U.S. PACIFIC MARINE MAMMAL DRAFT STOCK ASSESSMENTS: 2014

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Stock assessment reports and appendices revised in 2014. All others will be reprinted as they appear in the 2013 Pacific Region Stock Assessment Reports (Carretta *et al.* 2014).

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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 2014 Pacific marine mammal stock assessments include revised reports for 11 Pacific marine mammal stocks under NMFS jurisdiction, including six "strategic" stocks: Hawaiian monk seal, Southern Resident killer whale, Main Hawaiian Islands Insular false killer whale, Hawaii Pelagic false killer whale, California/Oregon/Washington sperm whale, and Western North Pacific gray whale. New abundance estimates are available for three stocks in the Pacific Islands region and five U.S. west coast stocks. New estimates of abundance for the California/Oregon/Washington stock of sperm whales are based on a Bayesian trend analysis that utilizes previously collected line-transect data (Moore and Barlow 2014), resulting in a more stable time series of abundance estimates. Mortality and serious injury estimates of California/Oregon/Washington sperm whales in California drift gillnets are updated, based on pooling additional years of data (>5 years) to reduce bias and improve precision in mean annual bycatch estimates (Carretta and Moore 2014). The combination of new abundance estimates and pooling of bycatch estimates over a longer time period for this stock of sperm whales results in mean annual bycatch estimates that no longer exceed PBR. In addition, a new stock assessment report for Western North Pacific gray whales is presented for the first time, prompted by new data showing that gray whales previously photographed in the western North Pacific utilize U.S. and Mexican waters. Stock Assessments for Alaska region marine mammals are published by the National Marine Mammal Laboratory (NMML) in a separate report.

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 2014 stock assessment reports were reviewed by the Pacific Scientific Review Group at the April 2014 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. and J.E. Moore. 2014. Recommendations for pooling annual bycatch estimates when events are rare. NOAA Technical Memorandum, NOAA-TM-NMFS-SWFSC-528. 11 p.

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Moore J.E., and Barlow J.P. 2014. Improved abundance and trend estimates for sperm whales in the eastern North Pacific from Bayesian hierarchical modeling. Endang. Species Res. 25:141-150.

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Revised ~~12/15/2011~~ 10/22/2014

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

~~The California sea lion (*Zalophus californianus*) is now considered to be a full species, separated from the Galapagos sea lion (*Z. wollebaeki*) and the extinct Japanese sea lion (*Z. japonicus*) (Brunner 2003, Wolf *et al*. 2007, Schramm et al. 2009).~~ 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 ~~of California sea lions~~ 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 ~~north~~ into Canadian waters, and movement of animals between U.S. waters and Baja California waters occurs. ~~has been documented, though the distance between the major U.S. and Baja California rookeries is at least 400 nmi.~~ Males from western Baja California rookeries may spend most of the year in the United States.

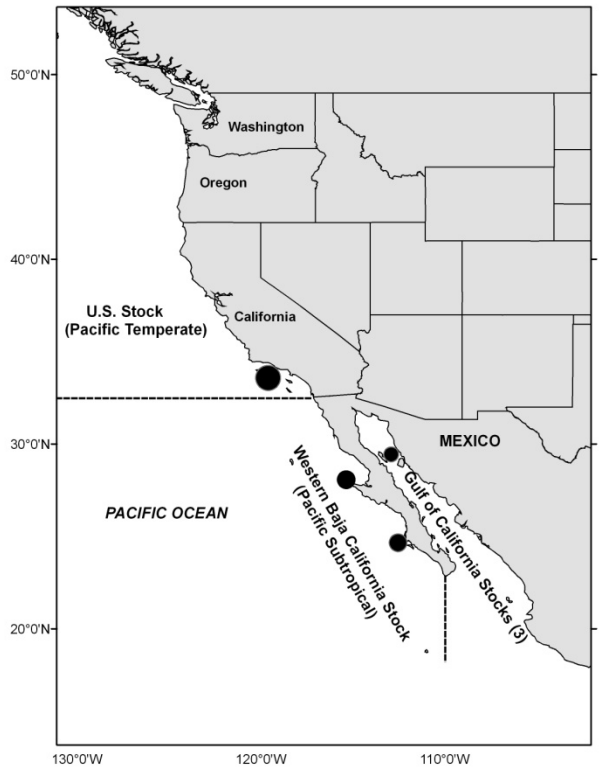


Figure 1. Geographic range of California sea lions showing stock boundaries and locations of major rookeries. The U.S. stock also ranges north into Canadian waters.

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. ~~The size of the p~~ 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. New 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.

Minimum Population Estimate

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.

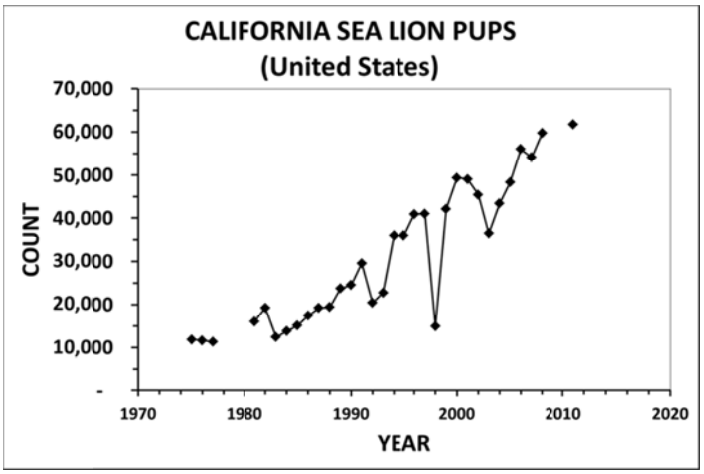


Figure 2. U.S. pup count index for California sea lions (1975-2011 ~~2008)~~. Trends in pup counts from 1975 through ~~2008~~ 2011 are shown for four rookeries in southern California and for haulouts in central and northern California. ~~Records of pup counts from 1975 to 2008 were compiled from Lowry and Maravilla (2005) and unpublished NMFS data.~~

Current Population Trend

Trends in pup counts from 1975 through ~~2008~~ 2011 are shown in Figure 2 for four rookeries in southern California and for haulouts in central and northern California. The number of pups at rookeries that were not counted were estimated using multiple regression~~s~~ 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, 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, ~~NMFS unpubl. data~~ 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)[1]. 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.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

~~A standard logistic growth model indicated that the maximum population growth rate (R_{max}) was 9.2 percent when pup counts from El Niño years (1983, 1984, 1992, 1993, 1998, and 2003) were removed (Figure 3).~~

[1] http://www.nmfs.noaa.gov/pr/health/mmume/californiasealions2013.htm

~~However, the apparent growth rate from the population trajectory underestimates the intrinsic growth rate because it does not consider human-caused mortality that was occurring during the time series.~~ ~~Here we~~We use the default maximum net productivity rate for pinnipeds (12% per year) (Wade and Angliss 1997).

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (153,337) 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, Wade and Angliss 1997); resulting in a PBR of 9,200 sea lions per year.

~~ANNUAL~~ HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Serious Injury Guidelines

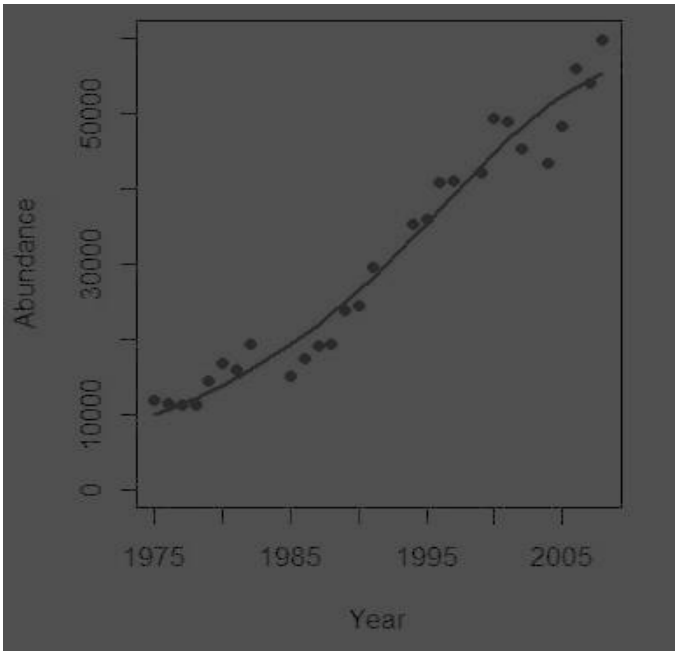
NMFS uses guidance from previous serious injury workshops, expert opinion, and analysis of historic injury cases to distinguish serious from non-serious injury (Angliss and DeMaster 1998, Andersen et al. 2008, NOAA 2012). NMFS defines serious injury as an "*injury that is more likely than not to result 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 incidentally in set and drift gillnet fisheries (Hanan et al. 1993; Barlow . 1994; Julian and Beeson, 1998; Carretta et al. 2005) and trawl fisheries along the U.S. west coast (Heery et al. 2010). Detailed information on these fisheries is provided in Appendix 1. Mortality estimates for the California set and drift gillnet fisheries and trawl fisheries are included in Table 1 for the five most recent years of monitoring (Carretta and Enriquez 2006, 2007, 2009a, 2009b, 2010, Heery et al. 2010). A controlled experiment during 1996-97 demonstrated that the use of acoustic warning devices (pingers) reduced sea lion entanglement rates considerably within the drift gillnet fishery (Barlow and Cameron 2003). However, entanglement rates increased again during the 1997 El Niño and continued during 1998. The reasons for the increase in entanglement rates are unknown. However, it has been suggested that sea lions may have foraged further offshore in response to limited food supplies near rookeries, which would provide opportunity for increased interactions with the drift gillnet fishery. Because of interannual variability in entanglement rates, additional years of data will be required to fully evaluate the effectiveness of pingers for reducing mortality of this particular species. Historically, the majority of California sea lion gillnet mortality was in the California halibut and white seabass set gillnet fishery (Julian and Beeson 1998), but this fishery has undergone regulatory changes that has reduced its range to southern California waters south of Pt. Arguello and has shifted fishing effort to greater than 3 nmi from the mainland or 1 nmi from the islands. There has~~



~~**Figure 3.** Fit of standard logistic growth curve to California sea lion pup counts, 1975-2008 (excluding El Niño years).~~

~~also been a considerable decline in fishing effort in this fishery since the early 1990s (see Figure 3 in Appendix 1). An observer program for the set gillnet fishery was in place during 2006 and 2007, although the only meaningful levels of observer coverage occurred in 2007. Annual estimates of bycatch mortality for this fishery are based solely on 2007 for that reason (Table 1). Logbook and observer data, and fishermen reports indicate that mortality of California sea lions occurs or has occurred in the past in the following fisheries: (1) California, Oregon, and Washington salmon troll; (2) Oregon and Washington non-salmon troll; (3) California herring purse seine; (4) California anchovy, mackerel, and tuna purse seine; (5) California squid purse seine; (6) Washington, Oregon, California and British Columbia, Canada salmon net pen ; (7) Washington, Oregon, and California groundfish trawl; (8) Washington, Oregon and California commercial passenger fishing vessels (NMFS 1995, M. Perez pers. comm, and P. Olesiuk pers. comm.) (9) California small mesh drift gillnet fishery, and (10) California anchovy, mackerel, and tuna purse seine. Not all of these fisheries continue to operate or have current observer programs.~~ 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, 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. ~~Stranding data from California, Oregon, and Washington during 2005-2009 show that an additional 55 sea lions died from unknown entangling net fisheries (Table 1). Animals are typically found on the beach or sometimes at sea with portions of gillnet wrapped around the carcass.~~ In addition to bycatch estimates from fishery observer programs, information on fishery-related sea lion deaths and serious injuries comes largely from strandings data (Carretta *et al.* 2013). ~~This~~ Strandings data represents a minimum number of animals killed or injured, as many entanglements are likely unreported or undetected.

~~Drift gillnet fisheries for swordfish and sharks exist along the entire Pacific coast of Baja California, Mexico and may take animals from the same population, but no quantitative estimates of recent mortality are available.~~

~~California sea lions injured by entanglement in gillnet and other man-made debris are observed at rookeries and haulouts (Stewart and Yochem 1987, Oliver 1991). The proportion of those entangled ranged from 0.08% to 0.35% of those hauled out, with the majority (52%) entangled in monofilament gillnets. Data from a marine mammal rehabilitation center showed that 87% of 87 rescued California sea lions were entangled in 4 to 4.5 inch mesh monofilament gillnet (Howorth 1994). Of California sea lions entangled in gillnets, 0.8% in set gillnets and 5.4% in drift gillnets were observed to be released alive from the net by fishers during 1991-1995 (Julian and Beeson 1998). Clearly, some are escaping from gillnets; however, the rate of escape from gillnets, as well as the mortality rate of these injured animals, is unknown.~~

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 ~~2005 and 2009~~ 2008 and 2012, there were ~~88~~ 124 California sea lion deaths / serious injuries attributed to hook and line fisheries, or an annual average of ~~18~~ 25 animals (~~NMFS Southwest and Northwest Regional Stranding Data, unpublished~~ Carretta *et al.*, 2014b). 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.* ~~and Enriquez~~ 2014a. ~~2006, 2007,~~ 2009~~a, 2009b~~, 2010, 2012a, 2012b; Heery *et al*. 2010; Jannot *et al*. 2011; Appendix 1). Mean annual takes are based on ~~2005-2009~~ 2008-2012 data unless noted otherwise.

Fishery Name	Year(s)	Data Type	Percent Observer Coverage	Observed Mortality	Estimated Mortality (CV in parentheses)	Mean Annual Takes (CV in parentheses)
CA/OR thresher shark/swordfish large mesh drift gillnet fishery	~~2005~~ ~~2006~~ ~~2007~~ 2008 2009 2010 2011 2012	observer	~~20.9%~~ ~~18.5%~~ ~~16.4%~~ 13.5% 13.3% 11.9% 19.5% 18.6%	~~1~~ ~~12~~ ~~8~~ 7 5 0 18 6	~~5 (0.97)~~ ~~64 (0.43)~~ ~~48 (0.65)~~ 51 (0.52) 37 (0.83) 0 (n/a) 92 (0.79) 32 (0.60)	~~41 (0.28)~~ 42 (0.50)

Fishery Name	Year(s)	Data Type	Percent Observer Coverage	Observed Mortality	Estimated Mortality (CV in parentheses)	Mean Annual Takes (CV in parentheses)
CA halibut and white seabass set gillnet fishery	~~2005~~ ~~2006~~ ~~2007~~ 2008 2009 2010 2011 2012	~~12 sets observed in 2006 and 248 sets observed in 2007~~	~~0%~~ ~~<1%~~ ~~17.8%~~ 0% 0% 12.5% 8.0% 5.5%	~~n/a~~ ~~0~~ ~~34~~ n/a n/a 25 6 18	~~n/a~~ ~~n/a~~ ~~190 (0.68)~~ n/a n/a 199 (0.30) 74 (0.39) 326 (0.33)	~~190 (0.68)~~[1] 200 (0.21)
CA small-mesh drift gillnet fishery for white seabass, yellowtail, barracuda, and tuna	~~2003~~ ~~2004~~ 2010 2011 2012	observer	~~11%~~ ~~11%~~ 0.7% 3.3% 4.6%	~~2~~ ~~1~~ 0 0 0	~~18 (0.71)~~ ~~9 (0.94)~~ 0 (n/a) 0 (n/a) 0 (n/a)	~~13.5 (0.57)~~ 0 (n/a)
CA anchovy, mackerel, sardine, and tuna purse-seine fishery	2004-2008	observer	~5%	2	n/a	≥2 (n/a)
WA, OR, CA domestic groundfish trawl fishery (includes at-sea hake and other limited-entry groundfish sectors)	~~2004~~ 2005 2006 2007 2008 2009	observer	~~99~~98% to 100% of tows in at-sea hake fishery ~~18% 26%~~ Generally less than 30% of landings observed in other groundfish sectors	~~8~~ 14 21 8 7 4	~~13 (n/a)~~ 21 (n/a) 95 (n/a) 31 (n/a) 13 (n/a) 10 (n/a)	~~34.6 (n/a)~~ 34 (n/a)
Unknown entangling net fishery	~~2005-2009~~ 2008-2012	stranding	n/a	~~55~~	n/a	~~≥ 55 (n/a)~~ ≥ 53 (n/a)
~~**Unknown pot or trap fishery**~~ **Unknown trawl fishery and bait barge net entanglement**	~~2005-2009~~ 2008-2012	stranding	n/a	~~1~~ 2	~~n/a~~	≥ ~~1~~ 2 (n/a)
Minimum total annual takes						~~≥ 337 (0.56)~~ ≥ 331 (0.14)

[1] ~~Only 2007 data is included in the mean annual take calculation for the CA halibut and white seabass fishery, due to the low observer coverage (<1%) in 2006.~~

Other Mortality

Live strandings and dead beach-cast California sea lions are regularly observed with gunshot wounds in California (Lowry and Folk 1987, ~~Deiter 1991, Barocchi et al. 1993,~~ Goldstein *et al*. 1999, ~~NMFS unpublished stranding data~~ Carretta *et al*. 2013). A summary of stranding records for ~~2005-2009~~ 2008 to 2012 from California, Oregon, and Washington ~~stranding databases~~ shows the following non-fishery related human-caused mortality and serious injuries: boat collisions (~~12 deaths~~ 13), car collisions (~~6 deaths~~ 3), entrainment in power plants (~~158 deaths~~ 59), shootings (~~113 deaths~~ 151), marine debris entanglement or ingestion (~~13 deaths~~ 37), research-related ~~permit-related takes~~ (~~3 deaths~~ 18), and ~~unknown~~ other sources, including dog attacks, harassment, seal bombs, stabbings, and, blunt force trauma (~~19 deaths~~ 10). Stranding records are a gross under-estimate of mortality and serious injury ~~and mortality~~ 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. ~~There are currently no estimates of the total number of California sea lions being killed or injured by guns, boat and car collisions, entrainment in power plants, marine debris, or gaffs, but the minimum number from 2005-2009 was 324, or an annual average of 65 animals.~~

Under authorization of MMPA Section 120, individually identifiable California sea lions have been killed or ~~captured~~ relocated since 2008 in response to their predation on endangered salmon and steelhead stocks in the Columbia River. ~~Captured~~ Relocated animals were transferred to aquaria and/or zoos. ~~Between April 2008 and September 2010, 40~~ Between 2009 and 2013, a total of 47 California sea lions were removed from this stock (~~30~~ 40 lethal removals and ~~10~~ 7 relocations to aquaria and/or zoos). The average annual mortality due to direct removals for the 2009-2013 period ~~April 2008 to September 2010 is 17~~ is 9.4 animals per year (relocations to aquaria/zoos are treated the same as mortality because animals are effectively removed from the stock).

Between ~~2005 and 2009~~ 2008 and 2012, ~~15~~ 18 California sea lions were incidentally killed, 2 seriously injured, and 8 non-serious injuries along the U.S. west coast during scientific trawl and longline operations conducted by NMFS ~~(Southwest Regional Office Stranding Program, unpublished data)~~ (Carretta *et al*., 2014b). The average annual research-related mortality and serious injury of California sea lions from ~~2005 to 2009~~ 2008 to 2012 is ~~3.0~~ 4.0 animals. ~~Sea lion mortality in 1998 along the central California coast has recently been linked to the algal-produced neurotoxin domoic acid (Scholin et al. 2000). Future mortality may be expected to occur, due to the sporadic occurrence of such harmful algal blooms.~~

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, entrainment in power plants, interactions with recreational hook and line fisheries, separation of mothers and pups due to human disturbance, dog bites, and vessel and vehicle strikes (Carretta et al. 2014b).

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 average annual commercial fishery mortality is ~~337~~ 331 animals per year (Table 1). Other sources of human-caused mortality (shootings, direct removals, recreational hook and line fisheries, tribal takes, entrainment in power plant intakes, etc.) average ~~94~~ 58 animals per year. Total human-caused mortality of this stock is at least ~~431~~ 389 animals per year. California sea lions are not considered "strategic" under the MMPA because total human-caused mortality is less than the PBR (9,200). The total fishery mortality and serious injury rate (~~337~~ 389 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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Revised ~~12/15/2011~~ 10/22/2014

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

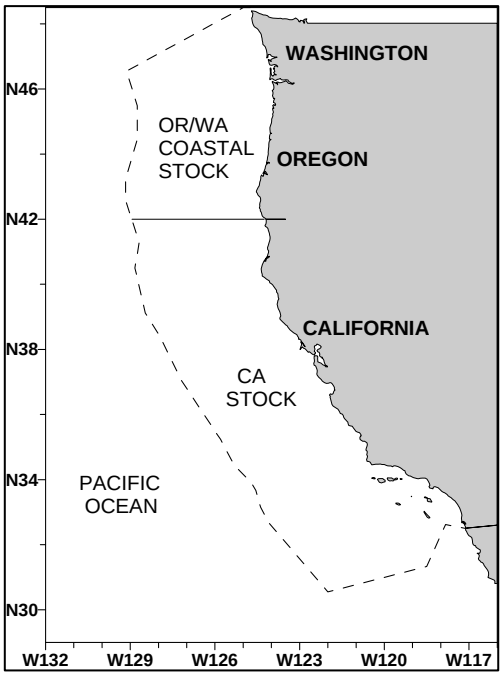


Figure 1. Stock boundaries for the California and Oregon/Washington coastal stocks of harbor seals. Dashed line represents the U.S. EEZ.

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

POPULATION SIZE

A complete count of all harbor seals in California is impossible because not all animals are hauled out at one time. ~~some are always away from the haulout sites.~~ A complete pup count (as is done for other pinnipeds in California) is also not possible because harbor seal ~~s are precocial, with~~ pups enter~~ing~~ the water almost immediately after birth. Population size is estimated by counting the number of seals ashore during the peak haul-out period (May

to July) and by multiplying this count by a correction factor equal to the inverse of the estimated fraction of seals on land. Harvey and Goley (2011) calculated a correction factor of 1.54 (CV=0.157) based on 180 seals radio-tagged in California. This correction factor is based on the mean of four date-specific correction factors (1.31, 1.38, 1.62, 1.84) calculated for central and northern California. Based on the most recent harbor seal counts during May-July of 2012 (20,109 animals) (~~19,608 in May-July 2009~~; NMFS unpublished data) and the Harvey and Goley (2011) correction factor, the harbor seal population in California in 2012 is estimated to number ~~30,196~~ 30,968 seals (CV=0.157).

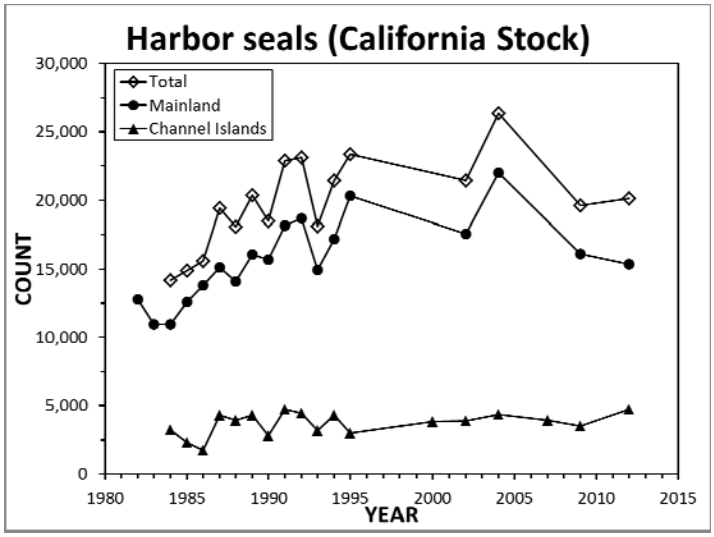


Figure 2. Harbor seal haulout counts in California during May to July (Hanan 1996; R. Read, CDFG unpubl. data; Lowry *et al*. 2008, NMFS unpubl. data from 2009-2012 surveys).

Minimum Population Estimate

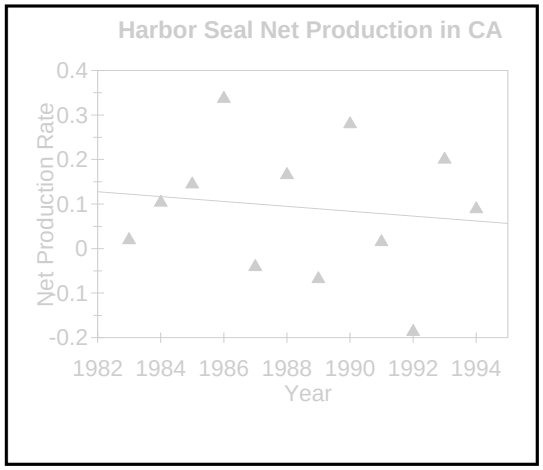
The minimum population size is estimated from the number of ~~seals counted~~ hauled out seals counted in ~~2009 (19,608)~~ 2012 (20,109), multiplied by the lower 20th percentile of the correction factor (1.36), or ~~26,667~~ 27,348 seals.

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

~~A realized rate of increase was calculated for the 1982-1995 period (when annual counts were available) by linear regression of the natural logarithm of total count versus year. The slope of this regression line was 0.035 (s.e.= 0.007) which gives an annualized growth rate estimate of 3.5%. The true rate of net production is greater than this observed growth rate because fishery and other human-caused mortality removes a fraction of the net production. Annual gillnet mortality may have been as high as 5-10% of the California harbor seal population in the mid-1980s; a kill this large would have depressed population growth rates appreciably. Net productivity was therefore calculated for 1980-1994 as the realized rate of population growth (increase in seal counts from year *i* to year *i*+1, divided by the seal count in year *i*) plus the human-caused mortality rate (fishery mortality in year *i* divided by population size in year *i*). Between 1983 and 1994, the net productivity rate for the California stock averaged 9.2% (Fig. 3). A regression shows a decrease in net production rates, but the decline is not statistically significant. Maximum net productivity rate cannot be estimated because measurements were not made when the stock size was very small. A current estimate of net production for the California harbor seal stock is difficult to determine because the fishery that was responsible for the most mortality (California halibut and white seabass set gillnet) has only been intermittently observed since the mid-1990s, and statewide annual counts of seals at rookeries are not available after 1995 (Fig. 2).~~



~~**Figure 3.** Net production rates and regression line estimated from haulout counts and fishery mortality.~~

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (~~26,667~~ 27,348) times one half the default maximum net productivity rate for pinnipeds (½ of 12%) times a recovery factor of 1.0 (for a stock of unknown status that is growing or for a stock at OSP, Wade and Angliss 1997), resulting in a PBR of ~~1,600~~ 1,641.

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Serious Injury Guidelines

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

Historical Takes

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

Fishery Information

A summary of known commercial fishery mortality and serious injury for this stock of harbor seals for the period 2008-2012 is given in Table 1. ~~More detailed information on these fisheries is provided in Appendix 1.~~ Historically, the set gillnet fishery for halibut and white seabass was the largest source of fishery mortality and remains the most likely fishery in California to interact with harbor seals today. Julian and Beeson (1998) reported a range of annual mortality estimates from 227 to 1,204 seals (mean = 584) from 1990 to 1994, based on 5% to 15% fishery observer coverage and representing between 1-2% of the estimated population size. The set gillnet fishery has been observed infreqently since 1995 and fishing effort has declined from approximately 5,000 trips in the early 1990s to 1,300 trips in 2012 (Carretta *et al*. 2014a.). ~~Regulations implemented in 1994 moved the fishery farther offshore in southern California, which may have reduced harbor seal entanglements in this region. The fishery was not observed again until 1999 and 2000 in Monterey Bay, although annual mortality estimates of 300-400 seals were still calculated based on 1990-1994 bycatch rates and 1999-2000 fishing effort (Cameron and Forney 2000, Cameron and Forney 2001, Carretta 2002, 2003). The observer program for this fishery was discontinued after 2000. In 2002 the fishery was subject to further area restrictions that effectively eliminated fishing north of Point Arguello, California. In 2006, the fishery was again observed at low levels (12 sets out of an estimated 1,300), with one observed death. In 2007, 248 sets were observed (~17% observer coverage) with 2 harbor seal deaths observed and a resulting mortality estimate of 11 animals (Table 1). Total effort in the set gillnet fishery has declined from approximately 4,000 sets annually to approximately 1,300 (Carretta and Enriquez 2009a). Stranding data from California between 2005 and 2009 include eight harbor seal deaths caused by hook and line fisheries. The total annual human-caused mortality from 2005 to 2009 from commercial fisheries is 18 animals per year (Table 1). There were also 7 harbor seal deaths attributed to recreational hook and line fisheries between 2005 and 2009 (NMFS, unpublished stranding data).~~

Other Mortality

NMFS stranding records for California for the period ~~2005-2009~~ 2008-2012 include the following human-caused mortality and serious injury not included in Table 1: shootings (~~2~~ 1), ship/vessel strikes (~~1~~ 3), entrainment in power plants (~~52~~ 40), hook and line fisheries (6), human-induced abandonment of pups or harassment (9), marine

debris entanglement (2), stabbing/gaff wounds (2), and research-related deaths (~~3~~ 1) (Carretta *et al*. 2014b.). The total non-fishery related mortality and serious injury for the period totals 64 harbor seals, or an annual average of 12.8 seals. ~~This results in an annual average of 12 harbor seal deaths per year for the years 2005-2009.~~

Table 1. Summary of available information on the mortality and serious injury of harbor seals (California stock) in commercial fisheries that might take this species (Carretta and Enriquez 2006, 2009, Carretta *et al*. 2014a; Heery *et al*. 2010); n/a indicates that data are not available. Mean annual takes are based on ~~2005-2009~~ 2008-2012 data unless noted otherwise.

Fishery Name	Year(s)	Data Type	Percent Observer Coverage	Observed Mortality	Estimated Mortality (CV in parentheses)	Mean Annual Takes (CV in parentheses)
CA halibut and white seabass set gillnet fishery	~~2005~~ ~~2006~~ ~~2007~~ 2008 2009 2010 2011 2012	observer	~~0%~~ ~~<1%~~ ~~17.8%~~ 0% 0% 12.5% 8.0% 5.5%	~~0~~ ~~0~~ ~~2~~ 0 0 3 0 0	~~n/a~~ ~~n/a~~ ~~11 (0.73)~~ n/a n/a 23 (0.59) n/a n/a	~~11 (0.73)[1]~~ 23 (0.59)
CA small-mesh drift gillnet fishery for white seabass, yellowtail, barracuda, and tuna	2010 2011 2012	observer	0.7% 3.3% 4.6%	0 0 0	0 (n/a) 0 (n/a) 0 (n/a)	0 (n/a)
~~**CA anchovy, mackerel, sardine, and tuna purse seine fishery**~~	~~2004-2006~~	~~observer~~	~~~2%~~	~~0~~	~~0~~	~~0~~
WA, OR, CA groundfish trawl (includes at-sea hake and other limited-entry groundfish sectors)	~~2004~~ 2005 2006 2007 2008 2009	observer	99% to 100% of tows in at-sea hake fishery; 18%-26% of landings in other groundfish sectors	~~1~~ 1 1 0 4 1	~~1 (n/a)~~ 1 (n/a) 1 (n/a) 0 (n/a) 29 (n/a) 1 (n/a)	~~6.4 (n/a)~~ 6.4 (n/a)
~~**CA squid purse seine fishery**~~	~~2004-2006~~	~~observer~~	~~~5%~~	~~0~~	~~0~~	~~0~~
Unknown net fisheries	~~2005-2009~~ 2008-2012	stranding	n/a	5	n/a	≥~~0.8~~ 1.0
Total annual takes						~~18 (0.73)~~ 30 (0.59)

~~[1] Only 2007 data is included in the mean annual take calculation for the CA halibut and white seabass fishery, due to the low observer coverage (<1%) in 2006.~~

STATUS OF STOCK

A review of harbor seal dynamics through 1991 concluded that their status relative to OSP could not be determined with certainty (Hanan 1996). California harbor seals are not listed as "endangered" or "threatened" under the Endangered Species Act nor designated as "depleted" under the MMPA. Annual human-caused mortality from commercial fisheries (~~18/yr~~ 30/yr) and other human-caused sources (~~13/year~~ 12.8/yr) is ~~31~~ 42.8 animals, which is less than the calculated PBR for this stock (~~1,600~~ 1,641), and thus they would not be considered a "strategic" under the MMPA. ~~The fishery that historically removed the largest numbers of harbor seals (halibut and white seabass set gillnet) has been observed only intermittently in recent years, but annual bycatch from 2007 when the fishery had ~18% observer coverage indicates that current rates of absolute bycatch are much lower than during the 1990s.~~ The average annual rate of incidental commercial fishery mortality (~~18~~ 30 animals) is less than 10% of the calculated PBR (~~1,600~~ 1,641 animals); therefore, fishery mortality is considered insignificant and approaching zero mortality and serious injury rate. The population size has increased since the 1980s when statewide censuses were first conducted. The highest population counts occurred in 2004 and subsequent counts in 2009 and 2012 have been lower~~.~~. ~~appears to be stabilizing at what may be its carrying capacity, and the fishery mortality is declining.~~ Expanding pinniped populations in general have resulted in increased human-caused serious

injury and mortality, due to shootings, entrainment in power plants, interactions with recreational hook and line fisheries, separation of mothers and pups due to human disturbance, dog bites, and vessel and vehicle strikes (Carretta et al. 2014b). ~~There are no known habitat issues that are of particular concern. Two unexplained harbor seal mortality events occurred in Point Reyes National Seashore involving at least 90 seals in 1997 and 16 seals in 2000. Necropsy of three seals in 2000 showed severe pneumonia; tests for morbillivirus were negative, but attempts are being made to identify another virus isolated from one of the three (F. Gulland, pers. comm.).~~ All west-coast harbor seals that have been tested for morbilliviruses were found to be seronegative, indicating that this disease is not endemic in the population and that this population is extremely susceptible to an epidemic of this disease (Ham-Lammé *et al*. 1999).

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Revised ~~10/30/2007~~ 10/22/2014

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

Northern elephant seals breed and give birth in California (U.S.) and Baja California (Mexico), primarily on offshore islands (Stewart et al. 1994), from December to March (Stewart and Huber 1993). ~~Males feed near the eastern Aleutian Islands and in the Gulf of Alaska, and females feed further south, south of 45°N (Stewart and Huber 1993; Le Boeuf et al. 1993).~~ Spatial segregation in foraging areas between males and females is evident from satellite tag data (Le Beouf et al. 2000). Males migrate to the Gulf of Alaska and western Aleutian Islands along the continental shelf to feed on benthic prey, while females migrate to pelagic areas in the Gulf of Alaska and the central North Pacific to feed on pelagic prey (Le Beouf et al. 2000). Adults return to land between March and August to molt, with males returning later than females. Adults return to their feeding areas again between their spring/summer molting and their winter breeding seasons.

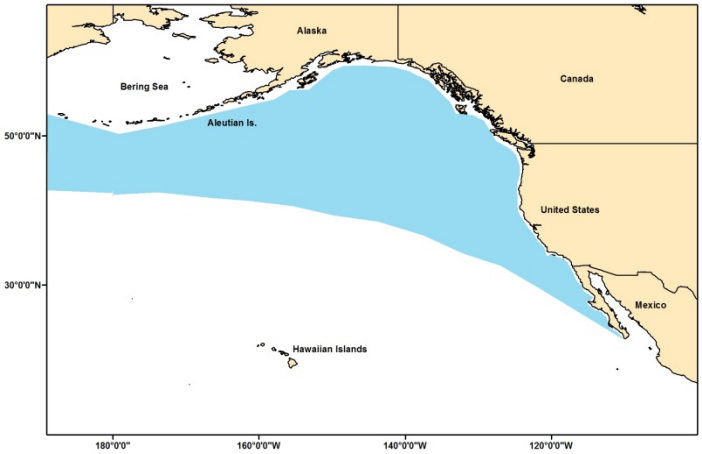


Figure 1. Pelagic range of northern elephant seals in the eastern North Pacific. Major breeding rookeries occur along the west coast of Baja California and the California coast, as described in Lowry et al. (2014).

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

POPULATION SIZE

A complete population count of elephant seals is not possible because all age classes are not ashore ~~at the same time~~ simultaneously. Elephant seal population size is ~~typically~~ estimated by counting the number of pups produced and multiplying by the inverse of the expected ratio of pups to total animals (McCann 1985). Based on counts of elephant seals at U.S. rookeries in 2010, Lowry *et al.* (2014) reported that 40,684 pups were born. Lowry *et al.* (2014) applied a multiplier of 4.4 to extrapolate from total pup counts to a population estimate of approximately 179,000 elephant seals. This multiplier is derived from life tables based on published elephant seal fecundity and survival rates, and reflects a population with approximately 23% pups (Cooper & Stewart, 1983; Le Boeuf & Reiter, 1988; Hindell, 1991; Huber et al., 1991; Reiter & Le Boeuf, 1991; Clinton & Le Boeuf, 1993; Le Boeuf et al., 1994; Pistorius & Bester, 2002; McMahon et al., 2003; Pistorius et al., 2004; Condit et al., 2014). ~~Stewart et al. (1994) used McCann's multiplier of 4.5 to extrapolate from 28,164 pups to a population estimate of 127,000 elephant seals in the~~

~~U.S. and Mexico in 1991. The multiplier of 4.5 was based on a non-growing population. Boveng (1988) and Barlow et al. (1993) suggest that a multiplier of 3.5 is more appropriate for a rapidly growing population such as the California stock of elephant seals. Based on the estimated 35,549 pups born in California in 2005 (Fig. 2) and this 3.5 multiplier, the California stock was approximately 124,000 in 2005.~~

Minimum Population Estimate

The minimum population size for northern elephant seals in 2010 can be estimated very conservatively as 81,368 ~~74,913~~seals, which is equal to twice the observed pup count (to account for the pups and their mothers) ~~plus 3,815 males and juveniles counted at the Channel Islands and central California sites in 2005 (Mark Lowry, NMFS unpubl. data)~~. ~~More sophisticated methods of estimating minimum population size could be applied if the variance of the multiplier used to estimate population size were known.~~

Current Population Trend

~~Based on trends in pup counts, northern elephant seal colonies were continuing to grow in California through 2005 (Figure 2), but appear to be stable or slowly decreasing in Mexico (Stewart et al. 1994).~~ The population is reported to have grown at 3.8% annually since 1988 (Lowry *et al*. 2014).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATE

~~Although growth rates as high as 16% per year have been documented for elephant seal rookeries in the U.S. from 1959 to 1981 (Cooper and Stewart 1983), much of this growth was supported by immigration from Mexico. The highest growth rate measured for the whole U.S./Mexico population was 8.3% between 1965 and 1977 (Cooper and Stewart 1983). A generalized logistic growth model indicates that the maximum population growth rate (R_{max}) is 11.7 percent (SE = 2.7) (Figure 3).~~ An annual growth rate of 17% for elephant seals in the U.S. from 1958 to 1987 is reported by Lowry *et al.* (2014), but some of this growth is likely due to immigration of animals from Mexico and the consequences of a small population recovering from past exploitation. From 1988 to 2010, the population is estimated to have grown 3.8% annually (Lowry et al. 2014). For this stock assessment report, we use the default maximum theoretical net productivity rate for pinnipeds, or 12% (Wade and Angliss 1997).

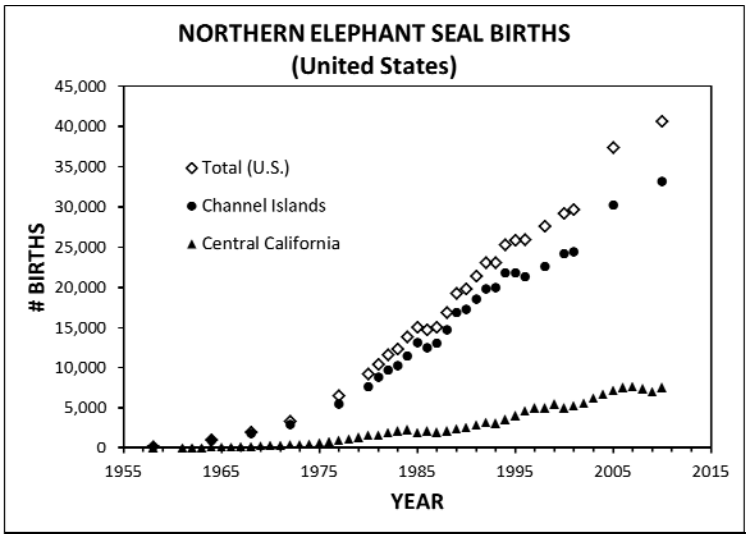


Figure 2. Estimated number of northern elephant seal births in California 1958-~~2005~~2010. Multiple independent estimates are presented for the Channel Islands 1988-91. Estimates are from Stewart et al. (1994), Lowry et al. (1996), Lowry (2002), Lowry et al. (2014), and unpublished data from Sarah Allen, Dan Crocker, Brian Hatfield, Ron Jameson, Bernie Le Boeuf, Mark Lowry, Pat Morris, Guy Oliver, Derek Lee, and William Sydeman.

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (~~74,913~~ 81,368) times one half the observed maximum net growth rate for this stock (½ of ~~11.7%~~ 12%) times a recovery factor of 1.0 (for a stock of unknown status that is increasing, Wade and Angliss 1997) resulting in a PBR of ~~4,382~~ 4,882.

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Serious Injury Guidelines

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

Fisheries Information

A summary of known commercial fishery mortality and serious injury for this stock of northern elephant seals is given in Table 1. More detailed information on these fisheries is provided in Appendix 1. ~~Stranding data reported to the California, Oregon, and Washington Marine Mammal Stranding Networks in 2000-2004 include elephant seal injuries caused by hook-and-line fisheries (two injuries) and gillnet fisheries (one injury).~~

Table 1. Summary of available information on the mortality and serious injury of northern elephant seals (California breeding stock) in commercial fisheries that might take this species (~~Carretta and Chivers 2004, Carretta et al. 2005a, 2005b, Perez 2003 , Perez 2003; Perez, in prep.; NMFS unpubl. Data~~ Carretta and Enriquez 2009, 2010, 2012a, 2012b, Carretta *et al.* 2014a). n/a indicates information is not available. Mean annual takes are based on ~~2000-2004~~ 2008-2012 data unless noted otherwise.

Fishery Name	Year(s)	Data Type	Percent Observer Coverage	Observed Mortality	Estimated Mortality (CV in parentheses)	Mean Annual Takes (CV in parentheses)
CA thresher shark/swordfish drift gillnet fishery	~~2000~~ ~~2001~~ ~~2002~~ ~~2003~~ ~~2004~~ 2008 2009 2010 2011 2012	observer data	~~22.9%~~ ~~20.4%~~ ~~22.1%~~ ~~20.2%~~ ~~20.6%~~ 13.5% 13.3% 11.9% 19.5% 18.6%	~~6~~ ~~1~~ ~~1~~ ~~1~~ ~~0~~ 0 0 0 0 0	~~26 (0.39)~~ ~~5 (0.94)~~ ~~5 (0.92)~~ ~~5 (1.00)~~ ~~0~~ 0 0 0 0 0	~~8 (0.40)~~ 0 (n/a)
CA halibut and white seabass set gillnet fishery ~~**CA angel shark/halibut and other species large mesh (>3.5") set gillnet fishery**[1]~~	~~2001[1]~~ ~~2002[1]~~ ~~2003[1]~~ ~~2004[1]~~ ~~2005[1]~~ 2008 2009 2010 2011 2012	observer data	~~0%~~ ~~0%~~ ~~0%~~ ~~0%~~ ~~0%~~ 0% 0% 12.5% 8.0% 5.5%	~~n/a~~ ~~n/a~~ ~~n/a~~ ~~n/a~~ ~~n/a~~ n/a n/a 0 0 0	~~n/a~~ ~~n/a~~ ~~n/a~~ ~~n/a~~ ~~n/a~~ n/a n/a 0 0 0	~~n/a~~ 0 (n/a)
CA small-mesh drift gillnet fishery for white seabass, yellowtail, barracuda, and tuna	2010 2011 2012	observer data	0.7% 3.3% 4.6%	0 0 0	0 0 0	0 (n/a)
WA, OR, CA domestic groundfish trawl fishery (includes at-sea hake and other limited-entry groundfish sectors)	2005 2006 2007 2008 2009	observer data	98% to 100% of tows in at-sea hake fishery Generally less than 30% of landings observed in other groundfish sectors	0 1 3 7 2	0 (n/a) 1 (n/a) 3 (n/a) 9 (n/a) 2 (n/a)	3 (n/a)

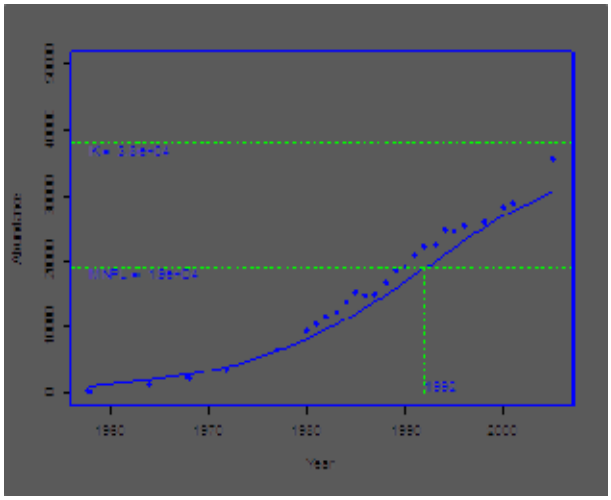
Fishery Name	Year(s)	Data Type	Percent Observer Coverage	Observed Mortality	Estimated Mortality (CV in parentheses)	Mean Annual Takes (CV in parentheses)
Unknown gillnet fishery	2008-2012	stranding	n/a	1	1 (n/a)	≥1
~~**WA, OR, CA domestic groundfish trawl (At sea processing Pacific whiting fishery only)**~~	~~2000~~ ~~2001~~ ~~2002~~ ~~2003~~ ~~2004~~	~~observer data~~	~~80.6%~~ ~~96.2%~~ ~~100%~~ ~~100%~~ ~~100%~~	~~1~~ ~~0~~ ~~0~~ ~~0~~ ~~3~~	~~1 (n/a)~~ ~~0 (n/a)~~ ~~0 (n/a)~~ ~~0 (n/a)~~ ~~3 (n/a)~~	~~0.8 (n/a)~~
~~**WA, OR, CA domestic groundfish trawl fishery (bottom trawl)**~~	~~2000-2004~~	~~observer~~	~~n/a~~	~~0~~	~~0~~	~~0 (n/a)~~
Total annual takes						~~> 8.8 (0.40)~~ ≥4.0 (n/a)

~~[1] The most recent observer data for the halibut set gillnet fishery is from 2000 in Monterey Bay only and there has not been a fishery-wide observer program since 1990-94. There are no current estimates of mortality for this fishery, as this would require assuming that current kill rates are comparable to kill rates observed between 1990-94 and extrapolation of mortality estimates using current estimates of fishing effort.~~

Although all of the mortality in Table 1 occurred in U.S. waters, some may be of seals from Mexico's breeding population that are migrating through U.S. waters. ~~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). The number of set gillnet vessels in this part of Mexico is unknown. The take of northern elephant seals in other North Pacific fisheries that have been monitored appears to be trivial (Barlow et al. 1993, 1994).~~

Other Mortality

~~Stranding databases for California, Oregon, and Washington states that are maintained by the National Marine Fisheries Service contain the following records of human-related elephant seal mortality and injuries in 2000-2004: (1) boat collision (three deaths), (2) power plant entrainment (one death), (3) shootings (four deaths) and (4) entanglement in marine debris (10 injuries). This results in a minimum annual average of 1.6 non-fishery related deaths for 2000-2004.~~ For the period 2008-2012, mortality and serious injuries from the following non-commercial fishery sources



~~**Figure 3.** Generalized logistic growth model of elephant seal pup counts, 1958-2005.~~

were documented: shootings (9); marine debris entanglement (7); hook and line fisheries (3); power plant entrainment (2); research-related (1); tar/oil (1); and vessel strike (1) (Carretta *et al*. 2014b). These non-commerical fishery sources of mortality and serious injury total 24 animals, or an average of 4.8 elephant seals annually (Carretta *et al*. 2014b).

STATUS OF STOCK

~~A generalized logistic growth model of pup counts indicated that the population reached its Maximum Net Productivity Level (MNPL) of 19,000 pups in 1992, but has not reached carrying capacity (K) at 38,200 pups per year (z = 1, Rmax = 0.117, n0 = 1,000, SE = 3,376, AICc = 500.3) (Figure 3). They~~ Northern elephant seals are not listed as "endangered" or "threatened" under the Endangered Species Act nor designated as "depleted" under the MMPA. Because their annual human-caused mortality (≥ ~~8.8~~ 8.8) is much less than the calculated PBR for this stock (~~4,382~~ 4,882), ~~they would~~ northern elephant seals are not ~~be~~ considered a "strategic" stock under the MMPA. The average rate of incidental fishery mortality for this stock over the last five years (~~>8.8~~ ≥ 4.0) also appears to be less than 10% of the calculated PBR; therefore, the total fishery mortality appears to be insignificant and approaching a zero mortality and serious injury rate. The population growth rate between 1958 and 1987 was 17% annually (Lowry *et al*. 2014). From 1988 to 2010, the population grew at an annual rate of 3.8% (Lowry *et al*. 2014). The population continues to grow, with most births occurring at southern California rookeries (Lowry et al. 2014). No estimate of carrying capacity is available for this population and the population status relative to OSP is unknown. ~~This annual rate of fishery mortality is negatively biased because it excludes mortality that likely occurs in the unobserved set gillnet fishery for halibut and angel shark, where average annual mortality was estimated at approximately 60 animals annually during the period 1996-2000. The population is continuing to grow and fishery mortality is relatively constant.~~ There are no known habitat issues that are of ~~particular~~ concern for this stock. However, expanding pinniped populations in general have resulted in increased human-caused serious injury and mortality, due to shootings, entrainment in power plants, interactions with recreational hook and line fisheries, separation of mothers and pups due to human disturbance, dog bites, and vessel and vehicle strikes (Carretta et al. 2014b).

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HAWAIIAN MONK SEAL (*Monachus 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). ~~On average, 10-15% of the seals migrate among the NWHI subpopulations (Johnson and Kridler 1983; Harting 2002). Thus, the NWHI subpopulations are not isolated, though different island~~ Though monk seal subpopulations ~~have~~ exhibit~~ed~~ considerable demographic independence, they are connected by animal movement throughout the species' range (Johanos *et al*. 2013). ~~Observed interchange of individuals among the NWHI and MHI regions is uncommon, but genetic~~ Genetic stock structure analysis (Schultz *et al*. 2011) further supports management of the species as a single stock.

POPULATION SIZE

The best estimate of the total population size is 1,153 ~~1,209~~. This estimate is the sum of estimated abundance at the six main Northwestern Hawaiian Islands subpopulations, an extrapolation of counts at Necker and Nihoa Islands, and an estimate of minimum abundance in the main Hawaiian Islands. In 2012, there was a marked reduction in field effort in the NWHI due to reduced program funding. Researchers were in the field in the NWHI from 30 to 44 days at each field site; a reduction of some 50% to 80% compared to typical recent years. The short field season resulted in greater uncertainty in population abundance and trends. The number of individual seals identified was used as the population estimate at NWHI sites where total enumeration was achieved, according to the criteria established by Baker *et al*. (2006). Where total enumeration was not achieved, capture-recapture estimates from Program CAPTURE were used (Baker 2004; Otis *et al*. 1978, Rexstad & Burnham 1991, White *et al*. 1982). When no reliable estimator was obtainable in Program CAPTURE (i.e., the model selection criterion was < 0.75, following Otis *et al*. 1978), the total number of seals identified was the best available estimate. ~~Finally, s~~ Sometimes capture-recapture estimates are less than the known minimum abundance (Baker 2004), and in these cases, the total number of seals actually identified was used. In ~~2011~~ 2012, total enumeration was not achieved ~~at~~ for any subpopulation ~~Laysan Island, Lisianski Island, Pearl and Hermes Reef and Kure Atoll, based on analysis of discovery curves~~. Capture-recature estimates were available for French Frigate Shoals, Lisianski Island and Pearl and Hermes Reef. Minimum abundance was used for Laysan Island, Midway Atoll and Kure Atoll. ~~French Frigate Shoals and Midway Atoll. Thus, a~~ Abundance at the six main NWHI subpopulations was estimated to be ~~909~~ 862 (including ~~141~~ 111 pups). Counts at Necker and Nihoa Islands are conducted from zero to a few times in a single year. Abundance is estimated by correcting the mean of all beach counts accrued over the past five years. The mean (±SD) of all counts (excluding pups) conducted between ~~2007 and 2011~~ 2008 and 2012 was ~~17.0 ± 5.4~~ 16.1 ± 5.8 at Necker Island and ~~31.5 ± 7.2~~ 32.2 ± 6.4 at Nihoa Island. The relationship between mean counts and total abundance at the reproductive sites indicates that total abundance can be estimated by multiplying the mean count by a correction factor of 2.89 (NMFS unpubl. data). Resulting estimates (plus the average number of pups known to have been born during ~~2006-2010~~ 2008-2012 are ~~52.3 ± 15.6~~ 49.9 ± 16.8 at Necker Island and ~~101.6 ± 20.8~~ 103.1 ± 18.5 at Nihoa Island.

Complete, systematic surveys for monk seals in the MHI were conducted in 2000 and 2001 (Baker and Johanos 2004). NMFS continues to collect information on seal sightings reported by a variety of sources, including a volunteer network, ~~reports from~~ the public, and directed NMFS observation effort. The total number of individually identifiable seals documented in ~~2011~~ 2012 was ~~146~~ 138, the current best minimum abundance estimate for the MHI.

Minimum Population Estimate

The total number of seals (~~909~~ 853) identified at the six main NWHI reproductive sites is the best estimate of minimum population size at those sites. Minimum population sizes for Necker and Nihoa Islands (based on the formula provided by Wade and Angliss (1997)) are ~~41 and 86~~ 38 and 89, respectively. The minimum abundance estimate for the main Hawaiian Islands in 2011 is ~~146~~ 138 seals. The minimum population size for the entire stock (species) is the sum of these estimates, or ~~1,182~~ 1,118 seals.

Current Population Trend

Current population trend is based solely on the six NWHI subpopulations because these sites have historically comprised virtually the entire species, while information on the remaining smaller seal aggregations has been inadequate to reliably evaluate abundance or trends. The total of mean non-pup beach counts at the six main reproductive NWHI subpopulations in ~~2011~~ 2012 is 69% lower than in 1958. The trend in total abundance at the six main NWHI subpopulations estimated as described above is shown in Figure 1. A log-linear regression of estimated abundance on year for the past 10 years (~~2002-2011~~ 2003-2012) estimates that abundance declined ~~-3.4% yr^{-1} (95% CI = -4.3% to -2.5% yr^{-1})~~ -3.3% yr^{-1} (95% CI = -4.2% to -2.3% yr^{-1}). The MHI monk seal population appears to be increasing with an intrinsic population growth rate estimated at 6.5% per year based on simulation modeling (Baker *et al*. 2011). Likewise, sporadic beach counts at Necker and especially Nihoa Islands, suggest positive growth. While these sites have historically comprised a small fraction of the total species abundance, the decline of the six main NWHI subpopulations, coupled with growth at Necker, Nihoa and the MHI may mean that these latter three sites now substantially influence the total abundance trend. The MHI, Necker and Nihoa Islands estimates, uncertain as they are, comprised 25% of the stock's estimated total abundance in ~~2011~~2012. Unfortunately, because of a lack reliable abundance estimates for these areas, their influence cannot currently be determined. NMFS is experimenting with remote camera systems that may improve data collection at Necker and Nihoa Islands.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Trends in abundance vary considerably among subpopulations. 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% yr^{-1} 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.

POTENTIAL BIOLOGICAL REMOVAL

Potential biological removal (PBR) is designed to allow stocks to recover to, or remain above, the maximum net productivity level (MNPL) (Wade 1998). An underlying assumption in the application of the PBR equation is that marine mammal stocks exhibit certain dynamics. Specifically, it is assumed that a depleted stock will naturally grow toward OSP (Optimum Sustainable Population), and that some surplus growth could be removed while still allowing recovery. The Hawaiian monk seal population is far below historical levels and has on average, declined ~~3.4%~~ 3.3% a year since 2002. Thus, the stock's dynamics do not conform to the underlying model for calculating PBR such that PBR for the Hawaiian monk seal is undetermined.

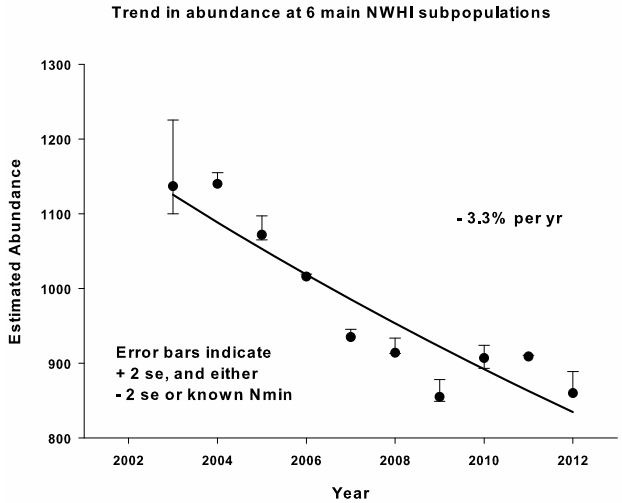

Figure 1. Trend in abundance of monk seals at the six main Northwestern Hawaiian Islands subpopulations, based on a combination of total enumeration and capture–recapture estimates. Error bars indicate ±2 s.e. (from variances of capture-recapture estimates). Fitted log-linear regression line is shown.

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

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~~ distinguish~~ing~~ serious from non-serious injury (Angliss and DeMaster 1998, Andersen *et al*. 2008, NOAA 2012). NMFS defines serious injury as an "*injury that is more likely than not to result in mortality*". ~~Injury determinations for stock assessments revised in 2013 or later incorporate the new serious injury guidelines, based on the most recent 5-year period for which data are available.~~

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 a relatively new and alarming trend (Table 1).

Table 1. Intentional and potentially intentional killings of Hawaiian monk seals in the MHI since 2009.

Year	Age/sex	Island	Cause of Death	Comments
2009	Subadult male	Kauai	Gunshot wound	
	Adult female	Kauai	Gunshot wound	Pregnant
	Adult male	Molokai	Gunshot wound	
2010	Juvenile female	Kauai	Multiple skull fractures, blunt force trauma	Intent unconfirmed
2011	Adult male	Molokai	Skull fracture, blunt force trauma	Intent unconfirmed
	Juvenile female	Molokai	Skull fracture, blunt force trauma	Intent unconfirmed
2012	Juvenile male	Kauai	Gunshot wound	
	Subadult male	Kauai	Skull fracture	Intent unconfirmed

~~In 2009, three seals (including a pregnant female) were shot and killed in the MHI (Baker *et al*. 2010). In 2010, a juvenile female seal was found dead died on Kauai due to multiple skull fractures caused by blunt force trauma. Whether this was an intentional killing or an accidental occurrence (e.g., boat strike) is not known. In 2011, two seals were found on the same general area of Molokai dead with skull fractures from blunt force trauma.~~ 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. ~~Nearshore gillnets have become a more common source of mortality recently. Three seals have been 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. Numerous cases of seals with embedded hooks are observed each year in the MHI.~~ In 2012 ~~2011, 9~~ 16 seals were observed hooked, four of which died as a result of ingesting hooks~~ none of which constituted serious injuries~~. The remaining 12 were non-serious hookings, although 5 of these would have been deemed serious had they not been mitigated by capture and hook removal. Several incidents involved hooks used to catch ulua (jacks, *Caranx* spp.). Nearshore gillnets became a more common source of mortality in the 2000's, 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 been documented since 2010. Most reported hookings and gillnet entanglements have occurred since 2000 (NMFS unpubl. data). 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 1). 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). Recent 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 2 ~~1~~. Summary of mortality, ~~and~~ 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.

Fishery Name	Year	Data Type	% Obs. coverage	Observed/Reported Mortality/Serious Injury	Estimated Mortality/ Serious Injury	Non-serious (Mi~~tig~~ated serious)[1]	Mean Takes (CV)
Pelagic Longline	~~2007~~ 2008 2009 2010 2011 2012	observer observer observer observer observer	~~20.1% & 100%[2]~~ 21.7% & 100%[2] 20.6% & 100%[2] 21.1% & 100%[2] 20.3% & 100%[2] 20.4% & 100%[2]	0 0 0 0 0	0 0 0 0 0	0 0 0 0 0	0 (0)
MHI Bottomfish[3]	~~2007~~ 2008 2009 2010 2011 2012	Incidental observations of seals	none	0 0 0 0 0	n/a	0 0 0 0 0	n/a
Nearshore[4]	~~2007~~ 2008 2009 2010 2011 2012	Incidental observations of seals	none	~~2~~ 0 0 1 0 4	n/a	9(3) 12(3) 11(2) 9 (3) 12 (5)	≥~~0.6~~ 1.0
Minimum total annual takes							≥~~0.6~~ 1.0

There are no fisheries operating in or near the NWHI. In the past, interactions between the Hawaii-based domestic pelagic longline fishery and monk seals were documented (Nitta and Henderson 1993). This fishery targets swordfish and tunas and does not compete with Hawaiian monk seals for prey. In October 1991, in response to 13 unusual seal wounds thought to have resulted from interactions with this fishery, NMFS established a Protected Species Zone extending 50 nautical miles around the NWHI and the corridors between the islands. Subsequently, no additional monk seal interactions with the swordfish or tuna components of the longline fishery have been observed.

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 purpose-designed 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). Indirect interactions

[1] Total non-serious injuries documented. In parentheses, number of injuries that would have been deemed serious had they not been mitigated (e.g., by de-hooking or disentangling.

[2] Observer coverage for deep and shallow-set components of the fishery, respectively.

[3] 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.

[4] Includes 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.

(i.e., involving competition for prey or consumption of discards) remain a topic of ongoing investigation.

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). A total of ~~323~~ 331 cases of seals entangled in fishing gear or other debris have been observed from 1982 to 2012 (Henderson 2001; NMFS, unpubl. data). ~~,including eight~~ Nine documented deaths resulted from entanglement in marine debris, including a pup at Midway Atoll in 2012 (Henderson 1990, 2001; NMFS, unpubl. data). 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 n*et al*one accounted for 88% of the debris by frequency (Donohue *et al*. 2001). Yet, 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

In the past 10 years (2003-2012 ~~2002-2011~~) two monk seals died during enhancement activities (in 2005 and 2006) and one died during research in 2007 (NMFS unpubl. data).

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). While disease effects on monk seal demographic trends are uncertain, there is concern that diseases of livestock, feral animals, pets or humans could be transferred to naïve monk seals in the MHI and potentially spread to the core population in the NWHI. In 2003 and 2004, two deaths of free-ranging monk seals were attributable to diseases not previously found in the species: leptospirosis and toxoplasmosis (R. Braun, pers. comm.). *Leptospira* bacteria are found in many of Hawaii's streams and estuaries and are associated with livestock and rodents. Cats, domestic and feral, are a common source of toxoplasma.

Habitat Issues

Poor juvenile survival rates and variability in the relationship between weaning size and survival suggest that prey availability is likely limiting recovery of NWHI monk seals (Baker and Thompson 2007, Baker *et al*. 2007, Baker 2008). Multiple strategies for improving juvenile survival are being considered and will be developed through an experimental approach in coming years (Baker and Littnan 2008, Baker *et al*. 2013). NMFS has produced a draft Programmatic Environmental Impact Statement on current and future anticipated research and enhancement activities[1]. A major habitat issue involves loss of terrestrial habitat at French Frigate Shoals, where 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. 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 abundance is increasing in the main Hawaiian Islands (Baker *et al*. 2011). Further, the excellent condition of pups weaned on these islands suggests that there may be 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

[1] http://www.nmfs.noaa.gov/pr/permits/eis/hawaiianmonksealeis.htm

and Johanos 2004). If the monk seal population continues to expand in the MHI, it may bode well for the species' recovery and long-term persistence. In contrast, 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, so that the potential impact of disturbance in the MHI is great. Intentional killing of seals (noted above) poses a very serious new concern. Also, the same fishing pressure that may have reduced the monk seal's competitors, is a source of injury and mortality. Finally, vessel traffic in the populated islands carries the potential for collision with seals and impacts from oil spills. The causes of two recent non-serious injuries (in 2010 and 2011) to seals were attributed to boat propellers. Thus, issues surrounding monk seals in the main Hawaiian Islands will likely become an increasing focus for management and recovery of this species.

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. The species is well below its optimum sustainable population (OSP) and has not recovered from past declines. Therefore, the Hawaiian monk seal is a strategic stock. Annual human-caused mortality for the most recent 5-year period (2008-2012~~ 2007-2011~~) was at least 2.6 ~~1.8~~ animals, including fishery-~~caused~~related mortality in nearshore gillnets and hook-and-line gear (>=1 ~~0.6~~/ yr, Table 2~~1~~), shooting-related deaths (>=0.8 ~~0.6~~ / yr), and blunt-force trauma deaths of unknown origin (>=0.8 ~~0.6~~ / yr, Table 1).

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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 waters to polar regions, 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).

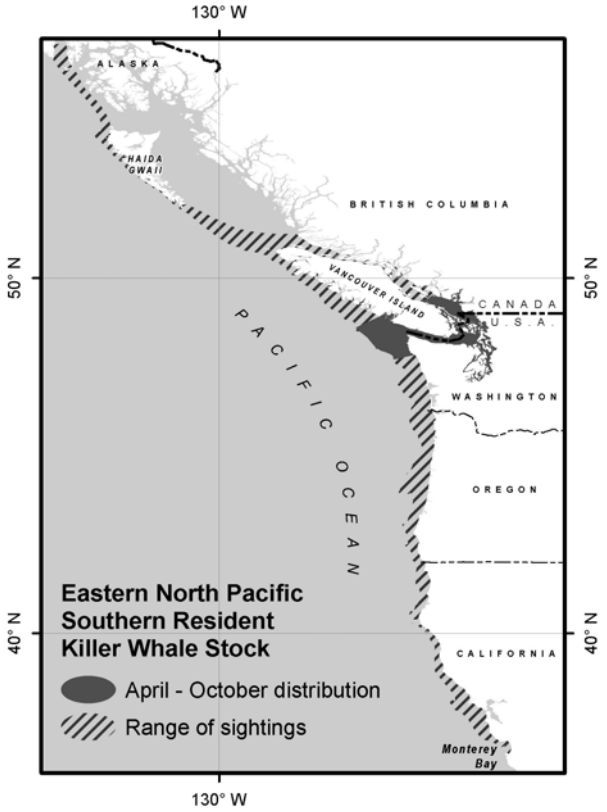

Figure 1. Approximate April - October distribution of the Eastern North Pacific Southern Resident killer whale stock (shaded area) and range of sightings (diagonal lines).

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 (N. Black, pers. comm., K. Balcomb, pers. comm.) 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 (NWFSC, unpubl. data).

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 Eastern North Pacific 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 ~~85~~ 82 whales in ~~2012~~ 2013 (Fig. 2; Ford et al. 2000; Center for Whale Research, unpubl. data). 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 (J. Ford, pers. comm.). 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 ~~2011~~ 2012 through 1 July ~~2012~~ 2013 includes ~~two~~ one new ~~calves~~ calf and the deaths of ~~a two~~ three post-reproductive adult females~~, a juvenile female~~, and a young adult male. ~~It~~ This does not include a post-reproductive female and a young adult male that were missing in fall 2013 ~~a calf observed in December 2011 that did not survive six months~~ (Center for Whale Research, unpubl. data).

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 ~~85~~ 82 animals.

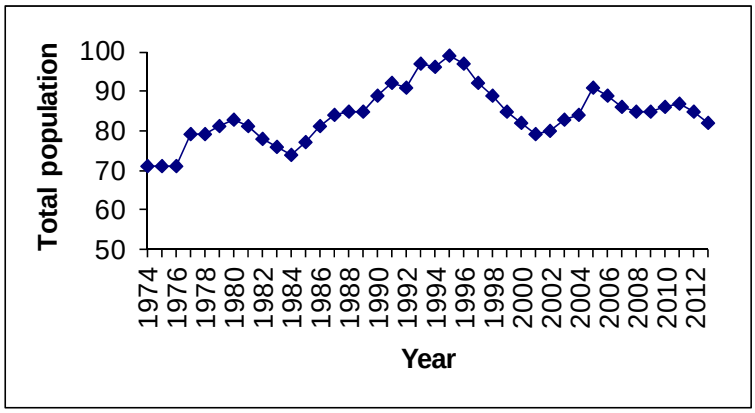

Figure 2. Population of Eastern North Pacific Southern Resident stock of killer whales, 1974-~~2012~~ 2013. Each year's count includes animals first seen and first missed; a whale is considered first missed the year after it was last seen alive (Ford et al. 2000; Center for Whale Research, unpubl. data).

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 1993 the Southern Resident stock increased approximately 35%, from 71 to 96 individuals (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 fluctuated and currently stands at ~~85~~ 82 animals as of the ~~2012~~ 2013 census (Ford et al. 2000; Center for Whale Research, unpubl. data).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate is currently unavailable for this stock of killer whales. Studies of 'resident' killer whale pods in British Columbia and Washington waters resulted in estimated population growth rates of 2.92% and 2.54% over the period from 1973 to 1987 (Olesiuk et al. 1990, Brault and Caswell 1993). For southern resident killer whales, estimates of the population growth rate have been made during the three periods when the population has been documented increasing since monitoring began in 1974. From 1974 to 1980 the population increased at a rate of 2.6%/year, 2.3%/year from 1985 to 1996, and 3.6%/year from 2002 to 2005 (Center for Whale Research, unpubl. data). A recent analysis of the long-term trend of southern resident population growth (1979-2011) indicated that there was a 5% probability of the maximum growth (R_{max}) exceeding 2.8% and a 1% chance of it exceeding 3.2% (Ward 2012). Hence, R_{max} is estimated to be 3.2% for southern resident killer whales and this value will be used for this stock.

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (~~85~~ 82) times one-half the default maximum net growth rate for cetaceans (½ of 3.2%) 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.

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~~ distinguish~~ing~~ serious from non-serious injury (Angliss and DeMaster 1998, Andersen et al. 2008, NOAA 2012). NMFS defines serious injury as an "*injury that is more likely than not to result in mortality*". ~~Injury determinations for stock assessments revised in 2013 or later incorporate the new serious injury guidelines, based on the most recent 5-year period for which data are available.~~

Fisheries Information

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 (NMFS NW Region, unpublished data). 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, unpublished data). A fishery experiment with 100% observer coverage and acoustic alarms on all set gillnets was conducted in 2008 and 2011. No killer whale bycatch was documented (Makah Fisheries Management, unpublished data).

An additional source of information on the number of killer whales killed or injured 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

~~According to Northwest Marine Mammal Stranding Network records, maintained by the NMFS Northwest Region, n~~ No human-caused killer whale mortality or serious injuries were reported from non-fisheries sources in ~~2007-2011~~2008-2012 (Carretta et al. ~~2013~~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 level of non-fishery human-caused mortality for this stock over the past five years (~~2007-2011~~2008-2012) is zero animals per year.

STATUS OF STOCK

Southern Resident killer whales were listed as endangered under the ESA in 2005. 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 are formally listed as "endangered" under the ESA and consequently the stock is automatically considered as a "strategic" stock under the MMPA. This stock was considered "depleted" prior to its 2005 listing under the ESA.

Habitat Issues

Several of the potential risk factors identified for this population have habitat implications. The summer range of this population, the inland waters of Washington and British Columbia, is the home to a large commercial whale watch industry as well as high levels of recreational boating and commercial shipping. There continues to be concern about potential for masking effects by noise generated from these activities on the whales' communication and foraging. In 2011 vessel approach regulations were implemented to restrict vessel from approaching closer than 200m. This population appears to be Chinook salmon specialists (Ford and Ellis 2006, Hanson et al. 2010), although other species, particularly chum, appear to be important in the fall (NWFSC unpubl. data). 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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Revised ~~12/04/2012~~ 10/22/2014

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

Sperm whales are distributed across the entire North Pacific and into the southern Bering Sea in summer, but the majority are thought to be south of 40°N in winter (Rice 1974; Rice 1989; Gosho et al. 1984; Miyashita et al. 1995). The International Whaling Commission (IWC) historically divided the North Pacific into two management regions (Donovan 1991) defined by a zig-zag line which starts at 150°W at the equator, is 160°W between 40-50°N, and ends up at 180°W north of 50°N; however, the IWC has not reviewed this stock boundary recently (Donovan 1991). Sperm whales are found year-round in California waters (Dohl et al. 1983; Barlow 1995; Forney et al. 1995), but they reach peak abundance from April through mid-June and from the end of August through mid-November (Rice 1974). Sperm whales are seen off Washington and Oregon in every season except winter (Green et al. 1992). Of 176 sperm whales that were marked with Discovery tags off southern California in winter 1962-70, only three were recovered by whalers: one off northern California in June, one off Washington in June, and another far off British Columbia in April (Rice 1974). Recent summer/fall surveys in the eastern tropical Pacific (Wade and Gerrodette 1993) show that although sperm whales are widely distributed in the tropics, their relative abundance declines westward towards the middle of the tropical Pacific (near the IWC stock boundary at 150°W) and declines northward towards the tip of Baja California. Sperm whale population structure in the eastern tropical Pacific is unknown, but the only photographic matches of known individuals from this area have been between the Galapagos Islands and coastal waters of South America (Dufault and Whitehead 1995) and between the Galapagos Islands and the southern Gulf of California (Jaquet et al. 2003), suggesting that eastern tropical Pacific animals constitute a distinct stock. No apparent hiatus in distribution between the U.S. EEZ off California and areas farther west, out to Hawaii were found during a survey designed specifically to investigate stock structure and abundance of sperm whales in the northeastern temperate Pacific (Barlow and Taylor 2005). Sperm whales in the California Current have been identified as demographically independent from animals in Hawaii and the Eastern Tropical Pacific, based on genetic analyses of single-nucleotide polymorphisms (SNPs), microsatellites, and mtDNA (Mesnick *et al*. 2011). For the Marine Mammal Protection Act (MMPA) stock assessment reports, sperm whales within the

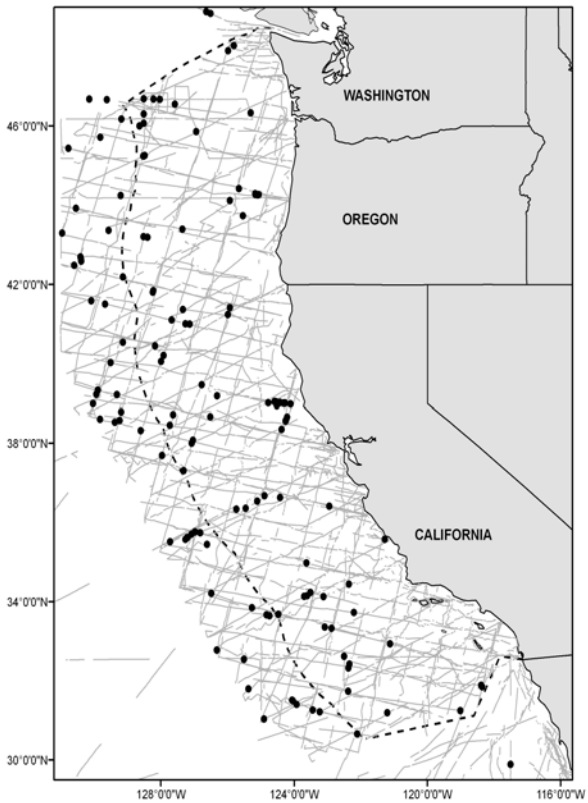

Figure 1. Sperm whale sighting locations from shipboard surveys off California, Oregon, and Washington, 1991-2008. Dashed line represents the U.S. EEZ, thin lines indicate completed transect effort of all surveys combined. See Appendix 2 for data sources and information on timing and location of survey effort.

Pacific U.S. EEZ are divided into three discrete, non-contiguous areas: 1) California, Oregon and Washington waters (this report), 2) waters around Hawaii, and 3) Alaska waters.

POPULATION SIZE

~~Barlow and Taylor (2001) estimated 1,407 (CV=0.39) sperm whales in California, Oregon, and Washington waters during summer/fall based on pooled 1993 and 1996 ship line transect surveys within 300 nmi of the coast. Barlow and Forney (2007) estimated 2,593 (CV= 0.30) sperm whales from a survey of the same area in 2001. A 2005 survey of this area resulted in an abundance estimate of 3,140 (CV=0.40) whales, which is corrected for diving animals not seen during surveys (Forney 2007). The most recent ship survey of the same area in 2008 resulted in an estimate of only 300 (CV = 0.51) sperm whales (Barlow 2010). The 2008 estimate is lower than all previous estimates within this region and may be due to interannual variability of sperm whale distribution. The most recent estimate of abundance for this stock is the geometric mean of the 2005 and 2008 summer/autumn ship survey estimates, or 971 (CV = 0.31) sperm whales. A combined visual and acoustic line transect survey conducted in the eastern temperate North Pacific in spring 1997 resulted in estimates of 26,300 (CV=0.81) sperm whales based on visual sightings, and 32,100 (CV=0.36) based on acoustic detections and visual group size estimates (Barlow and Taylor 2005). However, it is not known whether any or all of these animals routinely enter the U.S. EEZ. In the eastern tropical Pacific, the abundance of sperm whales has been estimated as 22,700 (95% C.I.=14,800-34,600; Wade and Gerrodette 1993), but this does not include areas where sperm whales are taken by drift gillnet fisheries in the U.S. EEZ and there is no evidence of sperm whale movements from the eastern tropical Pacific to the U.S. EEZ. Barlow and Taylor (2001) also estimated 1,640 (CV=0.33) sperm whales off the west coast of Baja California, but again there is no evidence for interchange between these animals and those off California, Oregon and Washington.~~

~~Large populations of sperm whales exist in waters several thousand miles west and south of California, Oregon, and Washington waters covered by this report; however, there is no evidence of sperm whale movements into this region from either the west or south and genetic data suggest that mixing to the west is unlikely. There is limited evidence of sperm whale movement from California to northern areas off British Columbia, but there are no abundance estimates for the latter area. The most precise and recent estimate of sperm whale abundance for this stock is therefore 971 (CV = 0.31) animals from the ship surveys conducted in 2005 (Forney 2007) and 2008 (Barlow 2010).~~ Previous estimates of sperm whale abundance from 2005 (3,140, CV=0.40, Forney 2007) and 2008 (300, CV=0.51, Barlow 2010) show a ten-fold difference that cannot be attributed to human-caused or natural population declines and likely reflect inter-annual variability in the movement of animals into and out of the study area. New estimates of sperm whale abundance in California, Oregon, and Washington waters out to 300 nmi are available from a trend-model analysis of line-transect data collected from six surveys conducted from 1991 to 2008 (Moore and Barlow 2014). Abundance trend models incorporate information from the entire 1991-2008 time series to obtain each annual abundance estimate, yielding estimates with less inter-annual variability. The trend model also uses improved estimates of group size and trackline detection probability (Moore and Barlow 2014). The new estimates are from methods similar to those previously used to estimate abundance trends for fin whales (Moore and Barlow 2011) and beaked whales in the California Current (Moore and Barlow 2013). Sperm whale abundance estimates based on the trend-model ranged between 2,000 and 3,000 animals for the 1991-2008 time series (Moore and Barlow 2014). The best estimate of sperm whale abundance in the California Current is the trend-based estimate corresponding to the most recent survey (2008), or 2,106 animals (CV=0.58). This estimate is corrected for diving animals not seen during surveys.

Minimum Population Estimate

The minimum population estimate for sperm whales is taken as the lower 20th percentile of the posterior distribution of abundance estimated from 2008 or 1,332 whales (Moore and Barlow 2014).~~The minimum population estimate for sperm whales is taken as the lower 20th percentile of the log-normal distribution of abundance estimated from the 2005-2008 summer/fall ship surveys off California, Oregon and Washington (Barlow and Forney 2007; Forney 2007) or approximately 751.~~

Current Population Trend

~~Sperm whale abundance varied off California between 1979/80 and 1991 (Barlow 1994) and between 1991 and 2008 (Barlow and Forney 2007). The most recent estimate from 2008 is the lowest to date, in sharp contrast to the highest abundance estimates obtained from 2001 and 2005 surveys. There is~~

~~no reason to believe that the population has declined; the most recent survey estimate likely reflects interannual variability in the study area. To date, there has not been a statistical analysis to detect trends in abundance. Although the population in the eastern North Pacific is expected to have grown since large-scale pelagic whaling stopped in 1980, the possible effects of large unreported catches are unknown (Yablokov 1994) and ongoing incidental ship strikes and gillnet mortality make this uncertain.~~ Moore and Barlow (2014) report that sperm whale abundance appeared stable from 1991 to 2008 (Figure 2), but that reliable conclusions on population trends could not be made because the precision of estimated growth rates was poor. However, they also reported that trends in the detection of single animals (presumably large, solitary males) apparently doubled over this time period. The authors could not determine if the apparent increase in sightings of single animals reflected an increase in the number of adult male sperm whales in the population or merely increased use of the U.S. west coast waters by adult males in recent years.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

There are no published estimates of the growth rate for any sperm whale population (Best 1993).

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (~~751~~ 1,332) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.1 (for an endangered stock with N_{min} <1,500; Taylor et al. 2003), resulting in a PBR of ~~1.5~~ 2.7.

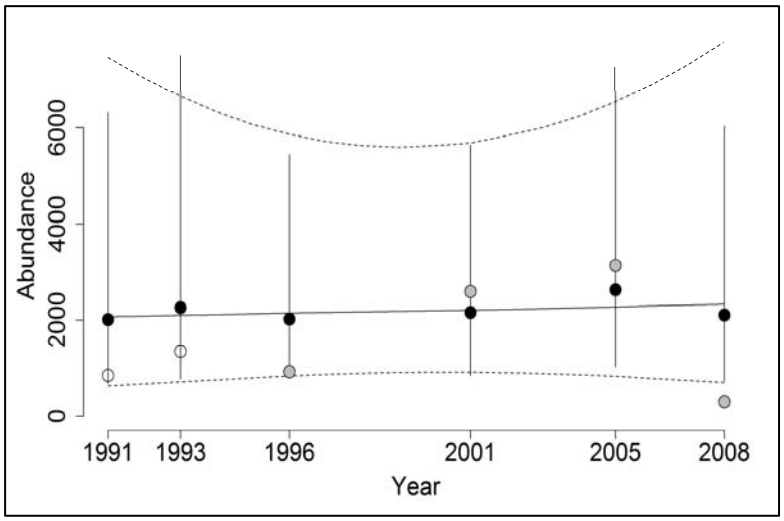

Figure 2. Trend-based estimates of sperm whale abundance in the California Current, 1991-2008 (Moore and Barlow 2014). Abundance estimates (posterior medians [●] and 95% CRIs) from the trend model, with fitted trend line and 95% CRIs for trend. For comparison, open and gray circles depict earlier published estimates from Barlow and Forney (2007) and Barlow (2010), with those for 1991 and 1993 [○] being for a smaller surveyed area.

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 distinguis 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".

Fishery Information

The fishery most likely to directly take sperm whales from this stock is the California thresher shark/swordfish drift gillnet fishery ~~California drift gillnet swordfish fishery~~ (Julian and Beeson 1998, Carretta and Enriquez 2012). Observed serious injury and mortality has been rarely observed in the gillnet fishery (10 animals during ~8,500 observed sets between 1990 and 2014). Given the historic long-term average observer coverage of ~15% for this fishery (Carretta and Barlow 2011), annual estimates of bycatch will always be either zero (if no sperm bycatch is observed) or at least 7 (if ≥ 1 observed), for estimates made using within-year ratio methods [*e.g.*, estimated bycatch = observed bycatch/percent observer coverage]. If the true average annual mortality and serious injury is > 0, but less than a few animals per year, and if observer coverage generally remains low, then multiple years of data need to be pooled to for unbiased estimation of a mean annual bycatch rates (Carretta and Moore, 2014). Pooling more years reduces bias (estimates of mean annual bycatch approaches the true rate) and provides increased precision of bycatch estimates to better estimate long-term annual mortality and serious injury. Most

marine mammal stock assessment reports utilize a 5-year time period for pooling bycatch estimates (NMFS 2005), but in the case of rare events, this 5-year time frame will yield biased estimates (systematic over- or underestimation of true bycatch) with insufficient precision (Carretta and Moore 2014, Moore and Merrick 2011). Since 2001, the drift gillnet fishery has been subject to a time/area closure that restricts most fishing to south of Point Conception, California, in waters generally shallower than 2,000 m, where bycatch risk to sperm whales is lower. The post-2000 time period best represents the current spatial state of the fishery and is used to calculate mean annual bycatch for sperm whales. Between 2001 and 2013, two sperm whales (one death and one serious injury in the same set) were entangled during 2,392 observed sets, resulting in a mean bycatch rate of 0.84 per 1,000 sets. Annual bycatch estimates for the 12-year period of 2001-2012 are presented for the drift gillnet fishery in Table 1 and are based on previously published estimates (Carretta et al. 2004, 2014a, Carretta and Chivers 2004, Carretta et al. 2005, Carretta and Enriquez 2006, 2007, 2009a, 2009b, 2010, 2012). ~~A summary of known fishery mortality and injury for this stock of sperm whales from 2006-2010 is given in Table 1.~~

Although acoustic pingers are known to reduce the entanglement of cetaceans in the California drift gillnet swordfish fishery (Barlow and Cameron 2003, Carretta et al. 2008, Carretta and Barlow 2011), it is unknown whether pingers have any effect on sperm whale entanglement in this fishery. Sperm whales have been observed entangled 10 times in over 8,000 observed drift gillnet sets since 1990 (Carretta and Enriquez 2012). Six entanglements occurred prior to pinger use in this fishery. Two entanglements (1996 and 1998) occurred in sets that did not use a full complement of pingers, and two animals were entangled in 2010 in a single net where a full complement of 40 pingers was used (Carretta and Enriquez 2012).

Other fisheries may injure or kill sperm whales, in the form of entanglement or ingestion of marine debris. Three separate sperm whale strandings in 2008 showed evidence of fishery interactions (Jacobsen et al. 2010; NMFS, unpublished stranding data). Two whales died from gastric impaction as a result of ingesting multiple types of floating polyethylene netting (Jacobsen *et al*. 2010). The variability in size and age of the ingested net material suggests that it was ingested as surface debris and was not the result of fishery depredation (Jacobsen *et al*. 2010). Net types recovered from the whales' stomachs included portions of gillnet, bait nets, and fish/shrimp trawl nets. A third whale in 2008 showed evidence of entanglement scars (NMFS, unpublished stranding data). Two sperm whales also died in 2004 as a result of marine debris ingestion (NMFS, unpublished data): one animal had monofilament gillnet in its stomach and the second animal had nets of differing types in its stomach. Mean annual takes for all fisheries (Table 1) are based on ~~2006-2010~~ 2001-2012 observer and stranding data (Carretta and Enriquez 2006, 2007, 2009a, 2009b, 2010, 2012, Carretta et al. 2005, Carretta and Chivers 2004, Carretta et al. 2004, 2014a, Jacobsen et al. 2010, NMFS unpublished stranding data). ~~This~~ Including estimates from fishery observer programs (16 animals/12 years = 1.3/yr) and strandings data (5 animals/12 years = 0.4/yr) results in an average estimate of ~~3.8 (CV=0.95)~~ 1.7 sperm whale deaths per year due to fishery-related causes for the period 2001 to 2012. The mean annual mortality from strandings represents a minimum value, as not all carcasses come ashore or are detected.

Table 1. Summary of available information on the incidental mortality and injury of sperm whales (CA/OR/WA stock) for commercial fisheries that might take this species. n/a indicates that data are not available. Mean annual takes are based on ~~2006-2010~~ 2001-2012 data unless noted otherwise.

Fishery Name	Year(s)	Data Type	Percent Observer Coverage	Observed mortality (and serious injury in parentheses)	Estimated mortality and serious injury (CV in parentheses)	Mean annual takes (CV in parentheses)
CA/~~OR~~ thresher shark/swordfish drift gillnet fishery	2001	observer	20.4%	0	0	~~3.2 (0.95)~~ 1.3 (0.95)
	2002		22.1%	0	0	
	2003		20.0%	0	0	
	2004		21.0%	0	0	
	2005		21.0%	0	0	
	2006		18.5%	0	0	
	2007		16.4%	0	0	
	2008		13.5%	0	0	
	2009		13.3%	0	0	
	2010		11.9%	1 (1)	16 (0.95)	
	2011		19.5%	0	0	
	2012		18.7%	0	0	
Unknown fishery	~~2006-2010~~	stranding	n/a	~~3~~	~~≥3~~	~~≥0.6~~

	2001-2012			5	≥5	≥ 0.4
Total annual takes						~~≥ 3.8 (0.95)~~ ≥ 1.7 (0.95)

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

Ship Strikes

One sperm whale died as the result of a ship strike in Oregon in 2007 (NMFS Northwest Regional Stranding data, unpublished). Another sperm whale was struck by a 58-foot sablefish longline vessel in 2007 while at idle speed (Jannot et al. 2011). The observer noted no apparent injuries to the whale. Based on the size and speed of the vessel relative to the size of a sperm whale, this incident was categorized as a non-serious injury (Carretta et al. 2013). ~~Sperm whale mortality and serious injuries attributed to ship strikes averaged 0.2 per year for 2006-2010.~~ For the most recent 5-year period of 2008 to 2012 for which data are available, no ship strikes of sperm whales were documented (Carretta et al. 2014b) and the mean annual average mortality and serious injury is zero whales. Ship strikes are assessed over the most recent 5-year period to reflect the degree of shipping risk to large whales since ship traffic routes changed in response to new ship pollution rules implemented in 2009 (McKenna *et al*. 2012, Redfern *et al*. 2013).

STATUS OF STOCK

The only estimate of the status of North Pacific sperm whales in relation to carrying capacity (Gosho et al. 1984) is based on a CPUE method which is no longer accepted as valid. Whaling removed at least 436,000 sperm whales from the North Pacific between 1800 and the end of legal commercial whaling for this species in 1987 (Best 1976; Ohsumi 1980; Brownell 1998; Kasuya 1998). Of this total, an estimated 33,842 were taken by Soviet and Japanese pelagic whaling operations in the eastern North Pacific from the longitude of Hawaii to the U.S. West coast, between 1961 and 1976 (Allen 1980), and approximately 1,000 were reported taken in land-based U.S. West coast whaling operations between 1919 and 1971 (Ohsumi 1980; Clapham et al. 1997). There has been a prohibition on taking sperm whales in the North Pacific since 1988, but large-scale pelagic whaling stopped in 1980. ~~As a result of this whaling, sperm~~ Sperm 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 status of sperm whales with respect to carrying capacity and optimum sustainable population (OSP) is unknown. Including both fishery and ship-strike mortality, the annual rate of kill and serious injury (~~4.0 per year~~ 1.7 per year) is ~~greater~~ less than the calculated PBR for this stock (~~1.5~~ 2.7). Total human-caused mortality is greater than 10% of the calculated PBR and, therefore, cannot be considered to be insignificant and approaching zero mortality and serious injury rate. Increasing levels of anthropogenic sound in the world's oceans has been suggested to be a habitat concern for whales, particularly for deep-diving whales like sperm whales that feed in the ocean's "sound channel".

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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 became extinct in the Atlantic by the early 1700s (Fraser 1970; Mead and Mitchell 1984), though one anomalous sighting 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 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).

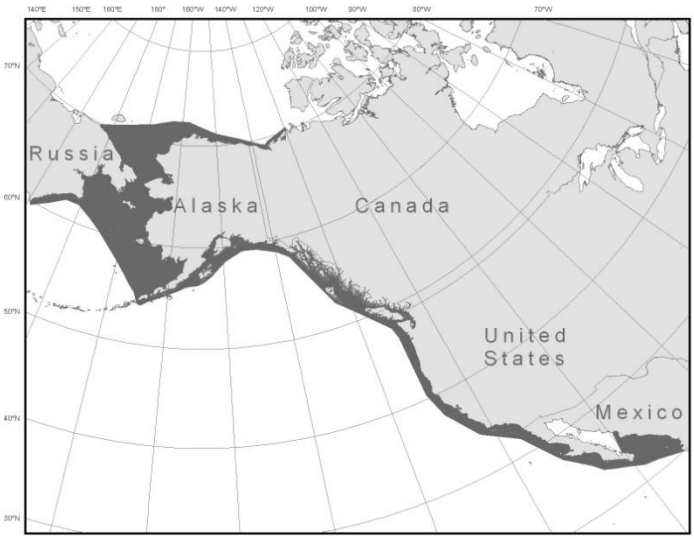

Figure 1. Approximate distribution of the Eastern North Pacific stock of gray whales (shaded area).

During summer and fall, most whales in the ENP population feed in the Chukchi, Beaufort and northwestern Bering Seas (Fig. 1). An exception to this is the relatively small number of whales (approximately 200) 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), ~~also known~~ referred to as the "Pacific Coast Feeding Group" (PCFG). Three primary wintering lagoons in ~~the ENP~~ 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). In combination, these studies have recorded a total of 27 gray whales observed in both the WNP and ENP. ~~Results from photo-identification (Urbán et al. 2012, Weller *et al*. 2012), genetic (Lang 2010), and telemetry studies (Mate *et al*. 2011) have documented spatial and temporal overlap between WNP and ENP gray whales. Observations include: (1) six whales photo-matched from Sakhalin Island to southern Vancouver Island, (2) two whales genetically matched from Sakhalin to Santa Barbara, California, (3) 13 whales photo-matched from Sakhalin Island to San Ignacio Lagoon, Mexico, and (4) two satellite-tagged whales that migrated from Sakhalin Island to the west coast of North America.~~ 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). ~~Although it is clear that some whales feeding in the WNP during the summer/fall migrate to the west coast of North America during the winter/spring, past and present observations of gray whales in the WNP off Japan, Korea and China during the winter/spring suggest that not all gray whales in the WNP share a common wintering ground (Weller and Brownell 2012).~~

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 Pacific Northwest~~ the study area during summer and autumn include two components: **1)** whales that frequently return to the area, display a high degree of intra-seasonal "~~residency~~ fidelity" and account for a majority of the sightings between 1 June and 30 November. Despite movement and interchange among sub-regions of the study area, some whales are more likely to return to the same sub-region where they were observed in previous years~~.~~; **2)**"visitors" from the northbound migration that are sighted only in one year, tend to be seen for shorter time periods in that year, and are encountered in more limited areas. Photo-identification (Gosho *et al*. 2011; Calambokidis *et al*. 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 ~~the utilization~~ 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 ~~how much~~ potential immigration levels into the PCFG ~~could occur~~ using simulations and produced results consistent with the empirical (mtDNA) analyses of Lang et al. (2011b). ~~Results~~ 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 ~~that were~~ most consistent with ~~those of~~ 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. ~~and participants noted that~~ 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. ~~Future~~ Additional research ~~efforts~~ is needed to better identify recruitment levels into the PCFG ~~will be necessary to~~ 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 ~~intends on preparing~~ 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 ~~PCFG within this report. Calculation of a PBR for this feeding~~

~~aggregation allows NMFS~~ to assess whether levels of human-caused mortality are likely to cause local depletion ~~within this population~~.

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 southbound counts were made during the 2007/2008, 2009/2010, and 2010/2011 surveys, from which abundance estimates are not yet available.~~ The most recent estimate of abundance for the ENP population is from the 2010/2011 southbound survey and is 20,990 (CV=0.05) whales (Durban *et al*. 2013) (Fig. 2). ~~2006/2007 southbound survey is 19,126 (CV=7.1%) whales (Laake *et al*. 2012).~~

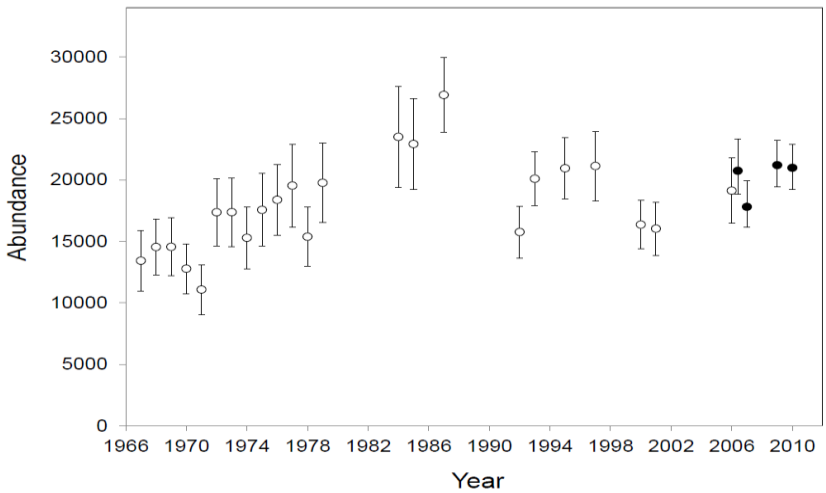

Figure 2. Estimated abundance of Eastern North Pacific gray whales from NMFS counts of migrating whales past Granite Canyon, California. ~~Error bars indicated 90% probability intervals. The solid line represents the estimated trend of the population with 90% intervals as dashed lines (after Punt and Wade 2012).~~ Open circles represent abundance estimates and 95% confidence intervals reported by Laake et al. (2012). Closed circles represent estimates and 95% posterior highest density intervals reported by Durban et al. (2013) for the 2006/7, 2007/8, 2009/10, and 2010/11 migration seasons.

Photographic mark-recapture abundance estimates for PCFG gray whales between 1998 and ~~2010~~ 2012, 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*. (~~2012~~ 2014). The ~~2010~~ 2012 abundance estimate for the defined range of the PCFG between 41°N to 52°N is 209 ~~188~~ (SE=15.4; CV= 0.07 ~~0.10~~).

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 (Figure 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). ~~Estimates of northbound calves from 2001 to 2012 ranged between 254 in 2010 and 1,528 in 2004, with high interannual variability (Perryman and Weller 2012).~~ 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): $N_{MIN} = N/\exp(0.842\times[\ln(1 +[CV(N)]^2)]^{½})$. Using the ~~2006/07~~ 2010/11 abundance estimate of ~~19,126~~ 20,990 and its associated CV of ~~0.071~~ 0.05 (Durban et al. 2013), N_{MIN} for this stock is ~~18,017~~ 20,125.

The minimum population estimate for PCFG gray whales is calculated as the lower 20th percentile of the log-normal distribution of the ~~2010~~ 2012 mark-recapture estimate ~~given above~~ of 209 (CV=0.07), or ~~173~~ 197 animals.

Current Population Trend

The population size of the ENP gray whale stock has increased over several decades despite an UME in 1999 and 2000 and has been relatively stable since the mid-1990s (see Fig. 2). ~~The estimated annual rate of increase, based on the abundance time series from Laake *et al*. (2012) is 3.2% with a standard error of 0.5% (Punt and Wade 2012).).~~

Abundance estimates of PCFG gray whales reported by Calambokidis et al. (~~2012~~ 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 R_{max} is also applied to PCFG gray whales, as it is currently the best estimate of R_{max} 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 (~~18,017~~ 20,125), times one-half of the maximum theoretical net population growth rate (½ x 6.2% = 3.1%), times a recovery factor of 1.0 for a stock above MNPL (Punt and Wade 2012), or ~~559~~ 624 animals.

The potential biological removal (PBR) level for PCFG gray whales is calculated as the minimum population size (~~173~~ 197 animals), times one half the maximum theoretical net population growth rate (½ x 6.2% = 3.1%), times a recovery factor of 0.5 (for a population of unknown status), resulting in a PBR of ~~2.7~~ 3.1 animals. 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.

~~ANNUAL~~ 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~~ distinguish~~ing~~ serious from non-serious injury (Angliss and DeMaster 1998, Andersen *et al*. 2008, NOAA 2012). NMFS defines serious injury as an "*injury that is more likely than not to result in mortality*". ~~Injury determinations for stock assessments revised in 2013 or later incorporate the new serious injury guidelines, based on the most recent 5-year period for which data are available.~~

Fisheries Information

No gray whales were observed entangled in California gillnet fisheries between ~~2007 and 2011~~ 2008 and 2012 (Carretta and Enriquez 2009~~a~~, ~~2009b~~, 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). Alaska gillnet fisheries largely lack observer programs, including those in Bristol Bay known to interact with gray whales. Most data on human-caused mortality and serious injury of gray whales ~~is~~ are from strandings, including at-sea reports of entangled animals alive or dead (Carretta et al. 2013, 2014b). 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.

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 ~~2007 to 2011~~ 2008 to 2012. Total observed human-caused fishery mortality and serious injury for ENP gray whales ~~for the period 2007 to 2011~~ is ~~11.25~~ 22.25 animals (~~3~~ 8 serious injuries, ~~5.25~~ 8.25 prorated serious injuries, and ~~3~~ 6 deaths), or ~~2.25~~ 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 ~~2007 to 2011~~ 2008 to 2012 is ~~1.75~~ 0.75 animals (~~1 serious injury and~~ 0.75 prorated serious injuries), or ~~0.35~~ 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.

Table 1. Human-caused deaths and serious injuries (SI) of gray whales from fishery-related and marine debris sources for the period ~~2007 to 2011~~ 2008 to 2012 as recorded by NMFS stranding networks and observer programs.

Date of observation	Location	PCFG range N 41- N 52 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 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 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 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 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 spongex 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 removed on 03/24 found to contain one dead ca sea lion and three dead sharks.	Dead
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	Petersburg, AK	No	Entangled in 50 lbs. Heavy monofilament webbing, cork line, and lead line, as well as over 200 lbs. Of bull kelp attached to gear; completely disentangled; leading edge of flukes had significant cuts and abrasions; overall body condition was poor; massive infestation of whale lice and barnacles; animal very emaciated and lacked any visible signs of recent feeding; observed the day after disentanglement swimming very slowly. Apparent health decline due to constricting and weighted entanglement.	SI
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.).	~~SI~~ 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
~~26-Jul-2007~~	~~Seattle WA~~	~~No[1]~~	~~Some gear was removed from the animal, swam away with gear still attached, tribal fishing nets, animal was not sighted again to remove more gear.~~	~~SI (0.75)~~
~~20-Apr-2007~~	~~Newport OR~~	~~No~~	~~Entangled in crab gear. skipper of nearby vessel removed 8 pots before he had to return to port due to darkness whale still had 8 buoys and several wraps of line around mid section, left pectoral flipper, and through mouth~~	~~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 ~~the coast of~~ Washington State (NMFS 2008). The spatial overlap of the Makah U&A and the summer distribution of PCFG whales has management implications. The 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. ~~Similarly,~~ 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 Tribe (Moore and Weller ~~2012~~2013). NMFS has published a notice of intent to prepare an environmental impact statement (EIS) on the proposed hunt (NMFS 2012) 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 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 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 Russian federation. The U.S. and Russia 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: ~~126 in 2007, 127~~ 130 in 2008, ~~115~~ 116 in 2009, 118 in 2010, ~~and~~ 128 in 2011, and 143 in 2012 (source: http://iwc.int/table_aboriginal). Based on this information, the annual subsistence take averaged ~~123~~ 127 whales during the 5-year period from ~~2007 to 2011~~ 2008 to 2012.

~~[1]For purposes of calculating annual human-caused mortality, this whale is counted as an ENP whale and not part of the PCFG. This determination is based on observations that PCFG whales are not known to enter Puget Sound and current estimates of PCFG population size exclude whales seen in this area (J. Calambokidis, Cascadia Research, personal communication).~~

Other Mortality

Ship strikes are a source of mortality for gray whales (Table 2). For the most recent five-year period, ~~2007-2011~~ 2008-2012, the total serious injury and mortality of ENP gray whales attributed to ship strikes is ~~10.8~~ 9.8 animals (including ~~eight~~ 7 deaths, ~~two~~ 2 serious injuries, and ~~three~~ 0.8 prorated serious injuries, or ~~2.2~~ 2.0 whales per year (Table 2, Carretta et al. 2013, Carretta et al. 2014b.). The total ship strike serious injury and mortality of gray whales observed in the PCFG range and season during this same period is 0.52 animals, or 0.1 whales per year (Table 2). 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 or do not have 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.

~~One PCFG gray whale was illegally killed by hunters in Neah Bay in September 2007 (Calambokidis *et al.* 2009).~~

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~~, Moore and Huntington 2008~~).

Table 2. Summary of gray whale serious injuries (SI) and deaths attributed to vessel strikes for the five-year period ~~2007-2011~~2008-2012. No vessel strikes were reported in 2012.

Date of observation	Location	PCFG range N 41 - N 52 AND season?	Description	Determination (SI prorate 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	SI

			of skin sample confirmed species.	
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	Ilwaco 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 betweeen 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
~~1-Jun-2007~~	~~Marin, CA~~	~~No~~	~~Carcass; 4 propeller-like wounds to body~~	~~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 ~~(ESA)~~(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 ~~of the environment~~ (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 around its average carrying capacity. This is consistent with a population approaching K.

~~Alter et al. (2007) used estimates of genetic diversity to infer that North Pacific gray whales may have numbered ~96,000 animals in both the western and eastern populations 1,100-1,600 years ago. The authors recommend that because the current estimate of the eastern stock of gray whales is at most 28-56% of this historic abundance, the stock should be designated as "depleted" under the MMPA. NMFS does not accept the recommendation made by Alter et al. (2007) for the following reasons. First, their analysis examines the historic population of the entire Pacific population of gray whales, while MMPA management occurs at the level of a stock, which in this case is the ENP stock. Second, NMFS relies on current carrying capacity in making MMPA determinations. Ecosystems change over time and with those changes, the carrying capacity of the ecosystem also changes. NMFS interprets carrying capacity to mean "current" carrying capacity in part because it is not reasonable to expect ecosystems to remain static over thousands of years. Thus, an estimate of stock abundance 1,100-1,600 years ago is not relevant to MMPA decision-making, even if such an estimate were available.~~

Based on ~~2007-2011~~2008-2012 data, the estimated annual level of human-caused mortality and serious injury for ENP gray whales includes Russian harvest (~~123~~127), mortality and serious injury from commercial fisheries (~~2.25~~ 4.45), and ship strikes (~~2.2~~ 2.0), totals ~~127~~ 133 whales per year, which does not exceed the PBR (~~558~~ 624). 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). Therefore, the ENP stock of gray whales is not classified as a strategic stock.

PCFG gray whales do not currently have a formal status under the MMPA, though the population size appears to have been stable since 2003, based on photo-ID studies (Calambokidis et al. ~~2012~~ 2014, IWC 2012). Total annual human-caused mortality of PCFG gray whales during the period 2008 to 2012 includes deaths due to commercial fisheries (~~0.35~~ 0.15/yr), and ship strikes (0.1/yr), or 0.25 whales annually. This does not exceed the PBR

level of ~~2.7~~ 3.1 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; Figure 1). 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). Further, 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).

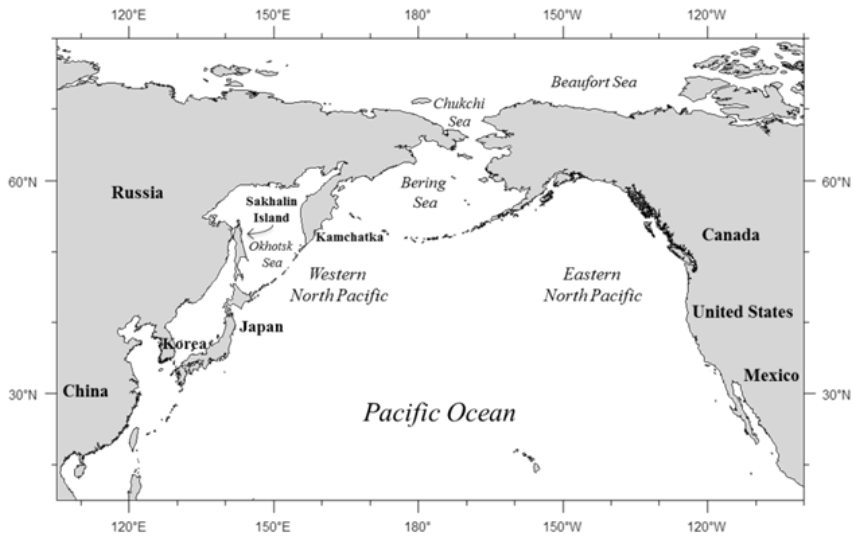

Figure 1. Range map of the Western North Pacific Stock of gray whales, including summering areas off Russia and wintering areas in the western and eastern Pacific.

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 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). In combination, these studies have recorded a total of 27 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. 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 between 1994 and 2011 on the gray whale summer feeding ground off Sakhalin Island in the WNP were 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

The minimum population estimate (N_{min}) for the WNP stock is calculated from Equation 1 from the PBR Guidelines (Wade and Angliss 1997): $N_{min} = N/\exp(0.842\times[\ln(1 +[CV(N)]^2)]½)$ and the abundance estimate of 140 (CV=0.043) whales from Cooke et al. (2013), resulting in a minimum population estimate of 135 gray whales on the summer feeding ground off Sakhalin Island in the WNP.

Current Population Trend

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 (135), times one-half the estimated maximum annual growth rate for a gray whale population (½ 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 0.06 WNP gray whales per year, or approximately 1 whale every 17 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 distinguis 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 incidental catches in coastal net fisheries (Weller et al. 2002; Kato et al. 2012; Weller et al. 2008; Weller et al. 2013a). 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). 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 (Bradford et al. 2009), further highlighting the overall risks coastal fisheries pose to WNP gray whales.

In summer 2013, salmon net fishing was observed for the first time on the gray whale feeding ground off Sakhalin Island. 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). 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.

HABITAT CONCERNS

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). As long as the WNP stock remains listed as endangered under the ESA, it will continue to be considered as depleted under the MMPA.

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Revised ~~6/4/2014~~ 10/22/2014

FALSE KILLER WHALE (*Pseudorca crassidens*): Hawaiian Islands Stock Complex – Main Hawaiian Islands Insular, Northwestern Hawaiian Islands, and Hawaii Pelagic Stocks

STOCK DEFINITION AND GEOGRAPHIC RANGE

False killer whales are found worldwide ~~mainly~~ in tropical and warm-temperate waters (Stacey et al. 1994). In the North Pacific, this species is well known from southern Japan, Hawaii, and the eastern tropical Pacific. ~~There are seven stranding records from Hawaiian waters since 1974 (Nitta 1991; Maldini et al. 2005, NMFS PIR Marine Mammal Response Network database), including one since 2007.~~ One on-effort sighting of false killer whales was made during a 2002 shipboard survey, and six during a 2010 shipboard survey of waters within the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands (Figure 1; Barlow 2006, Bradford et al. ~~2012~~ 2014). Smaller-scale surveys conducted around the main Hawaiian Islands (Figure 2) show that false killer whales are also encountered in near shore waters (Baird et al. 2005, Mobley et al. 2000), and a single on-effort and three off-effort sightings during ~~a~~ the 2010 Hawaiian Islands Cetacean Ecosystem Assessment Survey (HICEAS) shipboard survey reveal that the species also occurs near shore in the Northwestern Hawaiian Islands (Baird et al. 2013). This species also occurs in U.S. EEZ waters around Palmyra and Johnston Atolls (e.g., Barlow et al. 2008, Bradford & Forney 2013 ~~NMFS/PIR/PSD unpublished data~~) and American Samoa (Johnston et al. 2008, Oleson 2009).

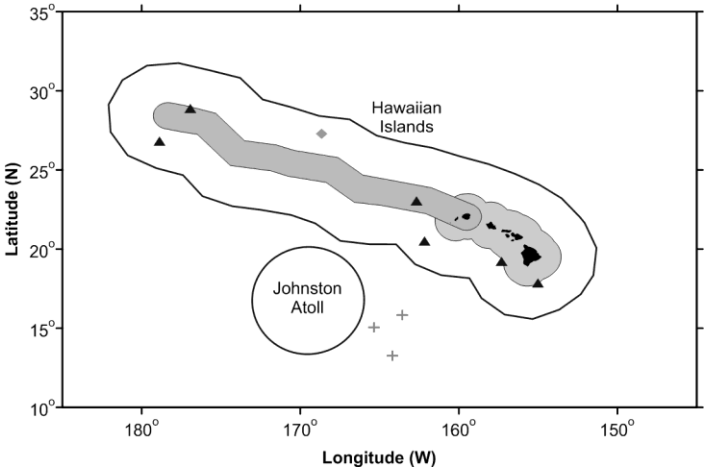

Figure 1. False killer whale on-effort sighting locations during standardized shipboard surveys of the Hawaiian Islands U.S. EEZ (2002, gray diamond, Barlow 2006; 2010, black triangles, Bradford et al. ~~2012~~ 2014, pelagic waters of the central Pacific south of the Hawaiian Islands (2005, gray crosses, Barlow and Rankin 2007) and the Johnston Atoll EEZ. Outer lines represent approximate boundary of U.S. EEZs; light shaded gray area is the main Hawaiian Islands insular false killer whale stock area, including overlap zone between MHI insular and pelagic false killer whale stocks; dark shaded gray area is the Northwestern Hawaiian Islands stock area, which overlaps the pelagic false killer whale stock area and part of the MHI insular false killer whale stock area.

Genetic, photo-identification, and telemetry studies indicate there are three demographically-independent populations of false killer whales in Hawaiian waters. Genetic analyses indicate restricted gene flow between false killer whales sampled near the main Hawaiian Islands (MHI), the Northwestern Hawaiian Islands (NWHI), and in pelagic waters of the Eastern (ENP) and Central North Pacific (CNP) (Chivers et al. 2007, 2010; Martien et al. 2011). Chivers et al. (2010) expanded on previous analyses ~~with~~ using additional samples and including analysis of 8 nuclear DNA (nDNA) microsatellites, revealing strong phylogeographic patterns consistent with local evolution of haplotypes nearly unique to false killer whales occurring nearshore within the Hawaiian Archipelago. Analysis of 21 additional samples collected during HICEAS in ~~a~~ 2010 ~~shipboard survey in Hawaiian waters~~ reveals significant differentiation in both mitochondrial DNA (mtDNA) and nDNA between false killer whales found near the MHI and the NWHI (Martien et al. 2011). Photographic–identification of individuals seen near the NWHI confirms that they do not associate with individuals near the MHI south of Kauai (Baird et al. 2013). Two false killer whales previously photographed near Kauai were seen in groups observed near Nihoa in the NWHI, and are not known to associate with animals from the MHI, suggesting geographic overlap of MHI and NWHI false killer whale populations near Kauai. Further evaluation of photographic and genetic data from individuals seen near the MHI suggests the occurrence of three separate social clusters (Baird et al. 2012, Martien et al. 2011), where mating ~~primarily~~ occurs primarily, though not exclusively within clusters~~, though some mating is known to occur between males and females of different social clusters~~ (Martien et al. 2011).

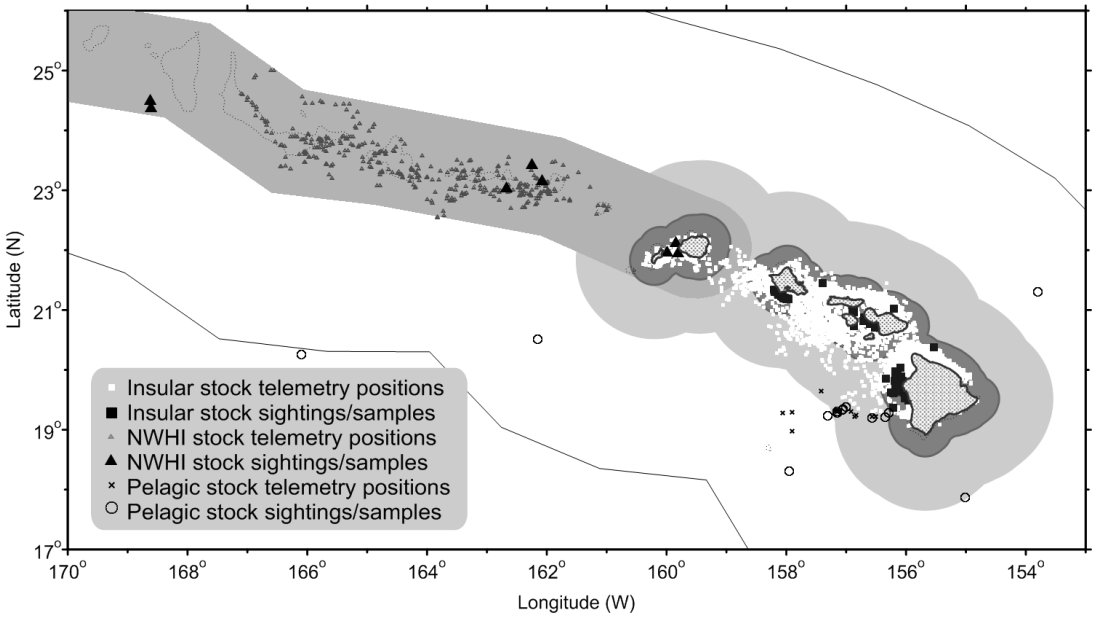

Figure 2. Sighting, biopsy, and telemetry records of false killer whale identified as being part of the MHI insular (square symbols), NWHI (triangle symbols), or pelagic (open and cross symbols) stocks. The dark gray area is the 40-km MHI insular core area; light gray area is the 40-km to 140-km MHI insular-pelagic overlap zone (Baird et al. 2010, Baird et al. 2013, ~~unpublished data; reproduced from Forney et al. 2010~~); medium gray area is the 50-nmi (93-km) Monument boundary extended to the east to encompass Kauai, representing the NWHI stock boundary. The MHI insular, pelagic, and NWHI stocks overlap in the vicinity of Kauai.

Fishery ~~O~~ observers have collected tissue samples for genetic analysis from cetaceans incidentally caught in the Hawaii-based longline fishery since 2003. Between 2003 and 2010, eight false killer whale samples, four collected outside the Hawaiian EEZ and four collected within the EEZ but more than 100 nautical miles (185km) from the main Hawaiian Islands ~~(see Figure 3)~~ were determined to have Pacific pelagic haplotypes (Chivers et al. 2010). At the broadest scale, significant differences in both mtDNA and nDNA are evident between pelagic false killer whales in the ENP and CNP strata (Chivers et al. 2010), although the sample distribution to the east and west of Hawaii is insufficient to determine whether the sampled strata represent one or more stocks, and where pelagic stock boundaries would be drawn.

~~Genetic, photographic, and telemetry data collected from Hawaiian false killer whales demonstrate the existence of a previously unknown stock of island-associated false killer whales in the NHWI, and support the current recognized boundaries of the MHI insular and pelagic stocks.~~ The three Hawaiian stocks of false killer whales have overlapping ranges. MHI insular false killer whales have been seen as far as 112 km from the main Hawaiian Islands, while pelagic stock animals have been seen within 42 km of the main Hawaiian Islands (Baird et al. 2008, Baird 2009, Baird et al. 2010, Forney et al. 2010). NWHI false killer whales have been seen as far as 93 km from the NWHI and near Kauai (Baird et al. 2012, Bradford et al. 2012, Martien et al. 2011). Animals seen within 40 km of each of the main Hawaiian Islands from Hawaii Island to Oahu are considered to belong to the MHI insular stock. Waters within 40 km of Kauai and Niihau are an overlap zone between the MHI insular and NWHI stocks, as individuals from both populations are known to occur there. Animals seen within 93 km of the NWHI, inside the Papahānaumokuākea Marine National Monument may belong to either the NWHI or pelagic stock, as animals from both stocks have been seen inside the Monument. Animals beyond 140 km of the MHI and beyond 93 km of the NWHI are considered to belong to the pelagic stock. The MHI insular and pelagic stocks overlap between 40 km and 140 km from shore contiguously between Oahu and Hawaii Island. All three stocks overlap within 40 km and 93 km around Kauai and Niihau, and the MHI insular and pelagic stocks overlap from 93 km to 140 km around these islands (Figure 2).

The pelagic stock includes animals found within the Hawaiian Islands EEZ and in adjacent international waters; however, because data on false killer whale abundance, distribution, and human-caused impacts are largely lacking for international waters, the status of this stock is evaluated based on data from U.S. EEZ waters of the Hawaiian Islands (NMFS 2005). The Palmyra Atoll stock of false killer whales is~~are~~ still considered to be a separate stock, because comparisons amongst false killer whales sampled at Palmyra Atoll and those sampled from the MHI insular stock and the pelagic ENP reveal restricted gene flow, although the sample size remains too low for robust

comparisons (Chivers et al. 2007, 2010). NMFS will obtain and analyze additional samples for genetic studies of stock structure, and will evaluate new information on stock ranges as it becomes available.

For the Marine Mammal Protection Act (MMPA) stock assessment reports, there are currently five Pacific Islands Region management stocks (Forney et al. 2011, Martien et al. 2011): 1) the Main Hawaiian Islands insular stock, which includes animals inhabiting waters within 140 km (approx. 75 nmi) of the main Hawaiian Islands, 2) the Northwestern Hawaiian Islands stock, which includes animals inhabiting waters within 93 km (50 nmi) of the NWHI and Kauai, 3) the Hawaii pelagic stock, which includes false killer whales inhabiting waters greater than 40 km (22 nmi) from the main Hawaiian Islands, including adjacent high seas waters, 4) the Palmyra Atoll stock, which includes animals found within the U.S. EEZ of Palmyra Atoll, and 5) the American Samoa stock, which includes animals found within the U.S. EEZ of American Samoa. Estimates of abundance, potential biological removal, and status determinations for the first three stocks are presented below; the Palmyra Atoll and American Samoa ~~S~~stocks are covered in separate reports.

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

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~~ distinguish~~ing~~ serious from non-serious injury (Angliss and DeMaster 1998, Andersen et al. 2008, NOAA 2012). NMFS defines serious injury as an "*injury that is more likely than not to result in mortality*". ~~Injury determinations for stock assessments revised in 2013 or later incorporate the new serious injury guidelines, based on the most recent 5-year period for which data are available.~~

Fishery Information

Interactions with false killer whales, including depredation of catch of a variety of pelagic fishes, have been identified in logbooks and NMFS observer records from Hawaii pelagic longline fishing trips (Nitta and Henderson 1993, Oleson et al. 2010, NMFS/PIR unpublished data). False killer whales have been observed feeding on mahi mahi, *Coryphaena hippurus*, and yellowfin tuna, *Thunnus albacares* (Baird 2009), and they have been reported to take large fish from the trolling lines of commercial and recreational fishermen (Shallenberger 1981). There are anecdotal reports of marine mammal interactions in the commercial Hawaii shortline fishery which sets gear at Cross Seamount and possibly around the main Hawaiian Islands. The shortline fishery is permitted through the State of Hawaii Commercial Marine License program, and until recently, no reporting systems existed to document marine mammal interactions.

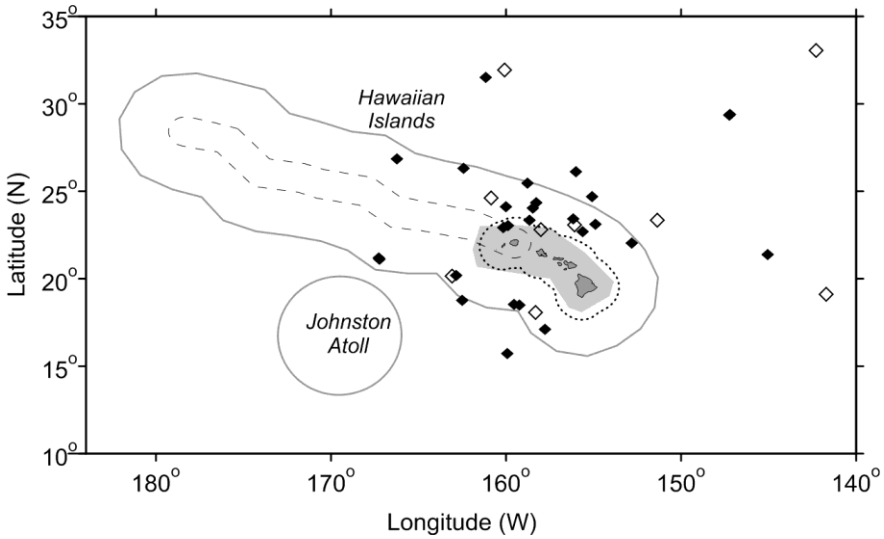

Figure 3. Locations of observed false killer whale takes (black diamonds) and possible takes (blackfish) of this species (open diamonds) in the Hawaii-based longline fisheries, 2008-2012 ~~2007-2011~~. Some take locations overlap. Solid gray lines represent the U.S. EEZ; the dotted line is the outer (140-km) boundary of the overlap zone between MHI insular and pelagic false killer whale stocks; the dashed line is the 93-km boundary of the NWHI stock; the gray shaded area is the February-September longline exclusion zone.

Baird and Gorgone (2005) documented high rates of dorsal fin disfigurements consistent with injuries from unidentified fishing line for false killer whales belonging to the MHI insular stock. A recent report included evaluation of additional individuals with dorsal fin injuries and suggested that the rate of interaction between false killer whales and various forms of hook and line gear may vary by population and social cluster, with the MHI insular stock showing the highest rate of dorsal fin disfigurements (Baird et al. 2014). It is unknown whether these injuries might have been caused by longline gear, shortline gear, or other commercial or recreational hook-and-line

gear used around the main Hawaiian Islands. Examination of a stranded MHI insular false killer whale in October 2013 revealed that this individual had five fishing hooks and fishing line in its stomach (NMFS PIR Marine Mammal Response Network). Although the fishing gear is not believed to have caused the death of the whale, the finding confirms that MHI insular false killer whales are consuming previously hooked fish or are interacting with hook and line fisheries in the MHI. Many of the hooks within the whale's stomach were not consistent with those currently allowed for use within the commercial longline fisheries and could have come from a variety of near shore fisheries. No estimates of human-caused mortality or serious injury are currently available for near shore hook and line or gillnet fisheries because these fisheries are not observed or monitored for protected species bycatch.

Table 1. Summary of available information on incidental mortality and serious injury (MSI) of false killer whales (Hawaiian Islands Stock Complex) and unidentified blackfish in commercial longline fisheries, by stock and EEZ area, as applicable (McCracken ~~2013~~2014). Mean annual takes are based on ~~2007-2011~~2008-2012 estimates unless otherwise indicated (a new alternative was explored in this report for prorating among three stocks). Information on all observed takes (T) and combined mortality ~~events &~~ serious injury~~ies (MSI)~~ is included. Total takes were prorated to deaths, serious injuries, and non-serious injuries based on the observed proportions of each outcome. Unidentified blackfish are pro-rated as either false killer whales or short-finned pilot whales according to their distance from shore (McCracken 2010). CVs are estimated based on the combined variances of annual false killer whale and blackfish take estimates and do not yet incorporate additional uncertainty introduced by prorating false killer whale~~s~~ takes in the overlap zone and prorating the takes of unidentified blackfish.

Fishery Name	Year	Data Type	Percent Observer Coverage	Observed total interactions (T), observed mortality events~~,~~ and serious injuries (MSI), and total estimated mortality and serious injury (MSI) of false killer whales by stock / EEZ region							
				Hawaii Pelagic Stock				Main Hawaiian Islands Insular Stock		Northwestern Hawaiian Islands Stock	
				Outside U.S. EEZs		Hawaiian EEZ					
				Obs. FKW T/MSI Obs. UB T/MSI	Estimated MSI (CV)	Obs. FKW T/MSI Obs. UB T/MSI	Estimated MSI (CV)	Obs. FKW T/MSI Obs. UB T/MSI	Estimated MSI (CV)	Obs. FKW T/MSI Obs. UB T/MSI	Estimated MSI (CV)
Hawaii-based deep-set longline fishery	2008	Observer data	22%	0 0	0 (-)	3/3 3/3	17 (0.4)	0 0	0 (-)	0 0	0 (-)
	2009		21%	7/7 0	39 (0.2)	3/3 0	12 (0.6)	0 0	0 (-)	0 0	0 (-)
	2010		21%	1/1 0	6 (1.4)	3/2 1/1	14 (0.4)	0 0	0 (-)	0 0	0 (-)
	2011		20%	0 1/0	2 (2.0)	3/2 1/1*	12 (0.5)	0 1/1*	0 (-)	0 0	0 (-)
	2012		20%	0 1/1	4 (2.0)	3/2* 0/0	8 (0.4)	2/2* 0/0	4 (0.4)	2/2* 0/0	1 (0.4)
Mean Estimated Annual Take (CV)					**9.9 (0.4)**		**12.7 (0.2)**		**0.9 (2.0)**		**0.4 (1.5)**
Hawaii-based shallow-set longline fishery	2008	Observer data	100%	0 1/1	~~0~~1	1/0 0	0	0 0	0	0 0	0
	2009		100%	0 0	0	1/1 0	1	0 0	0	0 0	0
	2010		100%	0 0	0	0 0	0	0 0	0	0 0	0
	2011		100%	0 1/1	~~0~~1	1/0 0	0	0 0	0	0 0	0
	2012		100%	0 0	0	1/0 0	0	0 0	0	0 0	0
Mean Annual Takes (100% coverage)					**0.3**		**0.3**		**0**		**0**
Minimum total annual takes within U.S. EEZ							**13.0 (0.2)**		**0.9 (2.0)**		**0.4 (1.5)**

Fishery Name	Year	Data Type	Percent Observer Coverage	Observed total interactions (T) and mortality events, and serious injuries (MSI), and total estimated mortality and serious injury (M&SI) of false killer whales by stock / EEZ region							
				Hawaii Pelgic Stock				Main Hawaiian Islands Insular Stock		Northwestern Hawaiian Islands Stock	
				Outside U.S. EEZs		Hawaiian EEZ					
				Obs. FKW T/MSI Obs. UB T/MSI	Estimated M&SI (CV)	Obs. FKW T/MSI Obs. UB T/MSI	Estimated M&SI (CV)	Obs. FKW T/MSI Obs. UB T/MSI	Estimated M&SI (CV)	Obs. FKW T/MSI Obs. UB T/MSI	Estimated M&SI (CV)
Hawaii-based deep-set longline fishery	2007	Observer data	20%	1/0 0	2 (3.4)	2/1 0	8 ( )	0 0	0 ( )	0 0	0 ( )
	2008		22%	0 0	0 ( )	3/3 3/3	17 ( )	0 0	0 ( )	0 0	0 ( )
	2009		21%	7/7 0	38 (0.2)	3/3 0	12 ( )	0 0	0 ( )	0 0	0 ( )
	2010		21%	1/1 0	6 (1.4)	3/2 1/1	14 ( )	0 0	0 ( )	0 0	0 ( )
	2011		20%	0 1/0	2 (0.6)	2/2 1/1*	11 ( )	0 1/1*	1 ( )	0 0	0 ( )
Mean Estimated Annual Take (CV)					9.6 (0.4)	-	12.4 (0.3)	-	0.1 (0.3)	-	0 (-)
Hawaii-based shallow-set longline fishery	2007	Observer data	100%	0 0	0	0 0	0	0 0	0	0 0	0
	2008		100%	0 1/1	1	1/0 0	0	0 0	0	0 0	0
	2009		100%	0 0	0	1/1 0	1	0 0	0	0 0	0
	2010		100%	0 0	0	0 0	0	0 0	0	0 0	0
	2011		100%	0 1/1	1	1/0 0	0	0 0	0	0 0	0
Mean Annual Takes (100% coverage)					0.3	-	0.2	-	0	-	0
Minimum total annual takes within U.S. EEZ				-	-	-	12.6 (0.3)	-	0.1 (0.3)	-	0

* False killer whale and unidentified blackfish takes within the Hawaiian stock overlap zones MHI insular/pelagic stock overlap zone are shown once for each stock. Within the MHI insular and pelagic overlap zones, but total estimates derived from these takes are first prorated among potentially affected stocks based on the distance from shore of the take location (see text above, and McCracken 2010). Then, within the 3-way NWHI/MHI insular/pelagic overlap zone, the estimates were further prorated based on the relative level of fishing effort in each zone and the density of each stock within each zone, as an alternative to assigning the entire estimated insular take to both insular stocks (MHI and NWHI).

There are two distinct longline fisheries based in Hawaii: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas, but are prohibited from operating within the Papahanaumokuakea Marine National Monument and within the Longline Exclusion Area around the main Hawaiian Islands. Between 2007 2008 and 2011 2012, three four false killer whales were observed hooked or entangled in the SSLL fishery (100% observer coverage) within the U.S. EEZ of the Hawaiian Islands, and 22 23 false killer whales were observed taken in the DSLL fishery (20-22% observer coverage) within Hawaiian waters or adjacent high-seas waters (excluding Palmyra Atoll EEZ waters) (Bradford & Forney 2014 2013). Based on an evaluation of the observer's description of each interaction and following the most recently developed criteria for assessing serious injury in marine mammals (NMFS 2012), two animals taken in the SSLL fishery within the Hawaii EEZ were considered not seriously injured, and one was considered seriously injured., and the level of injury could not be determined for one additional animal based on the observer's descriptions of the interaction. In the DSLL fishery, two one taken in Hawaiian waters within the range of the pelagic stock and one taken on the high seas were was considered not seriously injured and the level of injury could not be determined for one two additional animals based on the observers' descriptions of the interactions. The remaining 19 20 false killer whales taken in the DSLL fishery, eight in high seas waters, and eleven ten in the Hawaiian Islands EEZ pelagic stock range, and two in the three-way overlap zone between the

pelagic, MHI insular, and NWHI stocks were considered seriously injured (Bradford & Forney ~~2013~~ 2014). ~~Seven~~ Eight additional unidentified "blackfish" (unidentified cetaceans known to be either false killer whales or short-finned pilot whales) ~~that may have been false killer whales~~ were also seriously injured during ~~2007-2011~~ 2008-2012 (Bradford & Forney 2014 ~~2013~~). Additionally, one unidentified blackfish was taken on the high seas in the deep set longline fishery in 2011, but was not seriously injured (Table 1). ~~Five of the seven~~ Six of the eight seriously injured false killer whales ~~injuries~~ were taken in the DSLL fishery within U.S. EEZ waters, including one animal within the MHI insular/pelagic stock overlap zone ~~range~~, and the remaining two seriously injured ~~injuries~~ false killer whales were taken by the SSLL fishery on the high seas (Table 1 and Figure 3).

Takes of false killer whales of unknown stock within 140km of the Main Hawaiian Islands must be prorated to MHI insular, pelagic, or NWHI stocks. ~~in the MHI insular/pelagic stock overlap zone are prorated to one stock or the other assuming that densities of MHI insular stock animals decline and pelagic stock densities increase with distance from shore (McCracken 2010).~~ No genetic samples are available to establish stock identity for these takes, but all~~both~~ stocks are considered at risk of interacting with longline gear. The pelagic stock is known to interact with longline fisheries in waters offshore of the overlap zone, based on two genetic samples obtained by fishery observers (Chivers et al. 2008). MHI insular and NWHI false killer whales have been documented via telemetry to move far enough offshore ~~(112km)~~ to reach longline fishing areas, and animals from ~~this~~the MHI insular stock have a high rate of dorsal fin disfigurements consistent with injuries from unidentified fishing line (Baird and Gorgone 2005, Baird et al. 2014). ~~Finally, t~~ Takes of unidentified blackfish are prorated to each ~~stock~~ species based on distance from shore (McCracken 2010). The distance-from-shore model was chosen following consultation with the Pacific Scientific Review Group, based on the model's logic and performance ~~and simplicity~~ relative to a number of other ~~more complicated~~ models with similar output (McCracken 2010). ~~Proration of false killer whale takes within the MHI insular-pelagic overlap zone and of unidentified blackfish takes introduces unquantified uncertainty into the bycatch estimates, but until methods of determining stock identity for animals observed taken within the overlap zone are available, and all animals taken can be identified to species (e.g., photos, tissue samples), this approach ensures that potential impacts to all stocks are assessed.~~ Following proration of unidentified blackfish takes to species, total false killer whale take estimates within 140km of the MHI are first prorated to the MHI insular or pelagic stock assuming that the density of MHI insular stock animals declines and pelagic stock density increases with distance from shore as in the methods of McCracken (2010).

With the McCracken (2010) proration between MHI insular and pelagic stocks as a starting point, two alternatives were examined for allocating takes among the 3-stocks in the 140-km overlap zone. The first alternative partitioned the take within the 140-km zone among the 2 and 3-way overlap zones based on the relative level of fishing effort in each zone. Because a much greater proportion of fishing has occurred in the 2-way overlap zone between MHI insular and pelagic false killer whales than in the smaller overlap zone between all three stocks, the majority of takes were assigned to the 2-way overlap zone. The distance-from-shore model implemented by McCracken (2010) provides a relative probability of occurrence and density of MHI insular versus pelagic stock take within the 140km region given individual take locations in each year. Relative density and take rate were used within the 3-way overlap zone to compute an assumed constant proportion of take between these two stocks among the overlap zones. The NWHI stock density was then joined with these adjusted MHI insular and pelagic stock densities, and the total take estimate for that zone was prorated among the three stocks based on their relative densities in this zone. A similar approach was used to prorate take between the NWHI and pelagic stocks in a small area of overlap outside of 140km that is open to longline fishing. First, total pelagic stock take outside of 140km was partitioned based on the distribution of fishing effort in the NWHI-pelagic stock overlap and pelagic-only zones, then the take assigned to the small NWHI-pelagic overlap zone~~-~~ was prorated between stocks based on the relative densities of each stock. Using this approach, the 5-yr annual mortality and serious injury estimates of MHI insular, NWHI, and pelagic stocks are 0.9, 0.4, and 13.0, respectively.

As an alternative to this approach, GAMMS suggests assigning all take within an overlap zone to all potentially affected stocks. Using this approach all MHI insular stock take within the 140-km zone estimated following the initial proration (McCracken 2010) could be assigned to both MHI insular and NWHI stocks. This approach results in 5-yr annual mortality and serious injury estimates of MHI insular and NWHI stocks of 1.0, and a pelagic stock estimated take of 13.0. The overall status of each stock relative to PBR does not change versus the first approach described above.

The first proration approach is preferable, as it uses fishing location data and the relative densities of false killer whales to partition take among stocks. Based on these bycatch analyses, including the new alternative 3-way proration, estimates of annual and 5-yr average annual mortality and serious injury of false killer whales, by stock and EEZ area, are shown in Table 1. Estimates of mortality and serious injury (M&SI) include a pro-rated portion of the animals categorized as unidentified blackfish (UB). Although annual M&SI estimates are shown as whole numbers of animals, the 5-yr average M&SI is calculated based on the unrounded annual estimates. Proration of

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

Because of high rates of false killer whale mortality and serious injury in Hawaii-based longline fisheries, a Take Reduction Team (Team) was established in January 2010 (75 FR 2853, 19 January 2010). The Team was charged with developing recommendations to reduce incidental mortality and serious injury of the Hawaii pelagic, MHI insular, and Palmyra stocks of false killer whales in the DSLL and SSLL fisheries. The Team submitted a draft Take Reduction Plan (Plan) to NMFS (http://www.nmfs.noaa.gov/pr/pdfs/interactions/fkwtrp_draft.pdf), and NMFS published a final Plan based on the Team's recommendations (77 FR 71260, 29 November, 2012). Take reduction measures include gear requirements, time-area closures, and measures to improve captain and crew response to hooked and entangled false killer whales. The Plan became effective December 31, 2012, with gear requirements effective February 27, 2013. Additionally, the Plan includes non-regulatory measures that NMFS will implement to improve data quality and dissemination to the Team and the public. These measures were not in effect during 2008-2012, the period for which bycatch was estimated in this report. Bycatch estimation methods will need to be adjusted when 2013 takes are considered to account for changes in fishing gear and captain training intended to reduce the false killer whale serious injury rate.

MAIN HAWAIIAN ISLANDS INSULAR STOCK

POPULATION SIZE

A photographic mark-recapture study during 2000-2004 around the main Hawaiian Islands produced an estimate of 123 (CV=0.72) MHI insular false killer whales (Baird et al. 2005). This abundance estimate is based in part on data collected more than 8 years ago, and is considered outdated as a measure of current abundance (NMFS 2005). A Status Review for the MHI insular stock in 2010 (Oleson et al. 2010) used recent, unpublished estimates of abundance for two time periods, 2000-2004 and 2006-2009 in a Population Viability Analysis (PVA). These new estimates were based on more recent sighting histories and open population models, yielding more precise estimates for the two time periods. The new abundance estimate for the 2000-2004 period is 162 (CV=0.23) animals. Two separate estimates for 2006-2009 were presented in the Status Review; 151 (CV=0.20) and 170 (CV=0.21), depending on whether animals photographed near Kauai are included in the estimate ~~(Baird unpublished data)~~. The animals seen near Kauai included in the higher estimate have now been associated with the NWHI stock (Baird et al. 2013), such that the best estimate of population size for the MHI insular stock is the smaller estimate of 151 animals. However, it should be noted that even this smaller estimate may be positively-biased, because missed photo-ID matches were discovered after the analyses were complete (discussed in Oleson et al. 2010).

Minimum Population Estimate

The minimum population estimate for the MHI insular stock of false killer whales is the number of distinct individuals identified during ~~2008-2011~~ 2009-2012 photo-identification studies, or ~~129~~ 138 false killer whales (Baird, unpublished data). Recent mark-recapture estimates (Oleson et al. 2010) of abundance are known to have a positive bias of unknown magnitude due to missed matches, and therefore are not suitable for deriving a minimum abundance estimate.

Current Population Trend

Reeves et al. (2009) suggested that the MHI insular stock of false killer whales may have declined during the last two decades, based on sightings data collected near Hawaii using various methods between 1989 and 2007. Baird (2009) reviewed trends in sighting rates of false killer whales from aerial surveys conducted using consistent methodology around the main Hawaiian Islands between 1994 and 2003 (Mobley et al. 2000). Sighting rates during these surveys showed a statistically significant decline that could not be attributed to any weather or methodological changes. The Status Review of MHI insular false killer whales (Oleson et al. 2010) presented a quantitative analysis of extinction risk using a Population Viability Analysis (PVA). The modeling exercise was conducted to evaluate the probability of actual or near extinction, defined as a population reduced to fewer than 20 animals, given measured, estimated, or inferred information on population size and trends, and varying impacts of catastrophes, environmental stochasticity and Allee effects. All plausible models indicated the probability of decline to fewer than 20 animals within 75 years was greater than 20%. Though causation was not evaluated, all plausible models indicated the population has declined since 1989, at an average rate of -9% per year (95% probability intervals -5% to -12.5%), though some two-stage models suggested a lower rate of decline over the past decade (Oleson et al. 2010).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the MHI insular false killer whale stock is calculated as the minimum population estimate (~~129~~138) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.1 (for a stock listed as Endangered under the ESA and with minimum population size less than 1500 individuals; Taylor et al. 2000) resulting in a PBR of 0.3 false killer whales per year.

STATUS OF STOCK

The status of MHI insular stock false killer whales relative to OSP is unknown, although this stock appears to have declined during the past two decades (Oleson et al. 2010, Reeves et al. 2009; Baird 2009). MHI insular false killer whales are listed as "endangered" under the Endangered Species Act (1973) (77 FR 70915, 28 November, 2012). The Status Review report produced by the Biological Review Team (BRT) (Oleson et al. 2010) found that Hawaiian insular false killer whales are a Distinct Population Segment (DPS) of the global false killer whale taxon. Of the 29 identified threats to the population, the BRT considered the effects of small population size, including inbreeding depression and Allee effects, exposure to environmental contaminants (Ylitalo et al. 2009), competition for food with commercial fisheries (Boggs & Ito, 1993, Reeves et al. 2009), and hooking, entanglement, or intentional harm by fishers to be the most substantial threats to the population. The BRT concluded that Main Hawaiian Islands insular false killer whales were at high risk of extinction. Following additional information on the occurrence of another island-associated stock in the NWHI, the BRT reevaluated the DPS decision and concluded that the population still met the standard to be listed as a DPS (Oleson et al. 2012). Because MHI insular false killer whales are formally listed as "endangered" under the ESA, they are automatically considered as a "depleted" and "strategic" stock under the MMPA. Because the rate of mortality and serious injury to MHI insular false killer whales (0.9 animals per year) exceeds the PBR (0.3 animals per year), the total fishery mortality and serious injury for the MHI insular stock of false killer whales cannot be considered to be insignificant and approaching zero.
~~The estimated average annual human-caused mortality and serious injury from longline fisheries for this stock (0.1 animals per year) is less than the PBR (0.3), but is not approaching zero mortality and serious injury rate because it exceeds 10% of PBR (NMFS 2004). .~~

HAWAII PELAGIC STOCK

POPULATION SIZE

Analyses of a 2002 shipboard line-transect survey of the Hawaiian Islands EEZ resulted in an abundance estimate of 484 (CV = 0.93) false killer whales within the Hawaiian Islands EEZ outside of about 75 nmi of the main Hawaiian Islands (Barlow & Rankin 2007). A new abundance survey was completed in 2010 within the Hawaiian Islands EEZ and resulted in five on-effort detections of false killer whales attributed to the Hawaii pelagic stock. Analysis of the 2010 HICEAS shipboard line-transect data resulted in an abundance estimate of ~~1,503~~ 1,552 (CV=0.66) false killer whales outside of 40 km of the main Hawaiian Islands (Bradford et al. ~~2012~~ 2014). Bradford et al. (~~2012~~ 2014) reported that most (64%) false killer whale groups seen during the 2010 HICEAS survey were seen moving toward the vessel when detected by the visual observers. Together with an ~~significant~~ increase in sightings close to the trackline, these ~~this~~ behavioral data suggests vessel attraction is likely occurring and may be significant. Although Bradford et al. (~~2012~~ 2014) employed a half-normal model to minimize the effect of vessel attraction, the abundance estimate may ~~is likely~~ still be positively biased as a result of vessel attraction~~,~~ because groups originally outside of the survey strip, and therefore unavailable for observation by the visual survey team, may have moved within the survey strip and been sighted. There is some suggestion of such attractive movement within the acoustic data, though the extent of any bias created by this movement is unknown. A 2005 survey (Barlow and Rankin 2007) resulted in a separate abundance estimate of 906 (CV=0.68) false killer whales in international waters south of the Hawaiian Islands EEZ and within the EEZ of Johnston Atoll, but it is unknown how many of these animals might belong to the Hawaii pelagic stock.

Minimum Population Estimate

The minimum population size is calculated as the lower 20th percentile of the log-normal distribution (Barlow et al. 1995) of the 2010 abundance estimate for the Hawaiian Islands EEZ outside of 40 km from the main Hawaiian Islands (Bradford et al. ~~2012~~ 2014) or ~~906~~ 935 false killer whales. The minimum abundance estimate has not been corrected for vessel attraction and may be an over-estimate of minimum population size.

Current Population Trend

No data are available on current population trend. It is incorrect to interpret the increase in the abundance estimate from 2002 to 2010 as an increase in population size, given changes to the survey design in 2010 and the analytical framework specifically intended to better enumerate and account for overall group size, the low precision of each estimate, and a lack of understanding of the oceanographic processes that may drive the distribution of this stock over time. Further, estimation of the detection function for the 2002 and 2010 estimates relied on shared ~~very similar~~ data~~sets~~, such that the resulting abundance estimates are not statistically independent estimates and cannot be compared in standard statistical tests ~~of population size~~. Only a portion of the overall range of this population has been surveyed, precluding evaluation of abundance of the entire stock.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Hawaii pelagic stock of false killer whales is calculated as the minimum population estimate for the U.S. EEZ of the Hawaiian Islands (~~906~~ 935) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a stock of unknown status with a Hawaiian Islands EEZ mortality and serious injury rate CV <= 0.30; Wade and Angliss 1997), resulting in a PBR of ~~9.1~~ 9.4 false killer whales per year.

STATUS OF STOCK

The status of the Hawaii pelagic stock of false killer whales relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. No habitat issues are known to be of concern for this stock. This stock is not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. Following the NMFS Guidelines for Assessing Marine Mammal Stocks (NMFS 2005), the status of this transboundary stock of false killer whales is assessed based on the estimated abundance and estimates of mortality and serious injury within the U.S. EEZ of the Hawaiian Islands because estimates of human-caused mortality and serious injury from all U.S. and non-U.S. sources in high seas waters are not available, and because the geographic range of this stock beyond the Hawaiian Islands EEZ is poorly known. Because the rate of mortality and serious injury to false killer whales within the Hawaiian Islands EEZ (~~12.6~~ 13.0 animals per year) exceeds the PBR (~~9.1~~ 9.4 animals per year), this stock is considered a "strategic stock" under the MMPA. The total fishery mortality and serious injury for the Hawaii pelagic stock of false killer whales cannot be considered to be insignificant and approaching zero.

NORTHWESTERN HAWAIIAN ISLANDS STOCK

POPULATION SIZE

A 2010 line transect survey that included the waters surrounding the Northwestern Hawaiian Islands produced an estimate of 552 (CV = 1.09) false killer whales attributed to the Northwestern Hawaiian Islands stock (Bradford et al. ~~2012~~ 2014). This is the best available abundance estimate for false killer whales within the Northwestern Hawaiian Islands. Bradford et al. (~~2012~~ 2014) reported that most (64%) false killer whale groups seen during the 2010 HICEAS survey were seen moving toward the vessel when detected by the visual observers. Together with an ~~significant~~ increase in sightings close to the trackline, this behavioral data suggests vessel attraction is likely occurring and may be significant. Although Bradford et al. (~~2012~~ 2014) employed a half-normal model to minimize the effect of vessel attraction, because groups originally outside of the survey strip, and therefore unavailable for observation by the visual survey team, may have moved within the survey strip and been sighted. There is some suggestion of such attractive movement within the acoustic data, ~~the abundance estimate is likely still positively biased as a result of vessel attraction,~~ though the extent of any bias created by this movement is unknown.

Minimum Population Estimate

The minimum population size is calculated as the lower 20th percentile of the log-normal distribution (Barlow et al. 1995) of the 2010 abundance estimate for the Northwestern Hawaiian Islands stock (Bradford et al. ~~2012~~ 2014) or 262 false killer whales. This estimate has not been corrected for vessel attraction and may be positively biased.

Current Population Trend

No data are available on current population trend because there is only one estimate of abundance from 2010.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Northwestern Hawaiian Islands false killer whale stock is calculated as the minimum population estimate~~-~~ (262) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a stock of unknown status, Wade and Angliss 1997), resulting in a PBR of 2.6 false killer whales per year.

STATUS OF STOCK

~~The Northwestern Hawaiian Islands stock of false killer whales is not considered "strategic" under the 1994 amendments to the MMPA.~~ The status of false killer whales in Northwestern Hawaiian Islands waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Ylitalo et al. (2009) documented elevated levels of polychlorinated biphenyls (PCBs) in three of nine Hawaii insular false killer whales sampled, and biomass of some false killer whale prey species may have declined around the Northwestern Hawaiian Islands (Oleson et al. 2010, Boggs & Ito 1993, Reeves et al. 2009), though waters within the Papahānaumokuākea Marine National Monument have been closed to commercial longlining since 1991 and to other fishing since 2006. This stock is not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor as "depleted" under the MMPA. The estimated average annual human-caused mortality and serious injury from longline fisheries for this stock (0.4 animals per year) is less than the PBR (2.6), but is not approaching zero mortality and serious injury rate because it exceeds 10% of PBR (NMFS 2004). ~~The rate of fishery mortality and serious injury to Northwestern Hawaiian Islands false killer whales is unknown but may be insignificant and approaching zero,~~ However, given the current recognized geographic range of this stock is largely within the Marine National Monument, this stock is not likely exposed to high levels of fishing effort because commercial and recreational fishing is prohibited within Monument waters and longlines are excluded from the majority of the stock range. ~~Mortality and serious injury does not exceed the PBR (2.6) for this stock.~~

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Appendix 3. 2014 Pacific Marine Mammal Stock Assessment Reports. S=strategic stock; N=non-strategic stock. Shaded lines indicate reports revised in 2014. unk=unknown, undet=undetermined, n/a=not applicable.

Species	Stock Area	NMFS Center	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.	SWC	296,750	n/a	153,337	0.12	1	9,200	389	331	N	2007	2008	2011	2014
Harbor seal	California	SWC	30,968	n/a	27,348	0.12	1	1,641	43	30	N	2004	2009	2012	2014
Harbor seal	Oregon/Washington Coast	AKC	unk	unk	unk	0.12	1	unk	10.6	7.4	N	1999			2013
Harbor seal	Washington Northern Inland Waters	AKC	unk	unk	unk	0.12	1	unk	9.8	2.8	N	1999			2013
Harbor seal	Southern Puget Sound	AKC	unk	unk	unk	0.12	1	unk	3.4	1	N	1999			2013
Harbor seal	Hood Canal	AKC	unk	unk	unk	0.12	1	unk	0.2	0.2	N	1999			2013
Northern Elephant Seal	California breeding	SWC	179,000	n/a	81,368	0.12	1	4,882	8.8	4	N	2002	2005	2010	2014
Guadalupe Fur Seal	Mexico to California	SWC	7,408	n/a	3,028	0.12	0.5	91	0	0	S	1993			2000
Northern Fur Seal	California	AKC	12,844	n/a	6,722	0.12	1	403	2.6	≥0.4	N	2009	2010	2011	2013
Monk Seal	Hawaii	PIC	1,153	n/a	1,118	0.07	0.1	undet	≥2.6	≥1.0	S	2010	2011	2012	2014
Harbor porpoise	Morro Bay	SWC	2,917	0.41	2,102	0.04	0.5	21	≥0.6	≥0.6	N	2002	2007	2012	2013
Harbor porpoise	Monterey Bay	SWC	3,715	0.51	2,480	0.04	0.5	25	0	0	N	2002	2007	2011	2013
Harbor porpoise	San Francisco – Russian River	SWC	9,886	0.51	6,625	0.04	0.5	66	0	0	N	2002	2007	2011	2013
Harbor porpoise	Northern CA/Southern OR	SWC	35,769	0.52	23,749	0.04	1	475	≥0.6	≥0.6	N	2002	2007	2011	2013
Harbor porpoise	Northern Oregon/Washington Coast	AKC	21,487	0.44	15,123	0.04	0.5	151	≥3.0	≥3.0	N	2002	2010	2011	2013
Harbor porpoise	Washington Inland Waters	AKC	10,682	0.38	7,841	0.04	0.4	63	≥2.2	≥2.6	N	1996	2002	2003	2011
Dall's porpoise	California/Oregon/Washington	SWC	42,000	0.33	32,106	0.04	0.4	257	≥0.4	≥0.4	N	2001	2005	2008	2010
Pacific white-sided dolphin	California/Oregon/Washington	SWC	26,930	0.28	21,406	0.04	0.4	171	17.8	11.8	N	2001	2005	2008	2013
Risso's dolphin	California/Oregon/Washington	SWC	6,272	0.30	4,913	0.04	0.4	39	1.6	1.6	N	2001	2005	2008	2010
Common Bottlenose dolphin	California Coastal	SWC	323	0.13	290	0.04	0.5	2.4	0.2	0.2	N	2000	2004	2005	2008
Common Bottlenose dolphin	California/Oregon/Washington Offshore	SWC	1,006	0.48	684	0.04	0.4	5.5	≥2.0	≥2.0	N	2001	2005	2008	2013
Striped dolphin	California/Oregon/Washington	SWC	10,908	0.34	8,231	0.04	0.5	82	0	0	N	2001	2005	2008	2010
Common dolphin, short-beaked	California/Oregon/Washington	SWC	411,211	0.21	343,990	0.04	0.5	3,440	64	64	N	2001	2005	2008	2010
Common dolphin, long-beaked	California	SWC	107,016	0.42	76,224	0.04	0.4	610	13.8	13	N	2005	2008	2009	2012
Northern right whale dolphin	California/Oregon/Washington	SWC	8,334	0.40	6,019	0.04	0.4	48	4.8	3.6	N	2001	2005	2008	2010
Killer whale	Eastern North Pacific Offshore	SWC	240	0.49	162	0.04	0.5	1.6	0	0	N	2001	2005	2008	2010
Killer whale	Eastern North Pacific Southern Resident	NWC	82	n/a	82	0.032	0.1	0.13	0	0	S	2011	2012	2013	2014
Short-finned pilot whale	California/Oregon/Washington	SWC	760	0.64	465	0.04	0.4	4.6	0	0	N	2001	2005	2008	2010
Baird's beaked whale	California/Oregon/Washington	SWC	847	0.81	466	0.04	0.5	4.7	0	0	N	2001	2005	2008	2013
Mesoplodont beaked whales	California/Oregon/Washington	SWC	694	0.65	389	0.04	0.5	3.9	0	0	S	2001	2005	2008	2013
Cuvier's beaked whale	California/Oregon/Washington	SWC	6,590	0.55	4,481	0.04	0.5	45	0	0	S	2001	2005	2008	2013
Pygmy Sperm whale	California/Oregon/Washington	SWC	579	1.02	271	0.04	0.5	2.7	0	0	N	2001	2005	2008	2010
Dwarf sperm whale	California/Oregon/Washington	SWC	unk	unk	unk	0.04	0.5	undet	0	0	N	2001	2005	2008	2010
Sperm whale	California/Oregon/Washington	SWC	2,106	0.58	1,332	0.04	0.1	2.7	1.7	1.7	S	2001	2005	2008	2014
Gray whale	Eastern North Pacific	SWC	20,990	0.05	20,125	0.062	1.0	624	132	4.25	N	2009	2010	2011	2014
Gray whale	Western North Pacific (new report)	SWC	140	0.04	135	0.062	0.1	0.06	unk	unk	S			2011	2014
Humpback whale	California/Oregon/Washington	SWC	1,918	0.03	1,855	0.08	0.3	11.0	≥ 5.5	≥ 4.4	S	2009	2010	2011	2013
Blue whale	Eastern North Pacific	SWC	1,647	0.07	1,551	0.04	0.3	2.3	1.9	0	S	2005	2008	2011	2013
Fin whale	California/Oregon/Washington	SWC	3,051	0.18	2,598	0.04	0.3	16	2.2	0.6	S	2001	2005	2008	2013
Sei whale	Eastern North Pacific	SWC	126	0.53	83	0.04	0.1	0.17	0	0	S	2001	2005	2008	2010
Minke whale	California/Oregon/Washington	SWC	478	1.36	202	0.04	0.5	2.0	0	0	N	2001	2005	2008	2010
Rough-toothed dolphin	Hawaii	SWC	6,288	0.39	4,581	0.04	0.5	46	unk	unk	N		2002	2010	2013
Rough-toothed dolphin	American Samoa	PIC	unk	unk	unk	0.04	0.5	unk	unk	unk	unk	n/a	n/a	n/a	2010

Appendix 3. 2014 Pacific Marine Mammal Stock Assessment Reports. S=strategic stock; N=non-strategic stock. Shaded lines indicate reports revised in 2014. unk=unknown, undet=undetermined, n/a=not applicable.

Species	Stock Area	NMFS Center	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
Risso's dolphin	Hawaii	SWC	7,256	0.41	5,207	0.04	0.5	42	0.6	0.6	N		2002	2010	2013
Common Bottlenose dolphin	Hawaii Pelagic	SWC	5,950	0.59	3,755	0.04	0.5	38	0.2	0.2	N		2002	2010	2013
Common Bottlenose dolphin	Kaua'I and Ni'ihau	SWC	184	0.11	168	0.04	0.5	1.7	unk	unk	N	2003	2004	2005	2013
Common Bottlenose dolphin	O'ahu	SWC	743	0.54	485	0.04	0.5	4.9	unk	unk	N	2002	2003	2006	2013
Common Bottlenose dolphin	4 Islands Region	SWC	191	0.24	156	0.04	0.5	1.6	unk	unk	N	2002	2003	2006	2013
Common Bottlenose dolphin	Hawaii Island	SWC	128	0.13	115	0.04	0.5	1.1	unk	unk	N	2002	2003	2006	2013
Pantropical Spotted dolphin	Hawaii Pelagic	PIC	15,917	0.40	11,508	0.04	0.5	115.0	0	0	N		2002	2010	2013
Pantropical Spotted dolphin	O'ahu	PIC	unk	unk	unk	0.04	0.5	undet	unk	unk	N			n/a	2013
Pantropical Spotted dolphin	4 Islands Region	PIC	unk	unk	unk	0.04	0.5	undet	unk	unk	N			n/a	2013
Pantropical Spotted dolphin	Hawaii Island	PIC	unk	unk	unk	0.04	0.5	undet	unk	unk	N			n/a	2013
Spinner dolphin	Hawaii Pelagic	PIC	unk	unk	unk	0.04	0.5	undet	0	0	N		2002	2010	2013
Spinner dolphin	Hawaii Island	PIC	820	0.04	793	0.04	0.5	7.9	unk	unk	N	1994	2003	2011	2013
Spinner dolphin	Oahu / 4 Islands	PIC	355	0.09	329	0.04	0.5	3.3	unk	unk	N	1993	1998	2007	2013
Spinner dolphin	Kaua'I / Ni'ihau	PIC	601	0	509	0.04	0.5	5.1	unk	unk	N	1995	1998	2005	2013
Spinner dolphin	Kure / Midway	PIC	unk	unk	unk	0.04	0.5	undet	unk	unk	N		1998	2010	2013
Spinner dolphin	Pearl and Hermes Reef	PIC	unk	unk	unk	0.04	0.5	undet	unk	unk	N			n/a	2013
Spinner dolphin	American Samoa	PIC	unk	unk	unk	0.04	0.5	unk	unk	unk	unk			n/a	2010
Striped dolphin	Hawaii Pelagic	PIC	20,650	0.36	15,391	0.04	0.5	154	unk	unk	N		2002	2010	2013
Fraser's dolphin	Hawaii	PIC	16,992	0.66	10,241	0.04	0.5	102	0	0	N		2002	2010	2010
Melon-headed whale	Hawaiian Islands	PIC	5,794	0.20	4,904	0.04	0.5	49	0	0	N		2002	2010	2013
Melon-headed whale	Kohala Resident	PIC	447	0.12	404	0.04	0.5	4.0	0	0	N			2009	2013
Pygmy killer whale	Hawaii	PIC	3,433	0.52	2,274	0.04	0.5	23.0	0	0	N		2002	2010	2013
False killer whale	Northwestern Hawaiian Islands	PIC	552	1.09	262	0.04	0.5	2.6	0.4	0.4	N			2010	2014
False killer whale	Hawaii Pelagic	PIC	1,552	0.66	935	0.04	0.5	9.4	13	13	S		2002	2010	2014
False killer whale	Palmyra Atoll	PIC	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	PIC	151	0.20	129	0.04	0.1	0.3	0.9	0.9	S	2009	2010	2011	2014
False killer whale	American Samoa	PIC	unk	unk	unk	0.04	0.5	unk	unk	unk	unk	n/a	n/a	n/a	2010
Killer whale	Hawaii	PIC	101	1.00	50	0.04	0.5	1.0	0	0	N		2002	2010	2013
Pilot whale, short-finned	Hawaii	PIC	12,422	0.43	8,782	0.04	0.4	70	0.1	0.1	N		2002	2010	2013
Blainville's beaked whale	Hawaii Pelagic	PIC	2,338	1.13	1,088	0.04	0.5	11.0	0	0	N		2002	2010	2013
Longman's Beaked Whale	Hawaii	PIC	4,571	0.65	2,773	0.04	0.5	28.0	0	0	N		2002	2010	2013
Cuvier's beaked whale	Hawaii Pelagic	PIC	1,941	0.70	1,142	0.04	0.5	11.4	0	0	N		2002	2010	2013
Pygmy sperm whale	Hawaii	PIC	unk	unk	unk	0.04	0.5	undet	0	0	N		2002	2010	2013
Dwarf sperm whale	Hawaii	PIC	unk	unk	unk	0.04	0.5	undet	0	0	N		2002	2010	2013
Sperm whale	Hawaii	PIC	3,354	0.34	2,539	0.04	0.1	10.2	0.7	0.7	S		2002	2010	2013
Blue whale	Central North Pacific	PIC	81	1.14	38	0.04	0.1	0.1	0	0	S		2002	2010	2013
Fin whale	Hawaii	PIC	58	1.12	27	0.04	0.1	0.1	0	0	S		2002	2010	2013
Bryde's whale	Hawaii	PIC	798	0.28	633	0.04	0.5	6.3	0	0	N		2002	2010	2013
Sei whale	Hawaii	PIC	178	0.90	93	0.04	0.1	0.2	0.2	0.2	S		2002	2010	2013
Minke whale	Hawaii	PIC	unk	unk	unk	0.04	0.5	undet	0	0	N		2002	2010	2013
Humpback whale	American Samoa	SWC	unk	unk	150	0.106	0.1	0.4	0	0	S	2006	2007	2008	2009
Sea Otter	Southern	USFWS	2,826	n/a	2,723	0.06	0.1	8	≥0.8	≥0.8	S	2006	2007	2008	2008
Sea Otter	Washington	USFWS	n/a	n/a	1,125	0.2	0.1	11	≥0.2	≥0.2	N	2006	2007	2008	2008