WNP (Weller *et al.* 2008; Weller *et al.* 2013a). Despite the observed movements between the WNP and eastern North Pacific (ENP), genetic comparisons show significant mitochondrial and nuclear genetic differences between whales sampled in the ENP and those sampled on the feeding ground off Sakhalin Island in the WNP (LeDuc *et al.* 2002; Lang *et al.* 2011). While a few previously unidentified non-calves are identified annually, a recent population assessment using photo-identification data from 1994 to 2011 fitted to an individually-based model found that whales feeding off Sakhalin Island have been demographically self-contained, at least in recent years, as new recruitment to the population is almost exclusively a result of calves born to mothers from within the group (Cooke *et al.* 2013).

Historical evidence indicates that the coastal waters of eastern Russia, the Korean Peninsula and Japan were once part of the migratory route in the WNP and that areas in the South China Sea may have been used as wintering grounds (Weller *et al.* 2002; Weller *et al.* 2013a). However, contemporary records of gray whales off Asia are rare, with only 13 from Japanese waters between 1990 and 2007 (Nambu *et al.* 2010) and 24 from Chinese waters since 1933 (Wang 1984; Zhu 2002). The last known record of a gray whale off Korea was in 1977 (Park 1995; Kim *et al.* 2013). While recent observations of gray whales off the coast of Asia are infrequent, they nevertheless continue to occur, including: (1) March/April 2014—one or possibly two gray whales were sighted and photographed off the Shinano River in Teradomari (Niigata Prefecture) on the Sea of Japan coast of Honshu, Japan (Kato *et al.* 2014), (2) March 2012—a gray whale was sighted and photographed in Mikawa Bay (Aichi Prefecture), on the Pacific coast of Honshu, Japan (Kato *et al.* 2012), and (3) November 2011—a 13 m female gray whale was taken in fishing gear offshore of Baiqingxiang, China, in the Taiwan Strait (Zhu 2012).

Information from tagging, photo-identification and genetic studies show that some whales identified in the WNP off Russia have been observed in the [eastern North Pacific \(ENP\)](#), including coastal waters of Canada, the U.S. and Mexico (Lang 2010; Mate *et al.* 2011; Weller *et al.* 2012; Urbán *et al.* 2013; Mate *et al.* 2015). In combination,

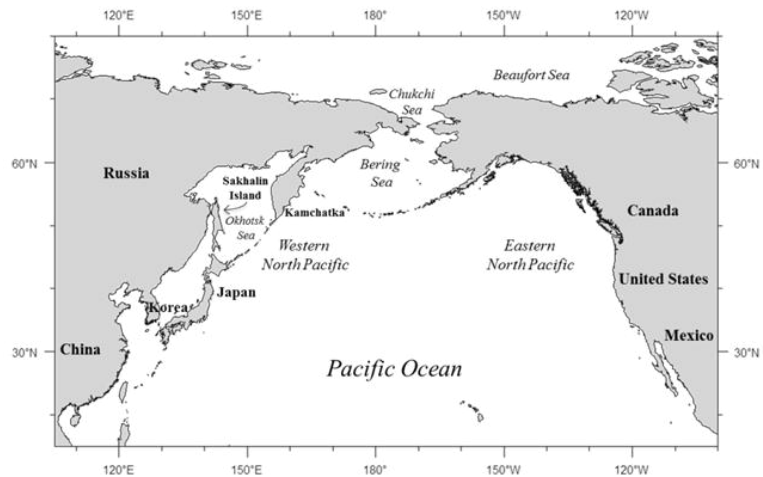


Figure 1. Range map of the Western North Pacific Stock of gray whales, including summering areas off Russia and wintering areas in the western and eastern Pacific.

these studies have recorded a total of ~~27~~ about 30 gray whales observed in both the WNP and ENP. Some whales that feed off Sakhalin Island in summer migrate east across the Pacific to the west coast of North America in winter, while others migrate south to waters off Japan and China (Weller *et al.* 2016). Despite the observed movements of some gray whales between the WNP and ENP, significant differences in their mitochondrial and nuclear DNA exist (LeDuc *et al.* 2002; Lang *et al.* 2011). Taken together, these observations indicate that not all gray whales in the WNP share a common wintering ground (Weller *et al.* 2013a).

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

POPULATION SIZE

Photo-identification data collected off Sakhalin Island between 1994 and ~~2011-2016~~ on the gray whale summer feeding ground off Sakhalin Island in the WNP were fitted to an individually-based population model (Cooke *et al.* 2016). Using the best fitting model, the aged 1+ (non-calf) population size was estimated to be 175 whales (Bayesian 95% CI 158-193) in 2016 (Cooke *et al.* 2016). used to calculate an abundance estimate of 140 (SE = ± 6, CV=0.043) whales for the age 1 plus (non-calf) population size in 2012 (Cooke *et al.* 2013). Some whales (approximately 70 individuals) sighted during the summer off southeastern Kamchatka have not been sighted off Sakhalin Island, but it is as yet unclear whether those whales are part of the WNP stock (IWC 2014).

Minimum Population Estimate

Although Cooke *et al.* (2016) did not report a coefficient of variation (CV) for their population size estimate, one can be approximated via simulation of a log-normal distribution, using their reported abundance and confidence limits. The estimated CV of the abundance estimate is 0.05, which is similar to previously reported estimates for this stock, using similar mark-recapture methods (Cooke *et al.* 2013). The minimum population estimate (N_{\min}) for the WNP stock is calculated from Equation 1 from the PBR Guidelines (Wade and Angliss 1997): $N_{\min} = N/\exp(0.842 \times [\ln(1 + [CV(N)]^2)]^{1/2})$ and the abundance estimate of 175 (CV=0.05) ~~140 (CV=0.043)~~ whales from Cooke *et al.* (2016), (2013), resulting in a minimum population estimate of 167 ~~135~~ gray whales on the summer feeding ground off Sakhalin Island in the WNP.

Current Population Trend

The Sakhalin Island population was estimated to be increasing from 2005 through 2015 at an average rate between 2-4% annually (Cooke *et al.* 2016). The WNP gray whale stock has increased over the last 10 years (2002-2012). The estimated realized average annual rate of population increase during this period is 3.3% per annum (± 0.5%) (Cooke *et al.* 2013).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (167) ~~(135)~~, times one-half the estimated maximum annual growth rate for a gray whale population ($\frac{1}{2}$ of 6.2% for the Eastern North Pacific Stock, Punt and Wade 2012), times a recovery factor of 0.1 (for an endangered stock with $N_{\min} < 1,500$, Taylor *et al.* 2003), and also multiplied by estimates for the proportion of the stock that uses U.S. EEZ waters (0.575) and the proportion of the year that those animals are in the U.S. EEZ (3 months, or 0.25 years) (Moore and Weller 2013), resulting in a PBR of 0.07 ~~0.06~~ WNP gray whales per year, or approximately 1 whale every 17-14 years (if abundance and other parameters in the PBR equation remained constant over that time period).

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

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

Today, a significant threat to gray whales in the WNP is ~~are~~ incidental catches in coastal net fisheries (Weller *et al.* 2002; ~~Kato~~ [Nakamura *et al.* 2017b](#); Weller *et al.* 2008; Weller *et al.* 2013a; [Burkanov *et al.* 2017](#)). Between 2005 and 2007, four female gray whales (including one mother-calf pair and one yearling) died in fishing nets on the Pacific coast of Japan. In addition, one adult female gray whale died as a result of a fisheries interaction in November 2011 off Pingtan County, China (~~Zhu 2012~~ [Wang *et al.* 2015](#)). An analysis of anthropogenic scarring of gray whales photographed off Sakhalin Island found that at least 18.7% (n=28) of 150 individuals identified between 1994 and 2005 had evidence of previous entanglements in fishing gear ~~but where the scars were acquired is unknown~~ (Bradford *et al.* 2009), ~~further highlighting the overall risks coastal fisheries pose to WNP gray whales.~~

~~In summer 2013, Trap nets for Pacific salmon net fishing was observed for the first time on the gray whale feeding ground off~~ [have been deployed in the feeding area off northeastern Sakhalin Island since 2013, resulting in two known entanglements and one probable entanglement mortality \(Burkanov *et al.* 2017\)](#). ~~Observations of whales within 100 m of salmon fishing nets have been made and a male gray whale was observed dragging fishing gear (rope), with a related injury on the caudal peduncle at the dorsal insertion point with the flukes (Weller *et al.* 2014).~~

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

Subsistence/Native Harvest Information

In 2005, the Makah Indian Tribe requested authorization from NOAA/NMFS, under the Marine Mammal Protection Act of 1972 (MMPA) and the Whaling Convention Act, to resume limited hunting of gray whales for ceremonial and subsistence purposes in the coastal portion of their usual and accustomed (U&A) fishing grounds off Washington State (NOAA ~~2008~~ [2015](#)). Observations of gray whales moving between the WNP and ENP highlight the need to estimate the probability of a gray whale observed in the WNP being taken during a hunt by the Makah Tribe (Moore and Weller 2013). Given conservation concerns for the WNP population, the Scientific Committee of the International Whaling Commission (IWC) emphasized the need to estimate the probability of a WNP gray whale being struck during aboriginal gray whale hunts (IWC 2012). Additionally, NOAA is required by the National Environmental Policy Act (NEPA) to prepare an Environmental Impact Statement (EIS) pertaining to the Makah’s request. The EIS needs to address the likelihood of a WNP whale being taken during the proposed Makah gray whale hunt.

To estimate the probability that a WNP whale might be taken during the proposed Makah gray whale hunt, four alternative models were evaluated. These models made different assumptions about the proportion of WNP whales that would be available for the hunt or utilized different types of data to inform the probability of a WNP whale being taken (Moore and Weller 2013). Based on the preferred model, the probability of striking at least one WNP whale in a single year was estimated to range from 0.006 – 0.012 across different scenarios for the annual number of total gray whales that might be struck. This corresponds to an expectation of ≥ 1 WNP whale strike in one of every 83 to 167 years. [This analysis was based on a 2012 abundance estimate of 155 \(95% CI 142-165\) which is slightly smaller than the 2016 abundance estimate of 175 \(95% CI 158-193\) reported by Cooke *et al.* \(2016\). It still represents the best estimate of WNP gray whale use of U.S. waters at this time.](#)

HABITAT ISSUES

Near shore industrialization and shipping congestion throughout the migratory corridors of the WNP gray whale stock represent risks by increasing the likelihood of exposure to pollutants and ship strikes as well as a general

degradation of the habitat. In addition, the summer feeding area off Sakhalin Island is a region rich with offshore oil and gas reserves. Two major offshore oil and gas projects now directly overlap or are in near proximity to this important feeding area, and more development is planned in other parts of the Okhotsk Sea that include the migratory routes of these whales. Operations of this nature have introduced new sources of underwater noise, including seismic surveys, increased shipping traffic, habitat modification, and risks associated with oil spills (Weller *et al.* 2002). During the past decade, a Western Gray Whale Advisory Panel, convened by the International Union for Conservation of Nature (IUCN), has been providing scientific advice on the matter of anthropogenic threats to gray whales in the WNP (see <http://www.iucn.org/wgwap/>). Ocean acidification could reduce the abundance of shell-forming organisms (Fabry *et al.* 2008, Hall-Spencer *et al.* 2008), many of which are important in the gray whales' diet (Nerini 1984).

STATUS OF STOCK

The WNP stock is listed as “Endangered” under the U.S. Endangered Species Act of 1973 (ESA) and is therefore also considered “strategic” and “depleted” under the MMPA. At the time the ENP stock was delisted, the WNP stock was thought to be geographically isolated from the ENP stock. Recent [Documentation of some whales moving between the WNP and ENP](#) seems to indicate otherwise (Lang 2010; Mate *et al.* 2011; Weller *et al.* 2012; Urbán *et al.* 2013). Other research findings, however, provide continued support for identifying two separate stocks of North Pacific gray whales, including: (1) significant mitochondrial and nuclear genetic differences between whales that feed in the WNP and those that feed in the ENP (LeDuc *et al.* 2002; Lang *et al.* 2011), (2) recruitment into the WNP stock is almost exclusively internal (Cooke *et al.* 2013), and (3) the abundance of the WNP stock remains low while the abundance of the ENP stock grew steadily following the end of commercial whaling (Cooke *et al.* 2013, 2017). As long as the WNP stock remains listed as endangered under the ESA, it will continue to be considered as depleted under the MMPA.

[In the past 5 years considerable effort has been undertaken to comprehensively assess the Pacific-wide stock structure of gray whales. For example, between 2014 and 2018 the International Whaling Commission \(IWC\) has convened five workshops on this matter. The objective of the workshops has been to develop a series of range-wide stock structure hypotheses, using all available data sources \(e.g. photo-id, genetics, tagging\), that can be tested within a modelling framework \(IWC 2017\). Completion of this work is scheduled for 2018-2019. Additionally, Cooke *et al.* \(2017\) conducted an updated assessment of gray whales in the WNP using an individually-based stage-structured population model with modified stock definitions that allows for the possibility of multiple feeding/breeding groups. Results from this work suggest that whales summering off Sakhalin Island and southeast Kamchatka, combined, appear to represent a genetically and demographically self-contained subpopulation that is characterized by preferential mating. In this scenario, whales identified feeding off Sakhalin represent about 2/3 of the combined Sakhalin Island-Kamchatka subpopulation. Further substructure within the subpopulation was not excluded by Cooke *et al.* \(2017\), including the possibility of less than 50 mature whales that breed only in the WNP. The IWC analysis is ongoing and the results of Cooke *et al.* \(2017\) are considered provisional pending further exploration of additional gray whale stock structure hypotheses.](#)

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

NMFS has conducted a global Status Review of humpback whales (Bettridge *et al.* 2015), and recently revised the ESA listing of the species (81 FR 62259, September 8, 2016 [NOAA 2016a](#)). NMFS is evaluating the stock structure of humpback whales under the MMPA, but no changes to current stock structure are presented at this time. However, effects of the ESA listing final rule on the status of the stock are discussed below.

Northern Hemisphere 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 currently 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, including the 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 data of individual whales (Calambokidis *et al.* 2001, Calambokidis *et al.* 2008). Photo-identification evidence also suggests strong site fidelity to feeding areas, but animals from multiple feeding areas converge on common winter breeding areas (Calambokidis *et al.* 2008). Baker *et al.* (2008) reported significant differences in mtDNA haplotype frequencies among different breeding and feeding areas in the North Pacific, reflecting strong matrilineal site fidelity to the 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.* 2008 [2013](#)). Among breeding areas, the greatest level of differentiation was found between Okinawa and Central America and most other breeding grounds (Baker *et al.* 2008 [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.* 2008 [2013](#)).

Along the U.S. west coast, [NMFS currently recognizes one humpback whale stock is currently recognized, including that includes two separate feeding groups](#): 1) a California and Oregon feeding group of whales that belong to the Central American and Mexican ~~distinct population segments (DPSs)~~ defined under the ESA (81 FR 62259, September 8, 2016 [NOAA 2016a](#)), and 2) a northern Washington and southern British Columbia feeding group that primarily includes whales from the Mexican DPS but also includes a small number of whales from the Hawaii and Central American DPSs (Calambokidis *et al.* 2008, Barlow *et al.* 2011, Wade *et al.* 2016). Very few photographic matches between these feeding groups have been documented (Calambokidis *et al.* 2008). [Calambokidis *et al.* \(2017a\) reported that approximately 70% of whales photographed in the southern Mexico and central America](#)

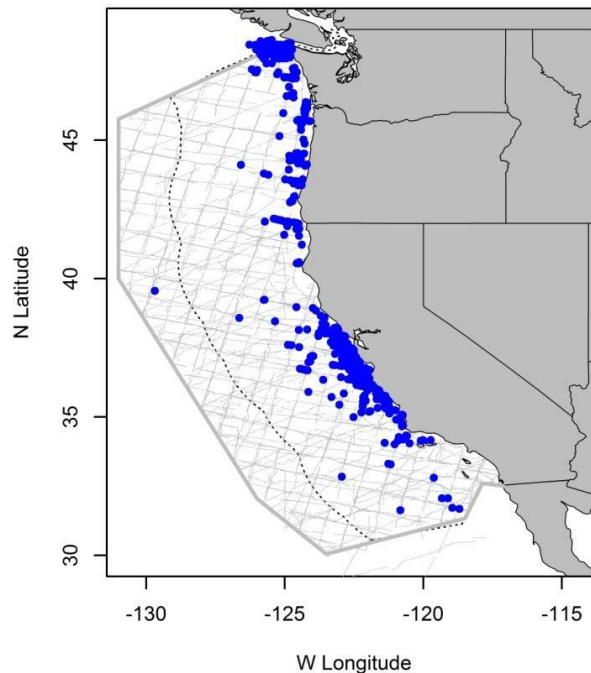


Figure 1. Humpback whale sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2014. Dashed line represents the U.S. EEZ, thin lines indicate completed transect effort of all surveys combined. See Appendix 2 for data sources and information on timing and location of survey effort.

[breeding ground regions 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.* 2017a).

For the Marine Mammal Protection Act (MMPA) stock assessment reports, the California/Oregon/Washington Stock is defined to include humpback whales that feed off the west coast of the United States, including animals from both the California-Oregon and Washington-southern British Columbia feeding groups (Calambokidis *et al.* 1996, Calambokidis *et al.* 2008, Barlow *et al.* 2011). Three other stocks are recognized in the U.S. MMPA Pacific stock assessment reports: the Central North Pacific Stock (with feeding areas from Southeast Alaska to the Alaska Peninsula), the Western North Pacific Stock (with feeding areas from the Aleutian Islands, the Bering Sea, and Russia), and the American Samoa Stock in the South Pacific (with largely undocumented feeding areas as far south as the Antarctic Peninsula).

POPULATION SIZE

Based on whaling statistics, the pre-1905 population of humpback whales in the North Pacific was estimated to be 15,000 (Rice 1978), but this population was reduced by whaling to approximately 1,200 by 1966 (Johnson and Wolman 1984). A photo-identification study in 2004-2006 estimated the abundance of humpback whales in the entire Pacific Basin to be 21,808 (CV=0.04) (Barlow *et al.* 2011). Barlow (2016) recently estimated 3,064 (CV= 0.82) humpback whales from a 2014 summer/fall ship line-transect survey of California, Oregon, and Washington waters.

Abundance estimates from photographic mark-recapture surveys conducted in California and Oregon waters every year from 1991 through 2011–2014 represent the most precise estimates (Calambokidis 2013 *et al.* 2017a). These estimates include only animals photographed in California and Oregon waters and not animals that are part of the separate feeding group found off Washington state and southern British Columbia (Calambokidis *et al.* 2009, 2017a). California and Oregon estimates range from approximately 1,100 1,400 to 2,600 2,400 animals, depending on the choice of recapture model and sampling period (Figure 2). The best estimate of abundance for California and Oregon waters is taken as the 2008–2011 2011–2014 Darroch–Chao estimate of 2,374 (CV=0.03) 1,729 (CV = 0.03) whales. [This estimate is considered the best of those reported by Calambokidis *et al.* 2017a because it accounts for individual capture heterogeneity](#), which is also the most precise estimate (Calambokidis *et al.* 2017a and Barlow 2013). This estimate includes virtually the entire Central American DPS, which was recently estimated to include 411 (CV=0.3) whales based on 2004-2006 photographic mark-recapture data (Wade *et al.* 2016). [However, the abundance estimate for the Central American DPS is ≥ 8 years old and is not considered a reliable estimate of current abundance \(NOAA 2016b\).](#)

Calambokidis *et al.* (2008 2017) reported a range of photographic mark-recapture abundance estimates (145–469) for ~~estimated~~ the northern Washington and southern British Columbia feeding group [population size](#) most recently ~~to be 526 (CV=0.23) animals~~ in 2005 [based on 2013 and 2014 mark-recapture data](#). The best model estimate from that paper (lowest AIC_c score) was reported as 189 (CV not reported) animals. This estimate is more than 8 years old and is outdated for use in stock assessments; however, because west coast humpback whale populations are growing (Calambokidis and Barlow 2013), this is still a valid minimum population estimate.

Combining abundance estimates from both the California/Oregon and Washington/southern British Columbia feeding groups (1,729 + 189 2,374 + 526) yields an estimate of 1,918 2,900 (CV ≈ 0.03) 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.](#) The approximate CV of 0.03 for the combined estimate reflects that a vast majority of the variance is derived from the California and Oregon estimate (CV=0.03) and that no CV was provided for the Washington state and southern British Columbia estimate.

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 1,876 2,784 animals.

Current Population Trend

Ship surveys provide some indication that humpback whales increased in abundance in California coastal waters between 1979/80 and 1991 (Barlow 1994) and between 1991 and 2014 (Barlow 2016), but this increase was

not steady, and estimates showed slight dips in 2001 and 2008. Mark-recapture population estimates had shown a long-term increase of approximately 8% per year (Calambokidis *et al.* 2009, Figure 2), but more recent estimates show variable trends a possible leveling-off of the population size (Figure 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 - 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 growth rate of the California/Oregon/Washington stock.

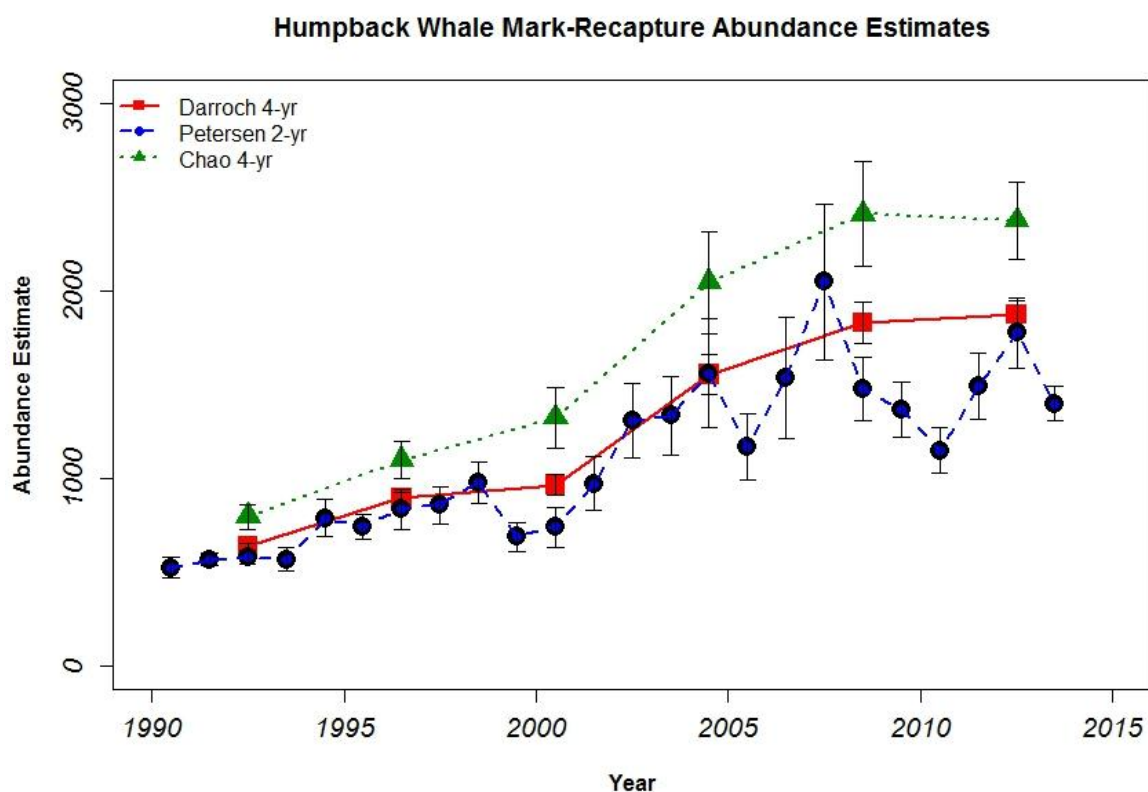


Figure 2. Mark-recapture estimates of humpback whale abundance in California and Oregon, 1991-2014, based on 3 different mark-recapture models and sampling periods (Calambokidis *et al.* 2017 and Barlow 2013). Vertical bars indicate ± 2 standard errors of each abundance estimate. Darroch and Chao models use 4-6 consecutive non-overlapping sample years. Estimates of humpback abundance in Washington and southern British Columbia waters are not shown, but the most-recent estimate is 526 (CV=0.23) whales for the 2-year period 2013-2014 (Calambokidis *et al.* 2017), except for the last estimates, which use the four most recent years, but overlap with the next-to-last estimate (Calambokidis and Barlow 2013).

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 in 1995-97 a greater proportion of calves were identified, and the 1997 reproductive rates for this population are closer to those reported for humpback whale populations in other regions (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). The current net productivity rate is unknown.

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (1,876 2,784) times one half the estimated population growth rate for this stock of humpback whales ($\frac{1}{2}$ of 8%) times a recovery factor of 0.3 (for an endangered species; see Status of Stock section below regarding ESA listing status with $N_{\min} > 1,500$ and $CV(N_{\min}) < 0.50$, Taylor *et al.* 2003), resulting in a PBR of 22 33.4. Because this stock spends approximately half its time outside the U.S. EEZ, the PBR allocation for U.S. waters is 11 16.7 whales per year.

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

A total of 71 123 human-related interactions involving humpback whales are summarized for the 5-year period 2011-2015 2012-2016 by Carretta *et al.* (2017a 2018a). These records include serious injuries, and non-serious injuries, and mortality involving pot/trap fisheries ($n=34$ 57), unidentified fishery interactions (26 49), vessel strikes (9 13), gillnet fisheries (1 3) and marine debris moorings (1 1). The number of serious injuries and mortalities for each category are summarized below. In addition to interactions with humpback whales, there were 19 21 entanglements and one vessel strike records of involving 'unidentified whales' (totaling 15 17 serious injuries and mortalities) during 2011-2015 2012-2016, some of which were certainly humpback whales (Carretta *et al.* 2018a, Carretta 2018). 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). The number of serious injuries of 'unidentified whales' during 2011-2015 was therefore, $15 / 5 = 3$ animals annually.

Fishery Information

Pot and trap fisheries fishery entanglements are the most commonly documented source of serious injury and mortality of humpback whales in U.S. west coast waters (Carretta *et al.* 2013, 2015, 2016a, 2018a), and entanglement reports have increased considerably since 2014. From 2011 to 2015 2012 to 2016, there were 34 57 documented observed interactions with pot and trap fisheries (Carretta *et al.* 2018a Carretta *et al.* 2017a, Jannot *et al.* 2016). One of the pot/trap records includes a prorated serious injury (0.75) in a recreational Dungeness crab pot, which is excluded from Table 1 commercial fishery totals and is detailed in the 'Other Mortality' section of this report. Twelve records (3 CA spot prawn pot + 8 Dungeness crab pot + 1 lobster pot) Eighteen records involved non-serious injuries resulting from human intervention to remove gear, or cases where animals were able to free themselves. Two records involved dead whales, including one humpback recovered in sablefish pot gear in offshore Oregon waters and one case where severed humpback flukes were found in southern California waters entangled in California Dungeness crab gear (Carretta *et al.* 2016, 2017a, 2018a). The remaining 20 36 pot/trap fishery injury cases, once evaluated per the NMFS serious injury policy, resulted in a total of 15.5 31.75 serious injuries / 5 years, or 3.1 6.4 humpback whales annually (Table 1). Documented 5-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 (16.75), unidentified pot/trap fishery (7.75), Washington/Oregon/California sablefish pot fishery (2.5), Washington Dungeness crab pot (0.75), California spot prawn (2.5), unknown commercial Dungeness crab pot fishery (0.75), and Oregon Dungeness crab pot (0.75) (Table 1). This includes 10.25 serious injuries (from 13 cases) in unidentified trap/pot fisheries, 2.25 serious injuries (from 3 cases) in California Dungeness crab pot, 1.5 serious injuries (from 2 cases) in the CA recreational Dungeness crab pot fishery, 0.75 serious injury (from 1 case) in a generic Dungeness crab pot fishery (state unknown), and 0.75 serious injury (from 1 case) in the CA spot prawn trap fishery. Including the 2 deaths attributed to pot/traps, the minimum level of annual mortality and serious injury across all pot/trap fisheries is 15.5 serious injuries + 2 mortalities = 17.5 whales / 5 years = 3.5 whales annually. Two records (totaling 1.5 serious injuries) are attributed to the recreational Dungeness crab fishery and thus, are not counted towards commercial fishery totals (but count against PBR, see Status of Stock Section). Thus, the number of commercial pot/trap fishery serious injuries and deaths totals 16 whales, or 16/5 = 3.2 whales annually (Table 1).

Table 1. Summary of available information on the observed and estimated incidental mortality and serious injury of humpback whales (California/Oregon/Washington stock) for commercial fisheries that are likely to take this species (Carretta *et al.* 2017a, Carretta *et al.* 2017b, 2018a, 2018b). All totals represent observed cases, except for the California swordfish and thresher shark drift gillnet fishery estimates shown in the first row. Mean annual takes are based on 2011-2015 2012-2016 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 (and serious injury)	Estimated mortality and serious injury (CV)	Mean Annual Takes (CV)
CA swordfish and thresher shark drift gillnet fishery	2011-2015 2012-2016	observer	24% 23%	0 ¹	0.1 (3) 0.2 (2.5)	< 0.02 (3) 0.04 (2.5)
CA halibut/white seabass and other species large mesh (≥3.5") set gillnet fishery	2010-2014 2012-2016	observer	9% <10%	0	0	0 (n/a)
CA spot prawn pot	2011-2015 2012-2016	Strandings / sightings	n/a	0 (0.75) 0 + 2.5	n/a	≥ 0.15 0.50 (n/a)
Unspecified pot or trap fisheries (includes generic 'Dungeness' crab gear not attributed to a specific state fishery)	2011-2015 2012-2016	Strandings / sightings	n/a	0 (11) 0 + 7.75	n/a	≥ 2.2 1.6 (n/a)
CA Dungeness crab pot	2011-2015 2012-2016	Strandings / sightings	n/a	1 (2.25) 1 + 15.75	n/a	≥ 0.65 3.4 (n/a)
OR Dungeness crab pot ²	2011-2015 2012-2016	Strandings / sightings	n/a	0 (0) 0 + 0.75	n/a	≥ 0 0.15 (n/a)
WA coastal-Dungeness crab pot	2011-2015 2012-2016	Strandings / sightings	n/a	0 (0) 0 + 0.75	n/a	≥ 0 0.15 (n/a)
WA/OR/CA limited entry sablefish pot	2014 2012-2016	observer	31% < 30%	1 (0) 1 + 1.5	n/a ³	≥ 0.2 0.50 (n/a)
unidentified fisheries (includes 'unidentified gillnet')	2011-2015 2012-2016	Strandings / sightings	n/a	3 (19) 3 + 35.75	n/a	≥ 4.4 7.75 (n/a)
Total Annual Takes						≥ 7.6 14.1 (n/a)

Gillnet (n=1 3) and unidentified fisheries (n=26 49) accounted for 27 52 interactions with humpback whales between 2011 and 2015 2012 and 2016 (Carretta *et al.* 2017a 2018a). 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 fisheries (Carretta *et al.* 2017a, 2018a). Three-Three records involved dead whales. The remaining 24 49 records, once evaluated per the NMFS serious injury policy, resulted in one non-serious injury four non-serious injuries and 19 35.75 serious injuries (16 cases x 0.75 = 12 prorated serious injuries, plus 7 non-prorated serious injuries). The total annual mortality and serious injury due to unidentified and gillnet fisheries from 2011 to 2015 2012 to 2016 sightings reports is 22 38.75 whales. The 5-year annual mean serious injury and mortality due to unidentified fisheries during this period is therefore 22 38.75/ 5 = 4.4 7.75 whales.

Three humpback whale entanglements (all released alive) were observed in the CA swordfish drift gillnet fishery from over 8,700 8,845 fishing sets monitored between 1990 and 2015 2016 (Carretta *et al.* 2017b 2018b). Some opportunistic sightings of free-swimming humpback whales entangled in gillnets may also originate from this fishery. The most recent model-based estimate of humpback whale bycatch in this fishery for 2011-2015 2012-2016 is 0.2 whales (CV=2.5) 0.4 whales (CV= 2.0), but it is estimated that only one-quarter of these entanglements represent serious injuries (Martin *et al.* 2015). The corresponding ratio estimate of bycatch for the same time period is zero (Carretta *et al.* 2017b 2018b). The model-based estimate is considered superior because it utilizes all 26-27 years of data for estimation, in contrast to the ratio estimate that uses only 2011-2015 2012-2016 data. The average

¹ There were no observations of humpback whales in this fishery during 2011-2015 2012-2016, but the model-based estimate of bycatch for this period results in a positive estimate of bycatch (Carretta *et al.* 2017b 2018b).

² There were 3 non-serious injuries involving humpback whales with this fishery from 2011-2015 2012-2016.

³ No estimate of total bycatch has been generated for this fishery.

annual estimated serious injury and mortality in the CA swordfish drift gillnet fishery is ~~0.02~~ 0.04 whales (~~0.1 whales~~ 0.2 whales / 5 years).

Unidentified whales represent approximately 15% of entanglement cases along the U.S. West Coast, (Carretta 2018). Observed entanglements may lack species IDs due to rough seas, distance from whales, or a lack of cetacean identification expertise. In previous 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) collectively found to be statistically-significant predictors of known-species entanglement cases (Carretta 2018). The species model was used to assign species ID probabilities for 21 unidentified whale entanglement cases ('novel data') during 2012-2016. The sum of species assignment probabilities for this 5-year period result in an additional 14.4 humpback whale entanglements for 2012-2016. 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 $14.4 \times 0.75 = 10.8$ additional humpback serious injuries are represented from the 21 unidentified whale entanglement cases during 2012-2016, or 2.2 humpback whales annually.

Total commercial fishery serious injury and mortality of humpback whales for the period ~~2011-2015~~ 2012-2016 is the sum of pot/trap fishery records (~~16~~ 31.75), plus unidentified fishery records (~~22~~ 38.75), plus estimates from the CA swordfish drift gillnet fishery (~~0.02~~ 0.2), or ~~38~~ 70.7 total whales. The mean annual serious injury and mortality from commercial fisheries during ~~2011-2015~~ 2012-2016 is ~~38~~ 70.7 whales / 5 years = ~~7.6~~ 14.1 whales (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.

Since 2015, NMFS has engaged in a multi-stakeholder process (including California State resource managers, fishermen, NGOs, and scientists) to identify/develop solutions and make recommendations to management and industry for reducing whale entanglements (see <http://www.opc.ca.gov/whale-entanglement-working-group/>). More recently, similar efforts to address the entanglement issue have also been initiated in Oregon and Washington. 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 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 ~~8~~ 14.75 additional serious injuries over the 5-year period ~~2011-2015~~ 2012-2016, or an additional ~~2.9~~ 2.9 humpback whales annually (Carretta *et al.* ~~2017a~~ 2018a).

Ship Strikes

~~Nine~~ Thirteen humpback whales (~~4~~ 8 deaths, ~~1.56~~ 2.6 serious injuries, and ~~3~~ 2 non-serious injuries) were reported struck by vessels between ~~2011 and 2015~~ 2012 and 2016 (Carretta *et al.* ~~2017a~~ 2018a). In addition, there was ~~one~~ one serious injury to an unidentified large whale from a ship strike during this time (Carretta *et al.* ~~2017a~~ 2018a). The observed average annual serious injury and mortality of humpback whales attributable to ship strikes during ~~2011-2015~~ 2012-2016 is ~~1.1~~ 2.1 whales per year (~~4~~ 8 deaths, plus ~~1.56~~ 2.6 serious injuries = ~~5.6~~ 10.6 per ~~5~~ years). Ship strike mortality was recently estimated for humpback whales in the California Current (Rockwood *et al.* 2017), using an encounter theory model (Martin *et al.* 2015), 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 in the region to estimate encounters that would result in mortality. The estimated number of annual ship strike deaths was 22 humpback whales, though this includes only the period July – November when whales are most likely to be present in the California Current and the time of year 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 ship strikes represents approximately 0.7% of the estimated population size of the stock (22 deaths / 2,900 whales). The results of Rockwood *et al.* (2017) also include a no-avoidance encounter model that results in a worst-case estimate of 48 humpback whale ship strike deaths per year, which represents 1.6% of the estimated population size. The number of vessel strikes attributable to each breeding ground DPS (Central America, Mexico) are 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 2012-2016 (2.6/yr) versus estimated vessel strikes (22/yr) indicates that the rate of detection for humpback whale vessel strikes is approximately 12%. The results of this study

were published while this report was being prepared and the results will be fully incorporated into the draft 2018 stock assessment report for this species.

Vessel traffic within the California Current represents a continued ship 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 ship traffic, and these factors may be influenced by economic and regulatory changes. For example, Moore *et al.* (2018) found that primary routes travelled by ships 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 to understand how variability in vessel traffic affects ship strike risk and mitigation strategies.

Other human-caused mortality and serious injury

A humpback whale was entangled in a research wave rider buoy in 2014. The whale is estimated to have been entangled for 3 weeks and had substantial necrotic tissue around the caudal peduncle. Although the whale was fully disentangled by a whale entanglement team, this animal was categorized as a serious injury⁴ 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.* 2016, 2017a). Additionally, one humpback whale was entangled in 2015 in recreational Dungeness crab pot gear, resulting in a prorated serious injury (0.75) (Carretta *et al.* 2018a). The total number of serious injuries from marine moorings sources (1) and recreational fisheries (0.75) for 2012-2016 is 1.75 whales, or 0.35 whales annually.

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, have been identified as 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 Endangered Species Conservation Act of 1969. This protection was transferred to the 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 estimated-observed annual mortality and serious injury due to commercial fishery entanglements in 2011-2015 2012-2016 (7.6 14.1/yr) (Table 1), non-fishery entanglements (0.2 0.2/yr), recreational crab pot fisheries (0.3 0.15/yr), serious injuries assigned to unidentified whale entanglements (2.2/yr) plus observed ship strikes (4.1 2.1/yr), equals 9.2 18.8 animals, which exceeds the PBR of 16.7 animals. Estimated vessel strike deaths are 22 humpback whales annually (Rockwood *et al.* 2017). The total observed + estimated annual human-caused mortality

⁴ This whale was initially listed as a non-serious injury in Carretta *et al.* (2016a) due to insufficient detail in the preliminary reporting. It is considered a serious injury for purposes of this stock assessment report.

of humpback whales is the sum of commercial fishery (14.1) + recreational fishery (0.15) + non-fishery entanglements (0.2/yr) + serious injuries assigned to unidentified whale entanglements (2.2/yr) + vessel strikes (22/yr) or 38.6 humpback whales annually. This exceeds the range-wide PBR estimate of 33.4 humpback whales. Although this is less than the stock's PBR (11) for U.S. waters, not all entangled or ship-struck whales are detected and the true rate of mortality and serious injury is almost certainly greater than 9.2. Other than the vessel strike estimates, Most—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 fraction of anthropogenic injuries and deaths to humpback whales that are undocumented on the U.S. west coast, but for vessel strikes, a comparison of observed vs. estimated annual vessel strikes suggests that approximately 12% of vessel strikes are documented. In addition to incidents involving humpback whales, an additional number of 'unidentified whales' (3/yr) (n=21) were seriously injured or killed between 2011–2015 2012–2016 (Carretta *et al.* 2018a). Prorating these unidentified entanglements to species results in an additional 10.8 humpback whale serious injuries / deaths over this period (Carretta 2018). some of which were certainly humpback whales, based on the observed proportion (40%) of all large whale injury cases identified as humpbacks during this period (Carretta *et al.* 2017a). Based on strandings and at sea observations, observed annual humpback whale mortality and serious injury in commercial fisheries (7.6 14.1/yr) is greater than 10% of the PBR; therefore, total fishery mortality and serious injury is not approaching zero mortality and serious injury rate. 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 have shown variable trends indicate a leveling-off of the population size (Calambokidis *et al.* 2017).

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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, and it has been [Stafford *et al.* 2001, Stafford 2003, and Monnahan *et al.* 2014](#) have proposed that these represent distinct populations with some degree of geographic overlap (Stafford *et al.* 2001, Stafford 2003, Monnahan *et al.* 2014). The northeastern call predominates in the Gulf of Alaska, [along](#) the U.S. West Coast, and [in](#) the eastern tropical Pacific, while the northwestern call predominates from south of the Aleutian Islands to the Kamchatka Peninsula in Russia, 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 their seasonal patterns (Stafford *et al.* 2001). Blue whales satellite-tagged off California in late summer have been [found to travel](#) to the eastern tropical Pacific and the Costa Rica Dome area in winter (Mate *et al.* 1999, Bailey *et al.* 2009). Photographs of blue whales in California have also 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 that [there has been a northward shift in blue whale distribution within the California Current, based on a series of vessel-based line-transect surveys from 1991-2014.](#)

Gilpatrick and Perryman (2008) showed 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~~ North Pacific.

For the Marine Mammal Protection Act (MMPA) stock assessment reports, the Eastern North Pacific Stock of blue whales includes animals found in the eastern North Pacific from the northern Gulf of Alaska to the eastern tropical Pacific. This definition is consistent with both the distribution of the northeastern call type, photogrammetric length determinations and with the known range of photographically identified individuals. Based on locations where the northeastern call type has been recorded, some individuals in this 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 certainly one of the most important feeding areas in summer and fall (Figure 1), but, increasingly, blue whales from this stock have been found feeding to the north and south of this area during summer and fall. Nine ‘biologically important areas’ (BIAs) for blue whale feeding are identified off the California coast by Calambokidis *et al.* (2015), including six 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, in the Gulf of California, and on the Costa Rica Dome. Given that these migratory destinations are areas of high productivity and given the observations of feeding in these areas, blue whales can be assumed to feed year round. Some individuals from this stock may be present year-round on the Costa Rica

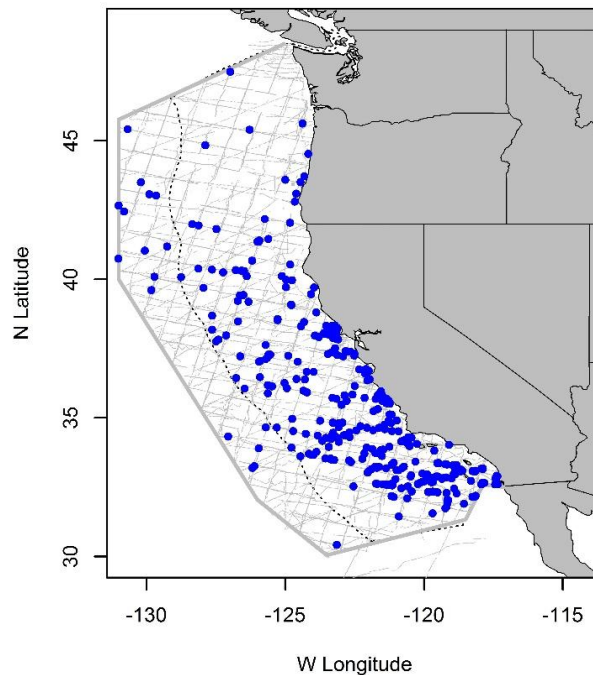


Figure 1. Blue whale sighting locations based on aerial and summer/autumn shipboard surveys off California, Oregon, and Washington, 1991-2014. Dashed line represents the U.S. EEZ; thin lines represent completed transect effort for all surveys combined.

Dome (Reilly and Thayer 1990). However, it is also possible that some southern hemisphere blue whales might occur north of the equator during the austral winter. One other stock of North Pacific blue whales (the Central North Pacific stock) is recognized in the Pacific Marine Mammal Protection Act (MMPA) Stock Assessment Reports.

POPULATION SIZE

The size of the feeding stock of blue whales off the U.S. West Coast has been estimated recently by both 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 and this variability has been 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. EEZ during summer and autumn line-transect surveys and estimates can be corrected for heterogeneity in sighting probabilities. Generally, the highest abundance estimates from line-transect surveys were obtained during the mid-1990s \(Figure 2\), when ocean conditions were colder than present-day. Since that time, line-transect abundance estimates within the California Current have declined, while estimates from mark-recapture studies have remained stable \(Figure 2\). Evidence for a northward shift in blue whale distribution includes increasing numbers of blue whales found in Oregon and Washington waters during a 1996-2014 line-transect surveys \(Barlow 2016\) and satellite tracks of blue whales in Gulf of Alaska and Canadian waters between 1994 and 2007 \(Bailey *et al.* 2009\). Line-transect abundance estimates from summer/autumn research vessel surveys in the California Current ranged between approximately 400 and 800 animals from 2001 to 2008 \(Barlow and Forney 2007, Barlow 2010\). These estimates are considerably lower than previous line-transect estimates of approximately 1,900 animals obtained between 1991 and 1996 \(Barlow 2010\) \(Figure 2\). \[An analysis of line-transect survey data from 1996-2014 provided a range of blue whale estimates from a high of approximately 2,900 whales in 1996 to a low of 900 whales in 2008 \\(Barlow 2016\\). The mean abundance estimate from the two most-recent line-transect surveys conducted in 2008 and 2014 is 1,146 \\(CV=0.33\\) whales. The lower abundance estimates appear to be related to a northward shift in the distribution of blue whales out of the study area \\(as far north as the Gulf of Alaska\\) and not a population decline \\(Barlow and Forney 2007, Calambokidis *et al.* 2009a\\). Mark-recapture estimates are often negatively biased by individual heterogeneity in sighting probabilities \\(Hammond 1986\\); however, Calambokidis *et al.* \\(2010\\) minimize such effects by selecting one sample that was taken randomly with respect to distance from the coast. Because some fraction of the population is always outside the survey area, the line-transect and mark recapture estimation methods provide different measures of abundance for this stock. Line transect estimates reflect the average density and abundance of blue whales in the study area during summer and autumn surveys, while mark recapture estimates can provide an estimate of total population size.\]\(#\)](#)

~~New photographic~~ [Photographic](#) mark-recapture estimates of abundance for the period 2005 to 2011 presented by Calambokidis and Barlow (2013) range from approximately 1,000 to 2,300 animals, with the most consistent estimates represented by a 4-yr sampling period Chao model that incorporates individual capture heterogeneity over time. The Chao model consistently yielded estimates of approximately 1,500 whales (Figure 2). The best estimate of blue whale abundance is taken from the Chao model results of Calambokidis and Barlow (2013) for the period 2008 to 2011, or 1,647 (CV=0.07) whales.

Minimum Population Estimate

The minimum population estimate for blue whales is taken as the lower 20th percentile of the log-normal distribution of abundance estimated from the mark-recapture estimate, or approximately 1,551.

Current Population Trend

Mark-recapture estimates provide the best indicator 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 Figure 2, there is no evidence of a population size increase in this blue whale population since the early 1990s. While the Petersen mark-recapture estimates show an apparent increase in blue whale abundance since 1996, the estimation errors associated with these estimates are also much higher than for the Chao estimates (Figure 2). 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 suggest that density dependence and not impacts from ship

strikes, explains the observed lack of a population size increase since the early 1990s. The authors [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.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Based on mark-recapture estimates from the US West Coast and Baja California, Mexico, Calambokidis *et al.* (2009b) estimated a rate of increase just under 3% per year, but [this estimate excludes the effects of anthropogenic mortality and serious injury on the population. Thus, the observed rate of population increase from mark-recapture estimates likely represents an underestimate of the maximum net productivity rate for this stock. 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 \(NMFS 2016\). it is not known if that corresponds to the maximum growth rate of this stock.](#)

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (1,551) [times one half the default maximum net growth rate for cetaceans \(½ of 4%\) times a recovery factor of 0.3](#) (for an endangered species which has a minimum abundance greater than 1,500 and a $CV_{Nmin} < 0.5$), resulting in a PBR of 9.3. Because whales in this stock spends approximately three quarters of their time outside the U.S. EEZ, the PBR allocation for U.S. waters is one-quarter of this total, or 2.3 whales per year.

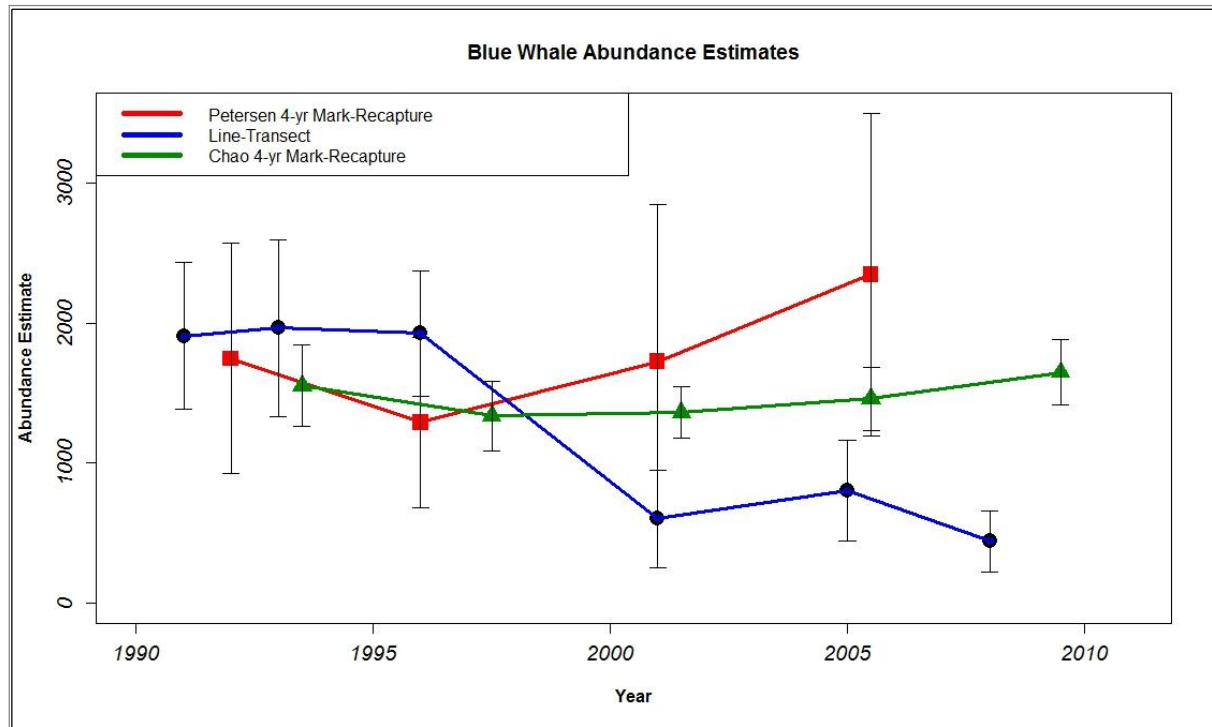


Figure 2. Estimates of blue whale abundance from line-transect and photographic mark-recapture surveys, 1991 to 2011 (Barlow and Forney 2007, Barlow 2010, Calambokidis and Barlow 2013). Vertical bars indicate ± 2 standard errors of each abundance estimate.

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 2012-2016 \(Carretta et al. 2018a\). Two additional prorated serious injuries were observed in unidentified fishing gear during the same period \(Table 1\). A seriously-injured blue whale was sighted entangled in unidentified pot/trap gear offshore of southern](#)

California in 2015, the first documented blue whale entanglement in a commercial fishery in this region (Carretta *et al.* 2017a). There have been no observed entanglements of blue whales in the California swordfish drift gillnet fishery during a 26~~27~~-year observer program that includes 8,711 8,845 observed fishing sets from 1990-2015 1990-2016 (Julian and Beeson 1998, Carretta *et al.* 2004, Carretta *et al.* 2017b 2018b). Some gillnet mortality of large whales may go unobserved because whales swim away with a portion of the net; however, fishermen report that large rorquals usually swim through nets without entangling and with very little damage to the nets. ~~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.~~ The total observed serious injury and mortality due to commercial fisheries during the period 2012-2016 is 4.5 whales, or 0.9 whales annually. This represents 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 IDs due to rough seas, distance from whales, or a lack of cetacean identification expertise in some cases. In previous 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) collectively found to be statistically-significant predictors of known-species entanglement cases (Carretta 2018). The species model was used to assign species ID probabilities for 21 unidentified whale entanglement cases ('novel data') during 2012-2016. Species probability assignments resulted in an additional 0.3 additional blue whale entanglements for 2012-2016, or 0.06 blue whales annually.

Annual entanglement rates of blue whales (observed) for the period of 2012-2016 is the sum of observed annual entanglements (0.9/yr), plus species probability assignments from unidentified whales (0.06/yr), or 0.96 blue whales annually.

Table 1. Summary of available information on ~~the-observed~~ incidental mortality and injury of blue whales (Eastern North Pacific stock) ~~for-from~~ commercial fisheries ~~that might take this species-~~ (Carretta *et al.* 2017a 2018a, 2017b 2018b). 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 (and serious injury)	Estimated mortality and/or serious injury (CV in parentheses)	Mean Annual Takes (CV in parentheses)
<u>CA Dungeness crab pot</u>	<u>2012-2016</u>	<u>Strandings and sightings</u>	<u>n/a</u>	<u>0 (2)</u>	<u>n/a</u>	<u>≥ 0.4</u>
Unidentified pot/trap fishery	<u>2011-2015</u> <u>2012-2016</u>	opportunistic reports <u>Strandings and sightings</u>	<u>n/a</u>	0 (1) <u>0 (1)</u>	1 (n/a) <u>n/a</u>	≥ 0.2 <u>0.2</u>
<u>Unidentified fishery</u>	<u>2012-2016</u>	<u>Strandings and sightings</u>	<u>n/a</u>	<u>0 (1.5)</u>	<u>n/a</u>	<u>≥ 0.3</u>
CA/OR thresher shark/swordfish drift gillnet fishery	<u>2011-2015</u> <u>2012-2016</u>	observer	24% <u>23%</u>	0	0	0 (n/a)
Total Annual Takes						0 (n/a) <u>≥ 0.9</u>

Ship Strikes

No ship strikes of blue whales were recorded in the most recent 5-year period, 2011-2015, but there was one ship strike serious injury of an unidentified large whale during this same period (Carretta *et al.* 2017a). Ship strikes were implicated in the deaths of four blue whales and the serious injury of a fifth whale between 2009 and 2013 (Carretta *et al.* 2015). Five deaths occurred in 2007, the highest number recorded for any year. The remaining four ship strike deaths occurred in 2009 (2) and 2010 (2). One blue whale ship strike death was observed during the most recent 5-year period of 2012-2016 (Carretta *et al.* 2018a), resulting in an observed annual average of 0.2 ship strike deaths. Observations of blue whale ship 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 ship strikes observed in a single year (2007) was 5 whales (Carretta *et al.* 2013). Over the 10-year period 2007-2016, 11 blue whale ship strikes were observed (Carretta *et al.* 2013, 2018a). In addition, 4 unidentified whales were also observed struck by ships during the same 10-year period. No methods have been developed to prorate the number of

unidentified whale ship strike cases to species, because 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 ship strikes have been in the southern California Bight, where large container ship ports overlap with seasonal blue whale distribution (Berman-Kowalewski *et al.* 2010). ~~Several blue whales have been photographed in California with large gashes in their dorsal surface that appear to be from ship strikes.~~ Including ship strike records identified to species and prorated serious injuries, blue whale mortality and injuries attributed to ship strikes in California waters was [one whale during 2012-2016 \(Carretta *et al.* 2018a\)](#) ~~zero during 2011-2015 (Carretta *et al.* 2017a)~~. NOAA previously implemented a mitigation plan that includes NOAA weather radio and U.S. Coast Guard advisory broadcasts to mariners entering the Santa Barbara Channel to be observant for whales, along with recommendations that mariners transit the channel at 10 knots or less. The Channel Islands National Marine Sanctuary also developed a blue whale/ship strike response plan, which involved weekly overflights to record whale locations. Documented ship strike deaths and serious injuries are derived from actual counts of whale carcasses [and at-sea sightings](#) and ~~should be~~ [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) stressed [that the number of observed ship strike deaths of blue whales in the California Current likely exceeds PBR.](#)

Ship strike mortality was recently estimated for blue whales in the California Current (Rockwood *et al.* 2017), using an encounter theory model (Martin *et al.* 2015), 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 results of this study were published while this report was being prepared and the results will be fully incorporated into the draft 2018 stock assessment report for this species.~~ [The estimated number of annual ship strike deaths was 18 blue whales, though this includes only the period July – November when whales are most likely to be present in the California Current and the time of year 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 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 ship strikes represents approximately 1% of the estimated population size of the stock \(18 deaths / 1,647 whales\). 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 ship strike deaths, which represents 2.4% of the estimated population size. Using the moderate level of avoidance model from Rockwood *et al.* \(2017\), estimated ship strike deaths of blue whales are 18 annually. A comparison of average annual ship strikes observed over the period 2012-2016 \(0.2/yr\) versus estimated ship strikes \(18/yr\) indicates that the rate of detection for blue whale vessel strikes is approximately 1%. Comparing the highest number of ship strikes observed in a single year \(5 in 2007\) with the estimated annual number \(18\) implies that ship strike detection rates have not exceeded 27% \(5/18\) in any single year.](#)

Impacts of ship 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, levels of ship strikes, and projected numbers of vessels using the region through 2050. The authors concluded (based on 10 ship strike deaths per year) that this stock was at 97% of carrying capacity in 2013 and that current ship strike levels do not pose a threat to the status of this stock. These authors also analyzed the status of the blue whale stock based on a ‘high case’ of annual ship 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 traffic within the California Current represents a continued ship 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 ship 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 to understand how variability in vessel traffic affects ship strike risk and mitigation strategies.](#)

Habitat Concerns

Increasing levels of anthropogenic sound in the world's oceans has been suggested to be 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). Behavioral responses were highly dependent upon the type of sound source 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. The authors stated that behavioral responses to such sounds are influenced by a complex interaction of behavioral state, environmental context, and prior exposure of individuals to such sound sources. One concern expressed by the authors is if blue whales did not habituate to such sounds near feeding areas that "repeated exposures could negatively impact individual feeding performance, body condition and ultimately fitness and potentially population health." Currently, no evidence indicates that such reduced population health exists, but such evidence would be difficult to differentiate from natural sources of reduced fitness or mortality in the population. 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

The reported take of North Pacific blue whales by commercial whalers totaled 9,500 between 1910 and 1965 (Ohsumi and Wada 1972). Approximately 3,000 of these were taken ~~from~~ along the west coast of North America from Baja California, Mexico to British Columbia, Canada (Tonnessen and Johnsen 1982; Rice 1992; Clapham *et al.* 1997; Rice 1974). Recently, Monnahan *et al.* (2014) estimated that 3,411 blue whales (95% range 2,593–4,114) were removed from the eastern North Pacific populations between 1905 and 1971. Blue whales in the North Pacific were given protected status by the IWC in 1966, but Doroshenko (2000) reported that a small number of blue whales were taken illegally by Soviet whalers after that date. As a result of commercial whaling, blue whales were listed as "endangered" under the Endangered Species Conservation Act of 1969. This protection was transferred to the 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 observed annual incidental mortality and injury rate (-0.2/year) from ship strikes from 2011-2015 2012-2016 is less than the calculated PBR (2.3) for this stock, but this rate does not include unidentified large whales struck by vessels, some of which may have been blue whales, nor does it include undetected and unreported ship strikes of blue whales. Estimated vessel strike mortality (18/yr) exceeds the PBR of 2.3 for this stock of blue whales. While Redfern *et al.* (2013) noted that the number of blue whales struck by ships in the California Current likely exceeds the PBR for this stock, Monnahan *et al.* (2015) proposed that observed-estimated ship strike levels of 10 – 35 whales annually did ~~do~~ 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 strikes: 97% of K with 10 annual strikes and 91% of K with 35 annual strikes. The highest estimates of blue whale ship 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. The current annual Observed and assigned levels of serious injury and mortality due to commercial fisheries (≥ 0.2 0.96) for this stock is are not less than 10% of the stock's PBR (2.3), and thus, commercial fishery take levels are not is approaching zero mortality and serious injury rate.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

Northern Hemisphere fin whales (*B. physalus physalus*) likely comprise distinct Pacific and Atlantic subspecies (Archer *et al.* 2013). Mizroch *et al.* (2009) described eastern and western North Pacific populations, based on a review of sightings data, catch statistics, recaptures of marked whales, blood chemistry data, and acoustics. The two populations are thought to have separate wintering and mating grounds off of 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-migratory populations exist in the Gulf of California (Tershy *et al.* 1993; Bérubé *et al.* 2002) and the East China Sea (Fujino 1960). Evidence of additional subpopulations near Sanriku-Hokkaido and the Sea of Japan exists, based on seasonal catch data and recaptures of marked animals (Mizroch *et al.* 2009). Fin whales occur throughout the North Pacific, from the southern Chukchi Sea to the Tropic of Cancer (Mizroch *et al.* 2009), but their wintering areas are poorly known. Fin whales are scarce in the eastern tropical Pacific in summer (Wade and Gerrodette 1993) and winter (Lee 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); and Oregon and Washington (Moore *et al.* 1998). Fin whales satellite-tagged in the Southern California Bight (SCB) appear to 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, with the whale moving 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.

Insufficient information exists to determine population structure, but from a conservation perspective it may be risky to assume panmixia in the entire North Pacific. This report covers the stock of fin whales found along the coasts of California, Oregon, and Washington. 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. Fin whales are present year-round in southern California waters, as evidenced by individually-identified whales photographed in all four seasons (Falcone and Schorr 2013). 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.

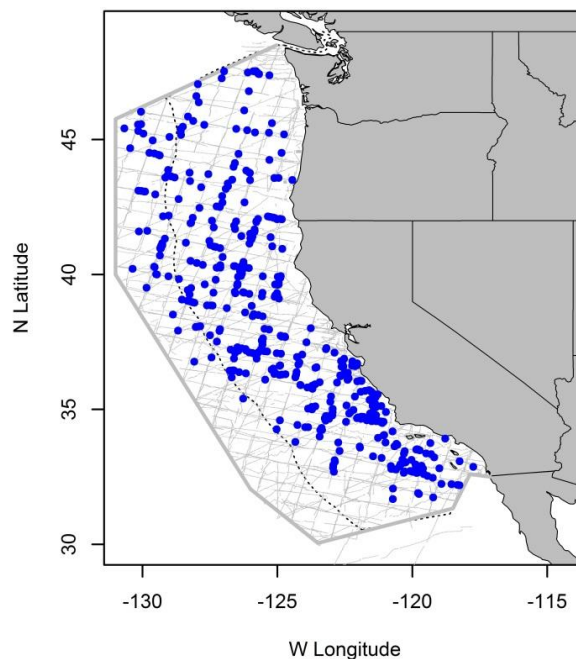


Figure 1. Fin whale sighting locations based on shipboard surveys off California, Oregon, and Washington, 1991-2014. Dashed line represents the U.S. EEZ; thin lines indicate completed transect effort of all surveys combined.

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. The best estimate of fin whale abundance in California, Oregon, and Washington waters out to 300 nmi is from a trend-model analysis of line-transect data from 1991 through 2014 (Nadeem *et al.* 2016; Fig. 2), which generated an estimate for 2014 of 9,029 (CV=0.12) whales. The new estimates are based on similar methods to those first applied to this population by Moore and Barlow (2011). However, the new abundance estimates are substantially 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 abundance estimated for 2014, or approximately 8,127 whales.

Current Population Trend

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 based on analysis of abundance data from line transect surveys conducted in the California Current between 1991 and 2014 (Nadeem *et al.* 2016, Figure 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 dispersal of animals into this area. Across the entire study area (waters off California, Oregon, and Washington), the mean annual abundance increase was 7.5%, although abundance appeared stable between 2008 and 2014. In all, there has been a roughly 5-fold 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 abundance trend for fin whales in Alaskan waters at a rate of 4.8% per year between 2001 and 2003.

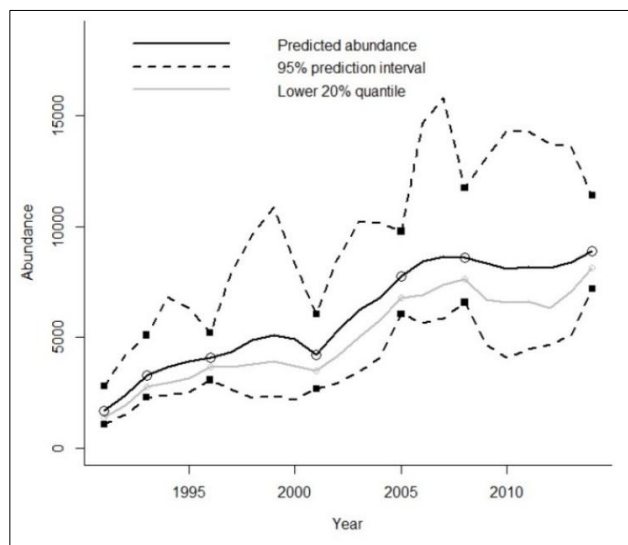


Figure 2. Trend-based estimates of fin whale abundance, 1991- 2014, with 95% Bayesian credible intervals (Nadeem *et al.* 2016).

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 (e.g., 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) times one half the default maximum net growth rate for cetaceans ($\frac{1}{2}$ of 4%) times a recovery factor of 0.5 (for an endangered species, with $N_{\min} > 5,000$ and $CV_{N_{\min}} < 0.50$, Taylor *et al.* 2003), resulting in a PBR of 81 whales.

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Fisheries Information

One fin whale death (in 1999) was observed in the California swordfish drift gillnet fishery from over 8,600 ~~8,845~~ observed sets between 1990 and 2014-2016 (Carretta *et al.* 2016a ~~2018a~~). Although no fin whales have been observed taken in the fishery since 1999, new model-based bycatch estimates include a very small estimate of 0.1 whales ($CV=3.7$) for the most recent 5-year period, 2010-2014 ~~2012-2016~~ (Carretta *et al.* 2016a ~~2018b~~). The large CV of this bycatch estimate is a consequence of the mean estimate being very small. This estimate is based on inclusion of 25-26 years of observer data spanning 1990-2014 ~~2016~~ and reflects a very low long-term observed bycatch rate scaled up to levels of unobserved fishing effort. Mean annual takes (<0.1) for this fishery (Table 1) are based on 2010-2014 ~~2012-2016~~ data. Some gillnet mortality of large whales may go unobserved because whales swim away with a portion of the net. One fin whale sighted at-sea was determined to be seriously injured (line cutting into the whale) as a result of interactions with unknown fishing gear during 2010-2014 ~~2012-2016~~ (Carretta *et al.* 2016b ~~2018b~~). Including systematic fishery observations in the CA swordfish drift gillnet fishery and opportunistic sightings of fishery-related injuries, the mean annual serious injury and mortality of fin whales for 2010-2014 ~~2012-2016~~ is ≥ 0.2 ~~0.5~~ whales (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.

Unidentified whales represent approximately 15% of entanglement cases along the U.S. West Coast, (Carretta 2018). Observed entanglements may lack species IDs due to rough seas, distance from whales, or a lack of cetacean identification expertise. In previous 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) collectively found to be statistically-significant predictors of known-species entanglement cases (Carretta 2018). The species model was used to assign species ID probabilities for 21 unidentified whale entanglement cases ('novel data') during 2012-2016. The sum of species assignment probabilities for this 5-year period result in an additional 0.26 fin whale entanglements for 2012-2016. 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 $0.26 \times 0.75 = 0.2$ additional fin whale serious injuries are represented from the 21 unidentified whale entanglement cases during 2012-2016. This represents a negligible annual estimate of 0.04 fin whales derived from sightings of unidentified entangled whales.

Table 1. Summary of available information on the incidental mortality and injury of fin whales (CA/OR/WA stock) for commercial fisheries that might take this species. The mean annual take estimate for unidentified fishery interactions includes negligible estimates of entanglements from unidentified whale entanglements (Carretta 2018).

Fishery Name	Data Type	Year(s)	Percent Observer Coverage	Observed (or self-reported)	Estimated Mortality (and serious injury)	Mean Annual Takes (CV in parentheses)
CA swordfish and thresher shark drift gillnet fishery	2010-2014 <u>2012-2016</u>	observer	22% 23%	0 ¹	0.1 (CV=3) <u>≥ 0.1 (CV=3.7)</u>	<0.1 (CV=3) <u><0.1 (CV=3.7)</u>
Unidentified fishery interactions	2010-2014 <u>2012-2016</u>	at-sea sightings	n/a	±2	0 (±2)	≥ 0.2 <u>0.4</u>

¹ There were no observations of fin whale entanglements in this fishery during 2010-2014, but the model-based estimate of bycatch for this period results in a positive estimate of bycatch (Carretta *et al.* 2016a).

Fishery Name	Data Type	Year(s)	Percent Observer Coverage	Observed (or self-reported)	Estimated Mortality (and serious injury)	Mean Annual Takes (CV in parentheses)
Minimum total annual takes						≥ 0.2 (CV=3) ≥ 0.5 (n/a)

Ship Strikes

Ship strikes were implicated in the deaths of ~~nine~~ 8 fin whales during 2010–2014 2012–2016 (Carretta *et al.* ~~2018b~~ 2015, Carretta *et al.* 2016b). During 2010–2014 2012–2016, there was one additional serious injury to an unidentified large whale attributed to a ship strike. 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. The average observed annual mortality and serious injury due to ship strikes is ~~1.8~~ 1.6 fin whales per year during 2010–2014 2012–2016. Documented ship strike deaths and serious injuries are derived from actual counts of whale carcasses and should be considered minimum values. Where evaluated, estimates of detection rates of cetacean carcasses are consistently quite low across different regions and species (<1% to 33%), highlighting that observed numbers underestimate true impacts (Carretta *et al.* 2016e, Kraus *et al.* 2005, Williams *et al.* 2011, Prado *et al.* 2013, Wells *et al.* 2015). Ship strike mortality was recently estimated for fin whales in the California Current (Rockwood *et al.* 2017), using an encounter theory model (Martin *et al.* 2015) 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 in the region to estimate encounters that would result in mortality. The estimated number of annual ship strike deaths was 43 fin whales, though this includes only the period July – November when whales are most likely to be present in the California Current and the time of year that overlaps with cetacean habitat models generated from line-transect surveys conducted during those months (Becker *et al.* 2016, Rockwood *et al.* 2017). This estimate was 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 ship strikes represents approximately < 0.5% of the estimated population size of the stock (43 deaths / 9,029 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 ship strike deaths per year, representing approximately 1% of the estimated population size. The authors also note that 65% of fin whale ship 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 ship 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 of fin whales killed or seriously injured by vessels is by directly comparing the documented number of ship strike deaths and serious injuries with annual estimates by Rockwood *et al.* (2017). Comprehensive coast-wide data on ship strike deaths and serious injuries assumed to result in death are compiled in annual reports on observed anthropogenic mortality for the 10-year period 2007 – 2016 (Carretta *et al.* 2013, 2018b). During this 10-year period, there were 15 observations of ship strike deaths and 1 serious injury assumed to result in the death of the whale, or 1.6 whales per year. The most conservative estimate of ship strike deaths from Rockwood *et al.* (2017) is 43 whales annually. The ratio of documented ship strike deaths (1.6/yr) to estimated annual deaths (43) implies a carcass recovery/documentation rate of 3.7%, which is lower than the worst-case estimate of 5% from Rockwood *et al.* (2017). There is uncertainty regarding the estimated number of ship strike deaths, however, it is apparent that carcass recovery rates of fin whales are quite low.

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 the California to Washington stock is automatically considered as a "depleted" and "strategic" stock under the MMPA. The total documented incidental mortality and serious injury (~~2.0~~ 2.1/yr) due to fisheries (~~0.2~~ 0.5/yr) and ship strikes (~~1.8~~ 1.6/yr) is less than the calculated PBR (81). Total fishery mortality is less than 10% of PBR and, therefore, may be approaching zero mortality and serious injury rate. Estimated vessel strike mortality

[in the population ranges between 43 and 95 whales annually, or 0.5 to 1% of the total estimated population size. These estimates of ship strike deaths are corrected for undocumented and undetected cases, as they are model-derived.](#) There is strong evidence that the population has increased since the early 1990s (Moore and Barlow 2011, Nadeem *et al.* 2016). Increasing levels of anthropogenic sound in the world's oceans has been suggested to be a habitat concern for whales, particularly for baleen whales that may communicate using low-frequency sound (Croll *et al.* 2002). Behavioral changes associated with exposure to simulated mid-frequency sonar, including no change in behavior, cessation of feeding, increased swimming speeds, and movement away from simulated sound sources has been documented in tagged *blue* whales (Goldbogen *et al.* 2013), but it is unknown if fin whales respond in the same manner to such sounds.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

The International Whaling Commission (IWC) only considers [recognizes](#) one stock of sei whales in the North Pacific (Donovan 1991, [Wada and Numachi 1991](#)), but some evidence exists for multiple populations (Masaki 1977; Mizroch *et al.* 1984; Horwood 1987). [Kanda *et al.* \(2006\)](#) reported that there is likely a single population of sei whales in the western North Pacific, based on [microsatellite analyses, for the region 37°N-45°N and 147°E-166°E](#). Sei whales are distributed far out to sea in temperate regions of the world and do not appear to be associated with coastal features. Whaling effort for this species was distributed continuously across the North Pacific between 45-55°N (Masaki 1977). Two sei whales that were tagged off California were later killed off Washington and British Columbia (Rice 1974) and the movement of tagged animals has been noted in many other regions of the North Pacific. Sei whales are rare in the California Current (Dohl *et al.* 1983; Barlow [1997](#)[2016](#); Forney *et al.* 1995; [Green *et al.* 1992](#) Mangels and Gerrodette 1994, Barlow 2016), but were the fourth most common whale taken by California coastal whalers in the 1950s-1960s (Rice 1974). They are extremely rare south of California (Wade and Gerrodette 1993; Lee 1993). Lacking additional information on sei whale population structure, sei whales in the eastern North Pacific (east of longitude 180°) are considered as a separate stock. [For the Marine Mammal Protection Act \(MMPA\) stock assessment reports, sei whales within the Pacific U.S. EEZ are divided into two discrete areas: 1\) California, Oregon and Washington waters \(this report\) and 2\) waters around Hawaii. The Eastern North Pacific stock includes animals found within the U.S. west coast EEZ and in adjacent high seas waters; however, because comprehensive data on abundance, distribution, and human-caused impacts are largely-unknown for high seas regions, the status of this stock is evaluated based on data from U.S. EEZ waters of the California Current \(NMFS 2005\).](#)

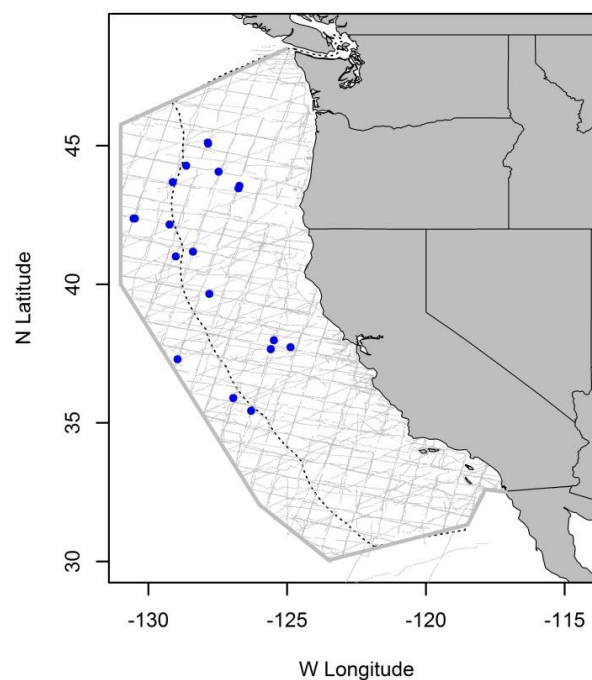


Figure 1. Sei whale sighting locations from shipboard surveys off California, Oregon, and Washington, 1991-2014. Dashed line represents the U.S. EEZ; thin lines indicate completed transect effort of all surveys combined.

POPULATION SIZE

Ohsumi and Wada (1974) estimated the pre-whaling abundance of sei whales to be 58,000-62,000 in the North Pacific. Later, Tillman (1977) used a variety of different methods to estimate the abundance of sei whales [abundance](#) in the North Pacific and revised this pre-whaling estimate to 42,000. His estimates for the year 1974 ranged from 7,260 to 12,620. ~~All [These previous studies](#) methods depended on using the history of catches and trends in CPUE or sighting rates; there have been no direct estimates of sei whale abundance in the entire (or eastern) North Pacific based on sighting surveys.~~ [Hakamada *et al.* \(2017\)](#) recently estimated sei whale abundance in the central and eastern North Pacific based on visual line-transect surveys from data collected between 2010 and 2012. The new estimate of 29,632 sei whales (CV =

0.242, 95% CI 18,576–47,267) represents the first systematic sighting survey abundance estimate for this species over a pelagic high-seas region. However, while the study area of Hakamada *et al.* (2007) included waters north of 40°N latitude and west of 135°W longitude, it excluded waters of the California Current. The estimated number of sei whales in the California Current Sei whale sightings in California, Oregon, and Washington waters during is based on extensive ship and aerial line-transect surveys between 1991-2014, where sightings have been relatively rare (Figure 1, Hill and Barlow 1992; Carretta and Forney 1993; Mangels and Gerrodette 1994; VonSaunders and Barlow 1999; Barlow 2003; Forney 2007; Barlow 2010; Barlow 2016). Green *et al.* (1992) did not report any sightings of sei whales in aerial surveys of Oregon and Washington. Abundance estimates for the two most recent line transect surveys of California, Oregon, and Washington waters in 2008 and 2014 out to 300 nmi are 311 (CV=0.76) and 864 (CV=0.40) sei whales, respectively (Barlow 2016). The best estimate of abundance for California, Oregon, and Washington waters out to 300 nmi is the unweighted geometric mean of the 2008 and 2014 estimates, or 519 (CV=0.40) sei whales (Barlow 2016).

Minimum Population Estimate

The minimum population estimate for sei whales is taken as the lower 20th percentile of the log-normal distribution of abundance estimated from 2008 and 2014 shipboard line-transect surveys, or approximately 374 whales.

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (374) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.1 (for an endangered species), resulting in a PBR of 0.75 whales.

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Fishery Information

The California swordfish drift gillnet fishery is the only fishery that is likely to take sei whales from this stock, but no fishery mortality or serious injuries have been observed from over 8,600 8,845 monitored fishing sets from 1990-2014-2016 (Carretta *et al.* 2017 2018a, Table 1). Mean annual takes for this fishery (Table 1) are based on 2010-2014 2012-2016 data. This results in an average estimate of zero sei whales taken annually. However, some gillnet mortality of large whales may go unobserved because whales swim away with a portion of the net.

Table 1. Summary of available information on the incidental mortality and injury of sei whales (eastern North Pacific stock) for commercial fisheries that might take this species. n/a indicates that data are not available. Mean annual takes are based on 2010-2014 2012-2016 data unless noted otherwise.

Fishery Name	Year(s)	Data Type	Percent Observer Coverage	Observed mortality (and injury in parentheses)	Estimated mortality (CV in parentheses)	Mean annual takes (CV in parentheses)
CA/OR thresher shark/swordfish drift gillnet fishery	2010-2014 2012 2013 2014 2015 2016	observer	22% 19% 37% 24% 20% 18%	0	0	0 (n/a)

Ship Strikes

There have ~~has~~ been ~~no~~ one documented ship strikes of a sei whales in the most recent 5-year period, ~~2010-2014~~ 2012-2016 (Carretta *et al.* ~~2016~~ 2018b), although some uncertainty exists over whether the strike occurred pre- or post-mortem. For purposes of this stock assessment report, the ship strike is considered as the probable cause of death. ~~although one ship strike death was reported in Washington in 2003 (NMFS Northwest Regional Office, unpublished data).~~ During ~~2010-2014~~ 2012-2016, there were ~~an~~ was one additional ~~eight injuries~~ serious injury of an unidentified large whales attributed to a ship strikes. 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. The average observed annual mortality due to ship strikes is ~~zero~~ 0.2 sei whales per year for the period ~~2010-2014~~ 2012-2016.

STATUS OF STOCK

The NMFS recovery plan for the sei whale (NMFS 2011) notes that basic information such as distribution, abundance, trends and stock structure is of poor quality or largely unknown, owing to the rarity of sightings of this species. Sei whales were estimated to have been reduced to 20% (8,600 out of 42,000) of their pre-whaling abundance in the North Pacific (Tillman 1977). The initial abundance has never been reported separately for the eastern North Pacific stock, but this stock was also probably depleted by whaling. Kanda *et al.* (2006) found a high level of genetic variation among sei whale samples in the western North Pacific and hypothesized that the population did not suffer from a genetic bottleneck due to commercial whaling. Sei whales are formally listed as "endangered" under the Endangered Species Act (ESA), and consequently the eastern North Pacific stock is automatically considered as a "depleted" and "strategic" stock under the Marine Mammal Protection Act (MMPA). Total known estimated fishery mortality is zero and therefore is approaching zero mortality and serious injury rate. ~~Although the~~ The current known rate of ship strike deaths and serious injuries is ~~zero~~ 0.2 annually, but it is likely that some sei whale ship strikes are unreported. 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 sei whales respond in the same manner to such sounds.

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SPINNER DOLPHIN (*Stenella longirostris longirostris*): Hawaiian Islands Stock Complex- Hawaii Island, Oahu/4-islands, Kauai/Niihau, Pearl & Hermes Reef, Midway Atoll/Kure, Hawaii Pelagic

STOCK DEFINITION AND GEOGRAPHIC RANGE

Six morphotypes within four subspecies of spinner dolphins have been described worldwide in tropical and warm-temperate waters (Perrin *et al.* 2009). The Gray's (or pantropical) spinner dolphin (*Stenella longirostris longirostris*) is the most widely distributed subspecies and is found in the Atlantic, Indian, central and western Pacific Oceans (Perrin *et al.* 1991). [Spinner dolphins in Hawaii belong to this subspecies. Unlike Gray's spinner dolphins in the eastern tropical Pacific \(ETP\), which are commonly found in pelagic waters. Within the central and western Pacific, spinner dolphins in Hawaii are island-associated and use shallow protected bays to rest and socialize during the day then move offshore at night to feed \(Norris and Dohl 1980; Norris *et al.* 1994\). Spinner dolphins in Hawaii are considered separate stocks from those in the ETP \(Perrin 1975; Dizon *et al.* 1994\). Andrews *et al.* \(2010\) found that mtDNA control region haplotype and nucleotide diversities of Hawaiian spinner dolphins are low compared with those from other geographic regions and suggested the existence of strong barriers to gene flow, both geographic and ecological. Her analyses also reveal significant genetic distinction, at both mtDNA and microsatellite loci, between spinner dolphins sampled in American Sāmoa and those sampled in the Hawaiian Islands \(Johnston *et al.* 2008, Andrews *et al.* 2010\).](#)

[Most research on spinner dolphins in Hawaii has occurred in nearshore waters surrounding the main Hawaiian Islands and at Midway and Kure Atoll in the northwestern Hawaiian Islands \(e.g. Norris *et al.* 1994, Karczmarski *et al.* 2005, Tyne *et al.* 2017\). Spinner dolphins have rarely been encountered in pelagic waters during large-scale line-transect surveys of the Hawaiian Archipelago. Summer/fall shipboard surveys of the waters within the U.S. Exclusive Economic Zone \(EEZ\) of the Hawaiian Islands, resulted in 8 sightings in 2002 and 2 sightings in 2010, though none of the 2010](#)

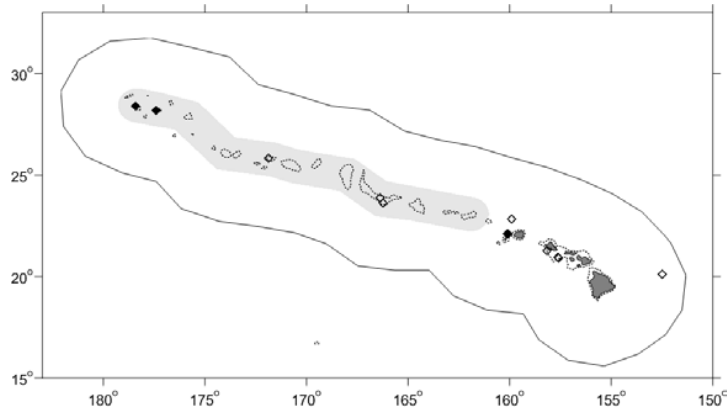


Figure 1. Spinner dolphin sighting locations during the 2002 (open diamonds) and 2010 (black diamonds) shipboard cetacean surveys of U.S. EEZ waters surrounding the Hawaiian Islands (Barlow 2006, Bradford *et al.* 2013; see Appendix 2 for details on timing and location of survey effort). Outer line indicates approximate boundary of survey area and U.S. EEZ. Gray shading indicates area of Papahānaumokuākea Marine National Monument. Dotted line represents the 1000 m isobath. Insular stock boundaries are shown in Figure 2.

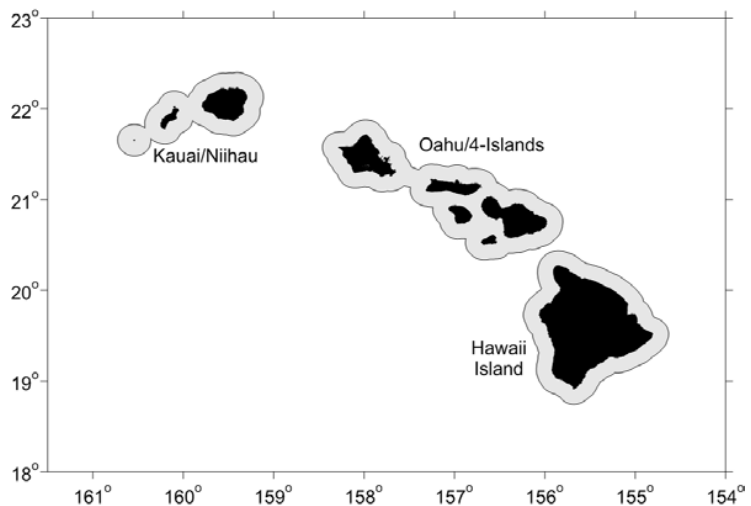


Figure 2. Spinner dolphin stock boundaries in the main Hawaiian Islands (Midway/Kure and Pearl and Hermes stock ranges not pictured). Animals outside of the defined island areas are considered to be part of the Hawaii pelagic stock.

sightings occurred during on-effort survey (Barlow 2006, Bradford *et al.* 2013; Figure 1).

Hawaiian spinner dolphins belong to a stock that is separate from animals in the eastern tropical Pacific (Perrin 1975; Dizon *et al.* 1994). The Hawaiian form is referable to the subspecies *S. longirostris longirostris*, which occurs pantropically (Perrin 1990). Andrews *et al.* (2010) found that mtDNA control region haplotype and nucleotide diversities of Hawaiian spinner dolphins are low compared with those from other geographic regions and suggested the existence of strong barriers to gene flow, both geographic and ecological. Her analyses also reveal significant genetic distinction, at both mtDNA and microsatellite loci, between spinner dolphins sampled in American Samoa and those sampled in the Hawaiian Islands (Johnston *et al.* 2008, Andrews *et al.* 2010). [The population structure of spinner dolphins in Hawaii has been assessed using genetic and movement data.](#) Andrews *et al.* (2010) also found significant genetic distinctions between spinner dolphins sampled at different islands within the Hawaiian Archipelago. Most significant was differentiation between animals sampled off the Kona Coast of Hawaii Island and animals sampled at all other Hawaiian Islands. Similarly, in the Northwestern Hawaiian Islands, spinner dolphins sampled at Midway and Kure were shown [are](#) not to be genetically distinct from each other, but are distinct from those sampled at all other islands. Andrews (2009) also found that none of the pairwise comparisons between French Frigate Shoals, Niihau, Kauai, and Oahu were statistically significant, [while](#) and [samples from](#) Oahu was [were](#) not significantly differentiated from Maui/Lanai. Assignment tests, which may provide information about recent gene flow, show that for most islands and atolls within the Hawaiian Archipelago, more samples [were](#) assigned to the island/atoll at which they were collected than to any other island. These patterns are supported by available photo-ID and animal movement data (Karczmarski *et al.* 2005). Spinner dolphin genetic data are lacking from some islands and atolls within the Hawaiian Archipelago (e.g., Molokai, Kahoolawe, Nihoa, Mokumanamana (Necker), Gardner Pinnacles, Laysan, and Lisianski). Sighting data confirms the presence of spinner dolphins at some of these locations (e.g., Molokai, Kahoolawe, Mokumanamana, and Gardner Pinnacles; PIFSC unpublished data), however, without genetic or photo-identification data it is difficult to evaluate connectivity between these dolphins and those at other islands.

Hill *et al.* (2010) proposed designation of island-associated stocks of spinner dolphins at Midway/Kure, Pearl and Hermes Reef, Kauai/Niihau, Oahu/4-Islands, and Hawaii Island based on microsatellite and mtDNA genetic data (Andrews *et al.* 2010), known movement patterns (Karczmarski 2005), and the geographic distances between the Hawaiian Islands. They suggested an offshore boundary for each island-associated stock at 10 nmi from shore based on anecdotal accounts of spinner dolphin distribution. Analysis of individual spinner dolphin movements suggests that few individuals move long distances (from one main Hawaiian Island to another) and no dolphins have been seen farther than 10 nmi from shore (Hill *et al.* 2011). Based on the maximum distance from shore observed for island-associated animals, a 10 nmi stock boundary has been assumed for management under the MMPA. Norris *et al.* (1994) suggested that spinner dolphins may move between leeward and windward shores of the main Hawaiian Islands seasonally.

For the Marine Mammal Protection Act (MMPA) stock assessment reports, there are six stocks within the U.S. EEZ of the Hawaiian Islands: 1) Hawaii Island, 2) Oahu/4-Islands, 3) Kauai/Niihau, 4) Pearl & Hermes Reef, 5) Kure/Midway, and 6) Hawaii Pelagic, including animals found both within the Hawaiian Islands EEZ (outside of island-associated boundaries) and in adjacent high seas waters. Because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of the Hawaii pelagic stock is evaluated based on data from U.S. EEZ waters of the Hawaiian Islands (NMFS 20052016). Spinner dolphins in the eastern tropical Pacific that may interact with tuna purse-seine fisheries are managed separately under the MMPA.

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

New Serious Injury Guidelines

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

Fishery Information

Information on fishery-related mortality of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaii-based fisheries cause marine mammal mortality and serious injury in other U.S. [waters](#) fisheries. Entanglement in gillnets and hooking or entanglement in various hook and line fisheries have been reported for small cetaceans in Hawaii (Nitta & Henderson, 1993). Although gillnet fisheries are not observed or monitored through any

State or Federal program, State regulations ban gillnetting around Maui and much of Oahu and require gillnet fishermen to monitor their nets for bycatch every 30 minutes. The bottomfish handline fishery in the Northwestern Hawaiian Islands was observed from 1990 to 1993, resulting in an estimate of 2.67 cetacean interactions per 1,000 landed fish, though none are thought to involve spinner dolphins (Kobayashi and Kawamoto 1995), and again in 2003 to 2005 (18-25% observer coverage) resulting in no incidental takes of cetaceans (NMFS PIR Observer Program). The bottomfish fishery is no longer permitted for the Northwestern Hawaiian Islands. Bottomfish fishermen in the main Hawaiian Islands claim interactions with dolphins that steal bait and catch are increasing. It is not known whether these alleged interactions result in serious injury or mortality of dolphins, nor whether spinner dolphins are involved. [Seven](#) Two-spinner dolphins [were](#) have been reported hooked or entangled by fishing gear [or marine debris](#) in the main Hawaiian Islands [from 2012 through 2016, five from the Hawaii Island stock and two from the Oahu/4-Islands stock](#) between 2007 and 2011 (Bradford & Lyman 2013 [in review](#)). One animal was seen in November 2009 off Lahaina, Maui (Oahu/4 Islands stock) with a hook embedded in its right lower jaw and through the tongue, preventing the dolphin from closing its mouth. The animal was seen again two days later, but has not been seen since. One additional spinner dolphin was seen in September 2011 off Kailua-Kona, Hawaii (Hawaii Island stock) with a section of netting entangled around its rostrum and trailing down its side. The animal was swimming behind other dolphins in the group and may not have been able to open its mouth. [All cases were reviewed following](#) Based on the description and photographs, both injuries are considered serious under the most recently developed [the](#) criteria for assessing serious injury in marine mammals (NMFS 2012). [In two separate cases off Kailua-Kona \(2012 and 2014\), a spinner dolphin was observed with line, net, or other debris entangled around its rostrum preventing the dolphin from opening its mouth, and in some cases with additional trailing gear \(see Bradford and Lyman in review for details\). Both cases were considered serious injuries given the potential of the line to impact the animal's ability to feed. In April 2013, a spinner dolphin was observed off Mahaiula Beach, Hawaii entangled in fishing gear \(300+ ft. of fishing line, float, glow stick and hook\). A swimmer cut the line close to the body, removing much of the trailing line and associated gear, but leaving several wraps of line around the dolphin's tail. This animal was considered seriously injured despite the removal of much of the gear because it was unclear whether the mitigation improved the animal's status. In June 2016, a spinner dolphin was observed off Kailua-Kona, Hawaii with a single wrap of small gauge fishing line around and cutting into its tail stock and trailing 40-50 feet behind. A diver removed most of the trailing line, reducing the length to about 6 feet. The animal was considered seriously injured because the constricting wrap remained and was possibly worsened by the attempt to remove the gear. In March 2014, a male spinner dolphin stranded off Keahole Pt, Hawaii with twine netting wrapped around its rostrum and peduncle. Examination revealed hemorrhage at the rostrum and peduncle and suggested the animal had drowned as a result of the entanglement. In March 2013 a spinner dolphin was observed off Waikiki, Oahu with a bag through its mouth and wrapped behind its head. This entanglement was considered a serious injury given the bag was unlikely to degrade causing an adverse health response. In January 2014 a spinner dolphin was observed at the entrance of Manele Bay, Lanai with red line/net wrapped around its rostrum and trailing down part of the body. This entanglement was considered a serious injury as the placement of the wrap could impact the animal's ability to feed. It is not possible to attribute \[any of the either-interactions involving fishing gear\]\(#\) to a specific fishery given insufficient details about the gear involved. There are \[eight\]\(#\) ~~six~~ additional reports between 1991 and ~~2011~~ 2006 of spinner dolphins found entangled, hooked, or shot \(Bradford & Lyman 2013\). No estimates of annual human-caused mortality and serious injury are available for nearshore hook and line or gillnet fisheries because these fisheries are not observed or monitored for protected species interactions.](#)

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 (SLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas. However, there are fishery closures within 25-75 miles from shore in the MHI and 50 miles from shore in the NWHI where insular or island-associated stocks occur. Between ~~2012~~ 2007 and ~~2016~~ 2011, no spinner dolphins were observed hooked or entangled in [either the deep-set \(20-22% observer coverage\) or shallow-set \(100% observer coverage\) longline fisheries operating in pelagic waters of the Hawaii EEZ and surrounding high-seas](#) the SLL fishery (100% observer coverage) or the DSLL fishery (20-22% observer coverage) (McCracken 2013, Bradford & Forney 2013 [in prep](#)).

HAWAII ISLAND STOCK POPULATION SIZE

Over the past few decades [several](#) abundance estimates have been produced from studies along the Kona coast of Hawaii Island. Norris *et al.* (1994) photo-identified 192 individuals [primarily within Kealekekua Bay](#) along the west coast of Hawaii and estimated 960 animals for this area in 1979-1980. ~~For the same study region,~~ Östman (1994) photo-identified 677 individual spinner dolphins [from a broader region, extending north to the Kohala Coast](#), from 1989 to 1992 and using the same estimation procedures as Norris *et al.* (1994), estimated a population size of

2,334 spinner dolphins. ~~New~~ From 2010 to 2012, open mark-recapture estimates based on intensive year-round photo-identification surveys for spinner dolphins occurred in Kauhako Bay, Kealakekua Bay, Honaunau Bay, and Makako Bay along the Kona Coast of Hawaii Island. These surveys represent the most systematic and geographically extensive surveys for spinner dolphins in this region. Several mark-recapture models were evaluated with available data to examine sampling design impacts. Models that utilized the most complete dataset yielded abundance estimates of 617 (CV=0.09) in 2011 and 665 (CV=0.09) in 2012 (Tyne *et al.* 2016). in 2010 and 2011 have resulted in an abundance estimate of 631 (CV=0.09) for the Hawaii Island stock (Tyne *et al.* 2013). These are the best available and most recent abundance estimates for this stock. Considerable seasonal variation in spinner dolphin occurrence on the leeward versus south and east sides of the island is ~~thought to~~ may occur, with lower abundance off the leeward Kona coast in the winter, potentially due to increased wind and swell in that region (Norris *et al.* 1994). Because the most recent abundance estimate is based on year-round surveys, ~~at least some of the animals~~ seasonally present on the leeward side ~~seasonally~~ have likely been seen. However, because only four Bays were surveyed, it is likely that some portion of the population is not included in this abundance estimate and the new estimate is an underestimate of total population size.

Minimum Population Estimate

The minimum population size is calculated as the lower 20th percentile of the log-normal 20th distribution (Barlow *et al.* 1995) around the 2012 ~~2011~~ abundance estimate for Hawaii Island, or ~~585~~ 617 spinner dolphins.

Current Population Trend

Quantitative trend analyses have not been conducted with the available data, as estimates from the 1970s and 1980s did not include year-round surveys and used a different survey area than the 2010-2011 surveys. Tyne *et al.* (2016) evaluated the impact of sampling intensity and frequency on the ability to detect trends within this population and estimated that 6 annual estimates resulting from 7 years of monthly surveys at all four monitored bays would be required to detect a 5% change in population size with 80% power. Abundance estimates resulting from surveys at 3 year intervals would detect change with fewer surveys, over a longer time period (9-12 years).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Hawaii Island stock is calculated as the minimum population estimate (617 ~~585~~) times one half the default maximum net growth rate for cetaceans ($\frac{1}{2}$ of 4%) times a recovery factor of 0.50 (for a stock of unknown status ~~with no estimated fishery mortality or serious injury within the U.S. EEZ of the Hawaiian Islands; Wade and Angliss 1997~~) resulting in a PBR of 6.2 ~~5.9~~ spinner dolphins per year.

STATUS OF STOCK

The Hawaii Island stock of spinner dolphins is not considered a strategic stock under the MMPA. The status of Hawaii Island spinner dolphins relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance for this stock. Spinner dolphins are not listed as “threatened” or “endangered” under the Endangered Species Act (1973), nor designated as “depleted” under the MMPA. Insufficient information is available to determine whether the total fishery mortality and serious injury for the Hawaii Island spinner dolphin stock is insignificant and approaching zero mortality and serious injury rate. Serious injuries from fishing gear or marine debris of unknown origin totaled 5 animals during 2012-2016, or 1 animal annually. This represents a minimum accounting of anthropogenic serious injuries, as not all cases are detected.

A habitat issue of increasing concern is the potential effect of swim-with-dolphin programs and other tourism activities on spinner dolphins around the main Hawaiian Islands (Danil *et al.* 2005, Courbis & Timmel 2009). A two year study of behavioral time-series data indicates that spinner dolphins off the leeward coast of Hawaii Island spatially and temporally partition their behavioral activities on a daily basis (Tyne *et al.* 2017), with resting behavior most common mid-day and travel and socializing in early morning and late afternoon. Foraging was not observed during the daytime. This behavior pattern suggests they are less resilient to human disturbance than other cetaceans. Further, Tyne *et al.* (2015) observed that spinner dolphins do not engage in rest behavior outside of sheltered bays, such that displacement from resting bays by tourist or other activities, would reduce rest time, with potential for long-term health consequences for the population. Heenehan *et al.* (2017a) measured acoustic response of spinner dolphins to human activities and found that dolphins increased vocal activity in two bays with predominantly dolphin-directed

[activities, but less so in bays with noise attributed to a broader range of human activities, suggesting greater behavioral disruption by human activities directed at the dolphins.](#)

All Hawaiian spinner dolphin stocks are potentially exposed to high levels of Navy sonar and frequent detonations during training exercises. The sensitivity of spinner dolphins to these sound levels is unknown and therefore the impact of these exercises on spinner dolphin stocks is unknown. [Naval sonar has been detected within spinner dolphin resting bays, with median sonar exposure levels between 24.7 and 45.8 dB above median sound levels on one occasion in 2011 \(Heenehan *et al.* 2017b\). Detection of the same sonar event at multiple spinner dolphin resting bays suggests that the entire spinner dolphin stock may be exposed to a single sonar event.](#)

[One spinner dolphin found stranded on Oahu has tested positive for *Morbillivirus* \(Jacob 2012\). Although *morbillivirus* is known to trigger lethal disease in cetaceans \(Van Bressem *et al.* 2009\), its impact on the health of the stranded animal is unknown \(Jacob 2012\). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters \(Jacob 2012\), raises concerns about the history and prevalence of this disease in Hawaii and potential impacts on Hawaiian cetacean populations. A spinner dolphin stranded off Hawaii Island was also determined to have died from infection with toxoplasmosis in 2015 \(Hawai'i Department of Land and Natural Resources 2018\). A retrospective analysis of all previously-stranded and archived spinner dolphins from Hawaii is underway to determine if others may have died from toxoplasmosis.](#) Spinner dolphins are not listed as “threatened” or “endangered” under the Endangered Species Act (1973), nor designated as “depleted” under the MMPA. Insufficient information is available to determine whether the total fishery mortality and serious injury for this Hawaii Island spinner dolphin stock is insignificant and approaching zero mortality and serious injury rate.

OAHU/4-ISLANDS STOCK POPULATION SIZE

As part of the Marine Mammal Research Program of the Acoustic Thermometry of Ocean Climate (ATOC) study, a total of twelve aerial surveys were conducted within 25 nmi of the main Hawaiian Islands in 1993, 1995 and 1998. An abundance estimate of 3,184 (CV=0.37) spinner dolphins was calculated from the combined survey data (Mobley *et al.* 2000), now representing the Kauai/Niihau, Oahu/4-Islands, and Hawaii Island stocks. [It is not feasible to partition this estimate into island-specific abundance estimates given the available data.](#) New mark-recapture estimates based on photo-identification studies have resulted in new seasonal abundance estimates for the Oahu/4-Islands stock. Closed capture models provide two separate estimates for the leeward coast of Oahu representing different time periods: 160 (CV = 0.14) for June to July, 2002; and 355 (CV = 0.09) for July to September 2007 (Hill *et al.* 2011). [Both the 2002 and 2007 estimates likely underestimate true abundance as they include only dolphins found off the leeward coast of Oahu, and do not account for individuals that may spend most of their time along other parts of Oahu or somewhere in the 4-Islands area.](#) The 2007 estimate is now more than 8 years old and therefore will no longer be used based on NMFS Guidelines for Assessing Marine Mammal Stocks (NMFS 2005/2016). The 2007 estimate is considered the best available estimate of the population size of the Oahu/4-Islands stock; however, it is likely an underestimate as it includes only dolphins found off the leeward coast of Oahu, and does not account for individuals that may spend most of their time along other parts of Oahu or somewhere in the 4-Islands area.

Minimum Population Estimate

[No minimum population estimate is available for this stock, as the most recent estimate of abundance is greater than 8 years old.](#) The minimum population size is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) from the 2007 abundance estimate for the summertime leeward coast of Oahu and the 4-Islands area, or 329 spinner dolphins. This minimum estimate is likely negatively biased, as it represents a minimum estimate of the number of dolphins, accounting only for those along the leeward Oahu coast in 2007; no data were included from the rest of the stock range.

Current Population Trend

There are insufficient data to evaluate trends in abundance for this stock.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for this species in Hawaiian waters. A default level of 4% is assumed for maximum net productivity rate.

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Oahu/4-Islands stock is calculated as the minimum population estimate (329) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery

factor of 0.50 (for a species of unknown status with no estimated fishery mortality or serious injury within the U.S. EEZ of the Hawaiian Islands; Wade and Angliss 1997), resulting in a PBR of 3.3 spinner dolphins per year. [Because there is no minimum population estimate for Oahu/4-Islands spinner dolphins, the potential biological removal \(PBR\) is undetermined.](#)

STATUS OF STOCK

The Oahu/4-Islands stock of spinner dolphins is not considered a strategic stock under the MMPA. The status of Oahu/4-Islands spinner dolphins relative to OSP is unknown, and there are insufficient data to evaluate [abundance trends in abundance for this stock. Spinner dolphins are not listed as “threatened” or “endangered” under the Endangered Species Act \(1973\), nor designated as “depleted” under the MMPA. Two serious injuries due to fishing gear and/or marine debris of unknown origin were documented from 2012-2016, or 0.4 animals annually. This represents a minimum accounting of anthropogenic serious injuries, as not all cases are detected. Insufficient information is available to determine whether the total fishery mortality and serious injury for this Oahu/4-Islands spinner dolphin stock is insignificant and approaching zero mortality and serious injury rate.](#)

A habitat issue of increasing concern is the potential effect of swim-with-dolphin programs and other tourism activities on spinner dolphins around the main Hawaiian Islands (Danil *et al.* 2005, Courbis and Timmel 2009). [A two year study including collection of behavioral time-series data indicates that spinner dolphins off the leeward coast of Hawaii Island spatially and temporally partition their activities on a daily basis \(Tyne *et al.* 2017\). Resting behavior is most common during midday, while travel and socializing occur in early morning and late afternoon. Foraging was not observed during the daytime. This behavior pattern suggests spinner dolphins are less resilient to human disturbance than other cetaceans. Further, Tyne *et al.* \(2015\) observed that spinner dolphins do not engage in rest behavior outside of sheltered bays, therefore, displacement from resting bays by human activities would reduce rest time, with potential for long-term population health consequences. Heenehan *et al.* \(2017\) measured spinner dolphin acoustic responses to human activities and found that the dolphins increased vocal activity in two bays with predominantly dolphin-directed activities, such as swim-with-dolphin programs. Increases in vocal activity was less pronounced in bays where noise was attributed to a broader range of human activities, suggesting greater behavioral disruption by dolphin-directed human activities.](#)

All Hawaiian spinner dolphin stocks are potentially exposed to high levels of Navy sonar and frequent detonations during training exercises. The sensitivity of spinner dolphins to these sound levels is unknown and therefore the impact of these exercises on spinner dolphin stocks is unknown. [Naval sonar has been detected within monitored spinner dolphin resting bays on Hawaii Island, with median sonar exposure levels between 24.7 and 45.8 dB above median sound levels on one occasion in 2011 \(Heenehan *et al.* 2017b\). Detection of the same sonar event at multiple spinner dolphin resting bays suggests that the entire spinner dolphin stock may be exposed to a single sonar event. Naval training also occurs near the other main Hawaiian Islands, suggesting the Hawaii Islands observations are not unique. Spinner dolphins are not listed as “threatened” or “endangered” under the Endangered Species Act \(1973\), nor designated as “depleted” under the MMPA. Insufficient data exist to determine whether the total fishery mortality and serious injury for this Oahu/4-Islands spinner dolphin stock is insignificant and approaching zero mortality and serious injury rate.](#)

One spinner dolphin found stranded on Oahu has tested positive for *Morbillivirus* (Jacob 2012). Although *morbillivirus* is known to trigger lethal disease in cetaceans (Van Bressemer *et al.* 2009), its impact on the health of the stranded animal is not known (Jacob 2012). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters (Jacob 2012), raises concerns about the history and prevalence of this disease in Hawaii and the potential population impacts on Hawaiian cetaceans. [A spinner dolphin stranded off Hawaii Island was also determined to have died from infection with toxoplasmosis in 2015 \(Hawai’i Department of Land and Natural Resources 2018\). A retrospective analysis of all previously stranded and archived spinner dolphins from Hawaii is now underway to determine if others may have died from the disease.](#)

KAUAI/NIIHAU STOCK POPULATION SIZE

As part of the Marine Mammal Research Program of the Acoustic Thermometry of Ocean Climate (ATOC) study, ~~a total of twelve~~ aerial surveys were conducted within 25 nmi of the main Hawaiian Islands in 1993, 1995 and 1998. An abundance estimate of 3,184 (CV=0.37) spinner dolphins was calculated from the combined survey data (Mobley *et al.* 2000), now representing the Kauai/Niihau, Oahu/4-Islands, and Hawaii Island stocks. ~~Those data are well over 8 years old and abundance estimates from these data are out of date. More recent~~ New mark-recapture estimates based on photo-identification studies have resulted in a new seasonal abundance estimate for the Kauai/Niihau stock. Closed capture models provide an estimate of 601 (CV = 0.20) spinner dolphins for the leeward

coast of Kauai for the period October to November 2005. This estimate is considered the best available estimate of the population size of the Kauai/Niihau stock; however, it is likely an underestimate as it includes only dolphins found off the leeward coast of Kauai, and does not account for individuals that may spend most of their time along other parts of Kauai, Niihau, or Kaula Rock. [The 2005 estimate is now more than 8 years old and therefore will no longer be used based on NMFS Guidelines for Assessing Marine Mammal Stocks \(NMFS 2016\).](#)

Minimum Population Estimate

[No minimum population estimate is available for this stock, as the most recent estimate of abundance is greater than 8 years old.](#) The minimum population size is calculated as the lower 20th percentile of the log-normal distribution (Barlow et al. 1995) from the leeward Kauai abundance estimate, or 509 spinner dolphins. This minimum estimate is likely negatively biased, as it represents a minimum estimate of the number of dolphins, accounting only for those along the leeward Kauai coast in 2005; no data were included from the rest of the stock range near Niihau or Kaula Rock.

Current Population Trend

There is only one abundance estimate available for the stock area of Kauai/Niihau from 2005 and thus, no trend analysis is possible.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for this species in Hawaiian waters. A default level of 4% is assumed for maximum net productivity rate.

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Kauai/Niihau stock is calculated as the minimum population estimate (509) ~~times~~ one half the default maximum net growth rate for cetaceans ($\frac{1}{2}$ of 4%) ~~times~~ a recovery factor of 0.50 (for a stock of unknown status with no estimated fishery mortality or serious injury within the U.S. EEZ of the Hawaiian Islands; Wade and Angliss 1997), ~~resulting in a PBR of 5.1 spinner dolphins per year.~~ [Because there is no minimum population estimate for Kauai/Niihau spinner dolphins, the potential biological removal \(PBR\) is undetermined.](#)

STATUS OF STOCK

The Kauai/Niihau stock of spinner dolphins is not considered a strategic stock under the MMPA. The status of Kauai/Niihau spinner dolphins relative to OSP is unknown, and there are insufficient data to evaluate abundance trends. [Spinner dolphins are not listed as “threatened” or “endangered” under the Endangered Species Act \(1973\), nor designated as “depleted” under the MMPA.](#) Insufficient data are available to determine whether the total fishery mortality and serious injury for this Kauai/Niihau spinner dolphin stock is insignificant and approaching zero mortality and serious injury rate.

A habitat issue of increasing concern is the potential effect of swim-with-dolphin programs and other tourism activities on spinner dolphins around the main Hawaiian Islands (Danil *et al.* 2005, Courbis & Timmel 2009). [A two year study including collection of behavioral time-series data indicates that spinner dolphins off the leeward coast of Hawaii Island spatially and temporally partition their activities on a daily basis \(Tyne *et al.* 2017\). Resting behavior is most common during midday, while travel and socializing occur in early morning and late afternoon. Foraging was not observed during the daytime. This behavior pattern suggests spinner dolphins are less resilient to human disturbance than other cetaceans. Further, Tyne *et al.* \(2015\) observed that spinner dolphins do not engage in rest behavior outside of sheltered bays, therefore, displacement from resting bays by human activities would reduce rest time, with potential for long-term population health consequences. Heenehan *et al.* \(2017\) measured spinner dolphin acoustic responses to human activities and found that the dolphins increased vocal activity in two bays with predominantly dolphin-directed activities, such as swim-with-dolphin programs. Increases in vocal activity was less pronounced in bays where noise was attributed to a broader range of human activities, suggesting greater behavioral disruption by dolphin-directed human activities.](#)

All Hawaiian spinner dolphin stocks are potentially exposed to high levels of Navy sonar and frequent detonations during training exercises. The sensitivity of spinner dolphins to these sound levels is unknown and therefore the impact of these exercises on spinner dolphin stocks is unknown. [Naval sonar has been detected within monitored spinner dolphin resting bays on Hawaii Island, with median sonar exposure levels between 24.7 and 45.8 dB above median sound levels on one occasion in 2011 \(Heenehan *et al.* 2017b\). Detection of the same sonar event at multiple spinner dolphin resting bays suggests that the entire spinner dolphin stock may be exposed to a single sonar](#)

[event. Naval training also occurs near the other main Hawaiian Islands, suggesting the Hawaii Islands observations are not unique.](#)

[One spinner dolphin found stranded on Oahu has tested positive for *Morbillivirus* \(Jacob 2012\). Although *morbillivirus* is known to trigger lethal disease in cetaceans \(Van Bressem *et al.* 2009\), its impact on the health of the stranded animal is not known \(Jacob 2012\). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters \(Jacob 2012\), raises concerns about the history and prevalence of this disease in Hawaii and the potential population impacts on Hawaiian cetaceans. A spinner dolphin stranded off Hawaii Island was also determined to have died from infection with toxoplasmosis in 2015 \(Hawai'i Department of Land and Natural Resources 2018\). A retrospective analysis of all previously stranded and archived spinner dolphins from Hawaii is now underway to determine if others may have died from the disease.](#) Spinner dolphins are not listed as “threatened” or “endangered” under the Endangered Species Act (1973), nor designated as “depleted” under the MMPA. Insufficient data are available to determine whether the total fishery mortality and serious injury for this Kauai/Niihau spinner dolphin stock is insignificant and approaching zero mortality and serious injury rate.

PEARL & HERMES REEF STOCK

POPULATION SIZE

There is no information on the abundance of the Pearl & Hermes Reef stock of spinner dolphins. A photo-identification catalog of individual spinner dolphins from this stock is available, though inadequate survey effort and low re-sighting rates prevent robust estimation of abundance.

Minimum Population Estimate

There is no information on which to base a minimum population estimate for the Pearl & Hermes Reef stock of spinner dolphins.

Current Population Trend

Insufficient data exists to assess population trends.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Pearl & Hermes Reef stock is calculated as the minimum population estimate times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a species of unknown status with no estimated fishery mortality or serious injury within the U.S. EEZ of the Hawaiian Islands; Wade and Angliss 1997). Because there is no minimum population estimate available for this stock the PBR for Pearl & Hermes Reef stock of spinner dolphins is undetermined.

STATUS OF STOCK

The Pearl & Hermes Reef stock of spinner dolphins is not considered a strategic stock under the MMPA. The status of Pearl & Hermes Reef spinner dolphins relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance for this stock. Spinner dolphins are not listed as “threatened” or “endangered” under the Endangered Species Act (1973), nor designated as “depleted” under the MMPA. Because the stock resides entirely within the Paphanaumokuakea Marine National Monument, where fishing is not permitted, it is assumed that the rate of mortality and serious injury within the stock area is zero. [It is unlikely that habitat issues facing spinner dolphin in the main Hawaiian Islands impact those in the northwestern Hawaiian Islands to the same magnitude given their relative isolation from tourism, military sonar activities, and urban water input to the environment. Pearl and Hermes stock spinner dolphins may be vulnerable to infection with *morbillivirus* or *Brucella*, though transmission through wild populations is not well understood and transmission may not necessarily be related to coastal proximity.](#)

MIDWAY ATOLL/KURE STOCK

POPULATION SIZE

In the Northwestern Hawaiian Islands, a multi-year photo-identification study at Midway Atoll resulted in a population estimate of 260 spinner dolphins based on 139 identified individuals (Karczmarski *et al.* 1998). This abundance estimate for the Midway Atoll/Kure stock of spinner dolphins is now more than 8 years old and therefore will no longer be used, based on NMFS Guidelines for Assessing Marine Mammal Stocks (NMFS 2005 [2016](#)). A 2010 shipboard line-transect survey within the Hawaiian EEZ resulted in a single off-effort sighting of spinner dolphins at

Kure Atoll. This sighting cannot be used within a line-transect framework; however, photographs of individuals may be used in the future to estimate the abundance of spinner dolphin at Midway Atoll/Kure using mark-recapture methods.

Minimum Population Estimate

The minimum population estimate for the Midway Atoll/Kure stock is now more than 8 years old and therefore will no longer be used (NMFS ~~2005~~2016). There is no current minimum population estimate available for this stock.

Current Population Trend

Insufficient data exists to assess population trends.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Midway Atoll/Kure stock is calculated as the minimum population estimate times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a species of unknown status with no estimated fishery mortality or serious injury within the U.S. EEZ of the Hawaiian Islands; Wade and Angliss 1997). The PBR for the Midway Atoll/Kure stock of spinner dolphins is undetermined because no minimum population estimate is available for this stock.

STATUS OF STOCK

The Midway Atoll/Kure stock of spinner dolphins is not considered strategic under the MMPA. The status of Midway Atoll/Kure spinner dolphins relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Spinner dolphins are not listed as “threatened” or “endangered” under the Endangered Species Act (1973), nor designated as “depleted” under the MMPA. Because the stock resides entirely within the Paphanaumokuakea Marine National Monument, where fishing is not permitted, it is assumed that the rate of mortality and serious injury within the stock area is zero. It is unlikely that habitat issues facing spinner dolphins in the main Hawaiian Islands impact those in the northwestern Hawaiian Islands to the same magnitude, given their relative isolation from tourism, military sonar activities, and urban water input to the environment. The Midway Atoll/Kure stock of spinner dolphins may be vulnerable to infection with morbillivirus or Brucella, though transmission through wild populations is not well understood and transmission may not necessarily be related to coastal proximity.

HAWAII PELAGIC STOCK POPULATION SIZE

A 2002 shipboard line-transect survey of the entire Hawaiian Islands EEZ resulted in an abundance estimate of 3,351 (CV=0.74) spinner dolphins (Barlow 2006); however, this estimate assumed a single Hawaiian Islands stock. Two of 8 sightings during the 2002 survey occurred in pelagic waters far outside of the current island-associated stock boundaries, suggesting at least some pelagic spinner dolphin occurrence in the archipelago. This estimate for the Hawaiian EEZ is more than 8 years old and therefore will no longer be used based on NMFS Guidelines for Assessing Marine Mammal Stocks (NMFS ~~2005~~2016). A 2010 shipboard line-transect survey within the Hawaiian EEZ did not result in any sightings of pelagic spinner dolphins.

Minimum Population Estimate

No minimum population estimate is available for this stock, as there were no sightings of pelagic spinner dolphins during a 2010 shipboard line-transect survey of the Hawaiian EEZ.

Current Population Trend

Insufficient data exists to assess population trends.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Hawaii pelagic stock is calculated as the minimum population estimate times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a species of unknown status with no estimated fishery mortality or serious injury within the U.S. EEZ of the Hawaiian Islands; Wade and Angliss 1997). Because there is no minimum population estimate for Hawaii pelagic spinner dolphins, the potential biological removal (PBR) is undetermined.

STATUS OF STOCK

The Hawaii pelagic stock of spinner dolphins is not considered a strategic stock under the MMPA. The status of Hawaii pelagic spinner dolphins relative to OSP is unknown, and there are insufficient data to evaluate abundance trends ~~in abundance for this stock~~. Spinner dolphins are not listed as “threatened” or “endangered” under the Endangered Species Act (1973), nor as “depleted” under the MMPA. While the estimated rate of fishery mortality and serious injury for this stock is zero in observed U.S. fisheries, this rate cannot be evaluated in the context of the Zero Mortality Rate Goal (ZMRG) (NMFS 2004) because ZMRG is calculated in the context of PBR (<10% of PBR), which is undetermined for this stock. This stock likely extends outside of U.S. EEZ waters, where international high seas fisheries may interact with and take animals from this stock.

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Appendix 3. Pacific reports revised in 2018 are highlighted. S=strategic stock, N=non-strategic stock. unk=unknown, undet=undetermined, n/a=not applicable.

Species (Stock Area)	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 (U.S.)	296,760	n/a	153,337	0.12	1	9,200	389	334	N	2007 2008 2011 2014	2014
	257,606		233,515			14,011	≥319	≥197		2012 2013 2014 2018	2018
Harbor seal (California)	30,968	n/a	27,348	0.12	1	1,641	43	30	N	2004 2009 2012 2014	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	2014
Guadalupe Fur Seal (Mexico to California)	20,000	n/a	15,830	0.137	0.5	542	≥3.2	≥3.2	S	2008 2009 2010 2016	2016
Northern Fur Seal (California)	14,050	n/a	7,524	0.12	1	451	1.8	≥0.8	N	2010 2011 2013 2015	2015
Monk Seal (Hawaii)	1,324	0.03	1,264	0.07	0.1	4.4	≥1.6	≥3.4	S	2013 2014 2015 2017	2017
	1,415	0.03	1,384			4.8	≥3.0	≥1.6		2014 2015 2016 2018	2018
Harbor porpoise (Morro Bay)	2,917	0.41	2,102	0.04	0.5	21	≥0.6	≥0.6	N	2002 2007 2012 2013	2013
Harbor porpoise (Monterey Bay)	3,715	0.51	2,480	0.04	0.5	25	0	0	N	2002 2007 2011 2013	2013
Harbor porpoise (San Francisco - Russian River)	9,886	0.51	6,625	0.04	0.5	66	0	0	N	2002 2007 2011 2013	2013
Harbor porpoise (Northern CA/Southern OR)	35,769	0.52	23,749	0.04	1	475	≥0.6	≥0.6	N	2002 2007 2011 2013	2013
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	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	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	2016
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	2016
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	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	2016
Common Bottlenose dolphin (California/Oregon/Washington Offshor)	1,924	0.54	1,255	0.04	0.45	11	≥1.6	≥1.6	N	2005 2008 2014 2016	2016
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	2016
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	2016
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	2016
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	2016
Killer whale (Eastern N Pacific Offshore)	240	0.49	162	0.04	0.5	1.6	0	0	N	2005 2008 2014 2016	2016
	300	0.1	276			2.8				2010 2011 2012 2018	2018
Killer whale (Eastern N Pacific Southern Resident)	83	n/a	83	0.035	0.1	0.44	0	0	S	2014 2015 2016 2017	2017
	77		77			0.13				2015 2016 2017 2018	2018
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	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	2017
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	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	2017

Appendix 3. Pacific reports revised in 2018 are highlighted. S=strategic stock, N=non-strategic stock. unk=unknown, undet=undetermined, n/a=not applicable.

Species (Stock Area)	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
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	0.9	0.7	S	2005 2008 2014	2017
Gray whale (Eastern N Pacific)	20,990	0.05	20,125	0.062	1.0	624	132	4.25	N	2009 2010 2011 2015	2014 2016 2018
	26,960	0.05	25,849			801	138	7.7			
Gray whale (Western N Pacific)	140	0.04	135	0.062	0.1	0.06	unk	unk	S		2011 2014 2014
	175	0.05	167			0.07				2012 2016	2018
Humpback whale (California/Oregon/Washington)	1,918	0.03	1,876	0.08	0.3	11.0	≥ 9.2	≥ 7.6	S	2005 2008 2014	2017
	2,900		2,784			16.7	≥ 38.6	≥ 14.1		2011 2013 2014	2018
Blue whale (Eastern N Pacific)	1,647	0.07	1,551	0.04	0.3	2.3	≥ 0.2	≥ 0.2	S	2005 2008 2011	2017 2018
							≥ 19	≥ 0.96			
Fin whale (California/Oregon/Washington)	9,029	0.12	8,127	0.04	0.5	81	≥ 2.0	≥ 0.2	S	2005 2008 2014	2016 2018
							≥ 43.5	≥ 0.5			
Sei whale (Eastern N Pacific)	519	0.4	374	0.04	0.1	0.75	0	0	S	2005 2008 2014	2016 2018
							≥ 0.2	≥ 0.2			
Minke whale (California/Oregon/Washington)	636	0.72	369	0.04	0.48	3.5	≥ 1.3	≥ 1.3	N	2005 2008 2014	2016
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)	72,528	0.39	52,833	0.04	0.5	423	2.1	2.1	N	2002 2010	2017
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)	11,613	0.43	8,210	0.04	0.5	82	0	0	N	2002 2010	2017
Common Bottlenose dolphin (Hawaii Pelagic)	21,815	0.57	13,957	0.04	0.5	140	0	0	N	2002 2010	2017
Common Bottlenose dolphin (Kaua'i and Ni'ihau)	n/a	n/a	97	0.04	0.5	1.0	unk	unk	N	2003 2012 2015	2017
Common Bottlenose dolphin (O'ahu)	n/a	n/a	n/a	0.04	0.5	undet	unk	unk	N	2002 2003 2006	2017
Common Bottlenose dolphin (4 Islands Region)	n/a	n/a	n/a	0.04	0.5	undet	unk	unk	N	2002 2003 2006	2017
Common Bottlenose dolphin (Hawaiian Island)	n/a	n/a	91	0.04	0.5	0.9	unk	unk	N	2002 2003 2006	2017
Pantropical Spotted dolphin (Hawaii Pelagic)	55,795	0.40	40,338	0.04	0.5	403.0	0	0	N	2002 2010	2017
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	2013 2018
Spinner dolphin (Hawaii Island)	634	0.04	585	0.04	0.5	5.9	unk	unk	N	1994 2003 2011	2013 2018
	665	0.09	617			6.2	≥ 1.0	unk		2010 2011 2012	2018
Spinner dolphin (O'ahu / 4 Islands)	355	0.09	329	0.04	0.5	3.3	unk	unk	N	1993 1998 2007	2013 2018
	n/a	n/a	n/a			undet	≥ 0.4			1998 2002 2007	2018

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Species (Stock Area)	N est	CV N est	N min	R max	Fr	PBR	Total Annual Mortality		Status	Recent Abundance Surveys				SAR Last
							Injury + Serious	Injury + Serious		1995	1998	2005	2013	
Spinner dolphin (Kaua'i / Ni'ihau)	604	0	509	0.04	0.5	5.1	unk	unk	N	1995	1998	2005	2013	
	n/a	n/a	n/a			undet								2018
Spinner dolphin (Kure / Midway)	unk	unk	unk	0.04	0.5	undet	unk	unk	N		1998	2010	2013	2018
Spinner dolphin (Pearl and Hermes Reef)	unk	unk	unk	0.04	0.5	undet	unk	unk	N			n/a	2013	2018
Spinner dolphin (American Samoa)	unk	unk	unk	0.04	0.5	undet	unk	unk	unk			n/a	2010	
Striped dolphin (Hawaii Pelagic)	61,021	0.38	44,922	0.04	0.5	449	0	0	N		2002	2010	2017	
Fraser's dolphin (Hawaii)	51,491	0.66	31,034	0.04	0.5	310	0	0	N		2002	2010	2017	
Melon-headed whale (Hawaiian Islands)	8,666	1.00	4,299	0.04	0.5	43	0	0	N		2002	2010	2017	
Melon-headed whale (Kohala Resident)	447	0.12	404	0.04	0.5	4.0	0	0	N			2009	2013	
Pygmy killer whale (Hawaii)	10,640	0.53	6,998	0.04	0.4	56.0	1.1	1.1	N		2002	2010	2017	
False killer whale (NW Hawaiian Islands)	617	1.11	290	0.04	0.4	2.3	0.4	0.4	N			2010	2017	
False killer whale (Hawaii Pelagic)	1,540	0.66	928	0.04	0.5	9.3	7.6	7.6	N		2002	2010	2017	
False killer whale (Palmyra Atoll)	1,329	0.65	806	0.04	0.4	6.4	0.3	0.3	N			2005	2013	
False killer whale (Main Hawaiian Islands Insular)	167	0.14	149	0.04	0.1	0.30	0.0	0.0	S		2013	2014	2015	2017
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)	146	0.96	74	0.04	0.5	0.7	0	0	N		2002	2010	2017	
Pilot whale, short-finned (Hawaii)	19,503	0.49	13,197	0.04	0.4	106	0.9	0.9	N		2002	2010	2017	
Blainville's beaked whale (Hawaii Pelagic)	2,105	1.13	980	0.04	0.5	10.0	0	0	N		2002	2010	2017	
Longman's Beaked Whale (Hawaii)	7,619	0.66	4,592	0.04	0.5	46.0	0	0	N		2002	2010	2017	
Cuvier's beaked whale (Hawaii Pelagic)	723	0.69	428	0.04	0.5	4.3	0	0	N		2002	2010	2017	
Pygmy sperm whale (Hawaii)	unk	unk	unk	0.04	0.5	undet	0	0	N		2002	2010	2013	
Dwarf sperm whale (Hawaii)	unk	unk	unk	0.04	0.5	undet	0	0	N		2002	2010	2013	
Sperm whale (Hawaii)	4,559	0.33	3,478	0.04	0.1	13.9	0.7	0.7	S		2002	2010	2017	
Blue whale (Central N Pacific)	133	1.09	63	0.04	0.1	0.1	0	0	S		2002	2010	2017	
Fin whale (Hawaii)	154	1.05	75	0.04	0.1	0.1	0	0	S		2002	2010	2017	
Bryde's whale (Hawaii)	1,751	0.29	1,378	0.04	0.5	13.8	0	0	N		2002	2010	2017	
Sei whale (Hawaii)	391	0.90	204	0.04	0.1	0.4	0.2	0.2	S		2002	2010	2017	
Minke whale (Hawaii)	unk	unk	unk	0.04	0.5	undet	0	0	N		2002	2010	2013	
Humpback whale (American Samoa)	unk	unk	150	0.106	0.1	0.4	0	0	S		2006	2007	2008	2009
Sea Otter (Southern)	2,826	n/a	2,723	0.06	0.1	8	≥0.8	≥0.8	S		2006	2007	2008	2008
Sea Otter (Washington)	n/a	n/a	1,125	0.2	0.1	11	≥0.2	≥0.2	N		2006	2007	2008	2008