NOAA Technical Memorandum NMFS-AFSC-421

Alaska Marine Mammal Stock Assessments, 2020

M. M. Muto, V. T. Helker, B. J. Delean, N. C. Young, J. C. Freed, R. P. Angliss, N. A. Friday, P. L. Boveng, J. M. Breiwick, B. M. Brost, M. F. Cameron, P. J. Clapham, J. L. Crance, S. P. Dahle, M. E. Dahlheim, B. S. Fadely, M. C. Ferguson, L. W. Fritz, K. T. Goetz, R. C. Hobbs, Y. V. Ivashchenko, A. S. Kennedy, J. M. London, S. A. Mizroch, R. R. Ream, E. L. Richmond, K. E. W. Shelden, K. L. Sweeney, R. G. Towell, P. R. Wade, J. M. Waite, and A. N. Zerbini

July 2021

U.S. DEPARTMENT OF COMMERCE

National Oceanic and Atmospheric Administration
National Marine Fisheries Service
Alaska Fisheries Science Center

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This document should be cited as follows:

Muto, M. M., V. T. Helker, B. J. Delean, N. C. Young, J. C. Freed, R. P. Angliss, N. A. Friday, P. L. Boveng, J. M. Breiwick, B. M. Brost, M. F. Cameron, P. J. Clapham, J. L. Crance, S. P. Dahle, M. E. Dahlheim, B. S. Fadely, M. C. Ferguson, L. W. Fritz, K. T. Goetz, R. C. Hobbs, Y. V. Ivashchenko, A. S. Kennedy, J. M. London, S. A. Mizroch, R. R. Ream, E. L. Richmond, K. E. W. Shelden, K. L. Sweeney, R. G. Towell, P. R. Wade, J. M. Waite, and A. N. Zerbini. 2021. Alaska marine mammal stock assessments, 2020. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-421, 398 p.

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NOAA Technical Memorandum NOAA-TM-AFSC-421

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ERRATA

Issuance of a Revised Version of the 2020 Alaska Marine Mammal Stock Assessment Reports

NOAA Technical Memorandum NMFS-AFSC-421

April 2022

Following publication of the 2020 Alaska Marine Mammal Stock Assessment Reports (SARs) in July 2021, NMFS recognized that it had not consulted with the Alaska Beluga Whale Committee (ABWC) over the change in the classification of the Eastern Bering Sea (EBS) beluga whale stock from "non-strategic" to "strategic." This failure to adhere to the requirement for consultation as established in the 1999 ABWC-NMFS co-management agreement led NMFS to temporarily withdraw the 2020 EBS beluga whale SAR to allow time for such a consultation, which, based on scientific issues raised during the consultation, subsequently resulted in the permanent withdrawal of the 2020 EBS beluga whale SAR. NMFS is now issuing a revised version of the 2020 Alaska Marine Mammal Stock Assessment Reports with the following changes:

- The 2020 EBS beluga whale SAR was replaced with the previous version of SAR, which was last updated in 2017.
- A note was added to the Beaufort Sea, Eastern Chukchi Sea, and Bristol Bay beluga whale SARs to indicate that NMFS is evaluating whether scientific issues raised by the ABWC regarding the EBS beluga whale stock may also be applicable to those SARs, and that any resulting changes will be reflected in a future SAR.
- The Preface, Contents, and Appendix 1 were updated to reflect that the EBS beluga whale Stock Assessment Report was not revised in 2020.
- Appendix 2 was updated to reflect information from the 2017 EBS beluga whale SAR.

The citation for the publication will remain the same:

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PREFACE

On 30 April 1994, Public Law 103-238 was enacted allowing significant changes to provisions within the Marine Mammal Protection Act (MMPA). Interactions between marine mammals and commercial fisheries are addressed under three new sections. This new regime replaced the interim exemption that had regulated fisheries-related incidental takes since 1988. Section 117, Stock Assessments, required the establishment of three regional scientific review groups to advise and report on the status of marine mammal stocks within Alaska waters, along the Pacific Coast (including Hawaii), and along the Atlantic Coast (including the Gulf of Mexico). This report provides information on the marine mammal stocks of Alaska under the jurisdiction of the National Marine Fisheries Service.

Each stock assessment includes, when available, a description of the stock's geographic range; a minimum population estimate; current population trends; current and maximum net productivity rates; optimum sustainable population levels and allowable removal levels; estimates of annual human-caused mortality and serious injury through interactions with commercial, recreational, and subsistence fisheries, takes by subsistence hunters, and other human-caused events (e.g., entanglement in marine debris, ship strikes); and habitat concerns. The commercial fishery interaction data will be used to evaluate the progress of each fishery towards achieving the MMPA's goal of zero fishery-related mortality and serious injury of marine mammals.

The Stock Assessment Reports should be considered working documents, as they are updated as new information becomes available. The Alaska Stock Assessment Reports were originally developed in 1995 (Small and DeMaster 1995). Revisions have been published for the following years: 1996 (Hill et al. 1997), 1998 (Hill and DeMaster 1998), 1999 (Hill and DeMaster 1999), 2000 (Ferrero et al. 2000), 2001 (Angliss et al. 2001), 2002 (Angliss and Lodge 2002), 2003 (Angliss and Lodge 2004), 2005 (Angliss and Outlaw 2005), 2006 (Angliss and Outlaw 2007), 2007 (Angliss and Outlaw 2008), 2008 (Angliss and Allen 2009), 2009 (Allen and Angliss 2010), 2010 (Allen and Angliss 2011), 2011 (Allen and Angliss 2012), 2012 (Allen and Angliss 2013), 2013 (Allen and Angliss 2014), 2014 (Allen and Angliss 2015), 2015 (Muto et al. 2016), 2016 (Muto et al. 2017), 2017 (Muto et al. 2018), 2018 (Muto et al. 2019), and 2019 (Muto et al. 2020). Each Stock Assessment Report is designed to stand alone and is updated as new information becomes available. The MMPA requires Stock Assessment Reports to be reviewed annually for stocks designated as strategic, annually for stocks where there is significant new information available, and at least once every 3 years for all other stocks. NMFS reviewed new information for 28 stocks in the Alaska Region in 2019-2020 and revised 22 Stock Assessment Reports under NMFS' jurisdiction: all 15 strategic stocks (Western U.S. Steller sea lions; northern fur seals; bearded seals; ringed seals; Cook Inlet beluga whales; AT1 Transient killer whales; Southeast Alaska, Gulf of Alaska, and Bering Sea stocks of harbor porpoise; sperm whales; Western North Pacific and Central North Pacific stocks of humpback whales; fin whales; North Pacific right whales; and bowhead whales) and 7 non-strategic stocks (spotted seals; ribbon seals; Beaufort Sea, Eastern Chukchi Sea, and Bristol Bay stocks of beluga whales; and Gulf of Alaska, Aleutian Islands, and Bering Sea Transient and West Coast Transient stocks of killer whales). The Stock Assessment Reports for all of the Alaska stocks, however, are included in this document to provide a complete reference. Those sections of each Stock Assessment Report containing substantial changes are listed in Appendix 1. The authors solicit any new information or comments which would improve future Stock Assessment Reports.

New abundance estimates were calculated for the following Alaska stocks in the 2020 Stock Assessment Reports. For explanations of why estimates have changed, see the individual report for each stock:

- Western U.S. Steller sea lions: The updated best model estimated count in 2019, derived from aerial photographic and land-based surveys in 2018 and 2019, is 52,932 sea lions. This is a decrease from the previous estimate of 53,624. The model estimated count is not a total population abundance estimate because the count has not been corrected for animals at sea during the surveys or for pups that are born before or die after the surveys.
- Eastern Pacific northern fur seals: The updated best abundance estimate, derived from counts on Sea Lion Rock in 2014, St. Paul and St. George Islands in 2014, 2016, and 2018, and Bogoslof Island in 2015, is 608,143 northern fur seals. This is a decrease from the previous estimate of 620,660.
- Eastern Chukchi Sea beluga whales: The updated best abundance estimate, derived from aerial surveys in summer 2017 in the Beaufort Sea (in an area and time period in which the Beaufort Sea and Eastern Chukchi Sea stocks do not overlap, as determined from satellite-tag data), is 13,305 beluga whales. This is a decrease from the previous estimate of 20,752, derived from aerial surveys of the northeastern Chukchi and Alaska Beaufort seas in 2012.
- Bristol Bay beluga whales: The updated best abundance estimate, derived from aerial surveys in 2016, is 2,040 beluga whales. This is an increase from the previous estimate of 1,926.
- Cook Inlet beluga whales: The updated best estimate of abundance in 2018, derived using a new analytical method on aerial survey data from 2014, 2016, and 2018, is 279 beluga whales. This is a decrease from the previous estimate of 327.

- West Coast Transient killer whales: The updated best estimate of abundance in 2018, derived from an analysis of photo-identification data from 1958 to 2018 for a subset of whales in British Columbia waters, is 349 killer whales. This is an increase from the previous estimate of 243, derived for a subset of whales in the inside waters of Southeast Alaska, British Columbia, and northern Washington.
- Bering Sea harbor porpoise: An abundance estimate for harbor porpoise in the eastern Bering Sea, derived from vessel surveys in association with pollock stock assessment surveys in 2008, is 5,713 harbor porpoise. However, this estimate is for only a small portion of the range of this stock and, because the survey data are more than 8 years old, the minimum population estimate (N_{MIN}) is now considered unknown and the potential biological removal (PBR) is considered undetermined.

The U.S. Fish and Wildlife Service (USFWS) has management authority for polar bears, sea otters, and walruses. Copies of the stock assessments for these species are included in Appendix 4 of this document for your convenience.

Ideas and comments from the Alaska Scientific Review Group (SRG) have significantly improved this document from its draft form. The authors wish to express their gratitude for the thorough reviews and helpful guidance provided by the Alaska Scientific Review Group members: John Citta, Beth Concepcion, Thomas Doniol-Valcroze, Mike Miller, Greg O'Corry-Crowe (Co-Chair in 2019-2020), Lorrie Rea, Megan Peterson (Co-Chair in 2019-2020), Eric Regehr, and Kate Stafford. We would also like to acknowledge the contributions from the NMFS Alaska Regional Office and the Communications Program of the Alaska Fisheries Science Center.

The information contained within the individual Stock Assessment Reports stems from a variety of sources. Where feasible, we have attempted to use only published material. When citing information contained in this document, authors are reminded to cite the original publications, when possible.

CONTENTS*

*NMFS Stock Assessment Reports and Appendices revised in 2020 are in boldface.

Revised 12/30/2020

STELLER SEA LION (*Eumetopias jubatus*): Western U.S. Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Steller sea lions range along the North Pacific Rim from northern Japan to California (Loughlin et al. 1984) (Fig. 1). Outside of the breeding season (late May to July), large numbers of individuals, especially juveniles and males, disperse widely, probably to access seasonally important prey resources (Jemison et al. 2018). This results in marked seasonal patterns of abundance in some parts of the range and potential for intermixing of animals that were born in different regions (Sease and York 2003; Baker et al. 2005; Jemison et al. 2013, 2018; Hastings et al. 2019). During the breeding season, sea lions, especially adult females, typically return to their natal rookery or a nearby breeding rookery to breed and pup (Raum-Suryan et al. 2002, Hastings et al. 2017).

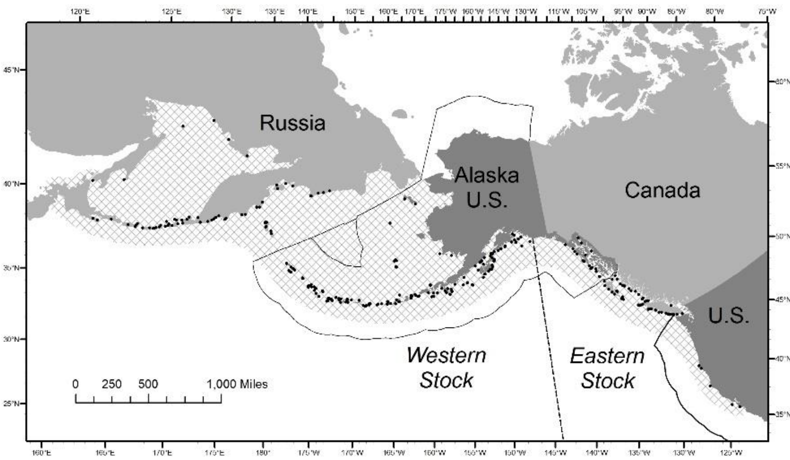

Figure 1. Generalized distribution (crosshatched area) of Steller sea lions in the North Pacific and major U.S. haulouts and rookeries (50 CFR 226.202, 27 August 1993), as well as active Asian and Canadian (British Columbia) haulouts and rookeries (points: Burkanov and Loughlin 2005, Olesiuk 2008). A black dashed line (144°W) indicates the stock boundary (Loughlin 1997) and a black line delineates the U.S. Exclusive Economic Zone.

Loughlin (1997) considered the following information when classifying stock structure based on the phylogeographic approach of Dizon et al. (1992): 1) Distributional data: geographic distribution continuous, yet a high degree of natal site fidelity and low (<10%) exchange rate of breeding animals among rookeries; 2) Population response data: substantial differences in population dynamics (York et al. 1996); 3) Phenotypic data: differences in pup mass (Merrick et al. 1995, Loughlin 1997); and 4) Genotypic data: substantial differences in mitochondrial DNA (Bickham et al. 1996). Based on this information, two distinct population segments (DPSs) of Steller sea lions were recognized in the U.S.: the Eastern DPS, which includes animals born east of Cape Suckling, Alaska (144°W), and the Western DPS, which includes animals born at and west of Cape Suckling (Loughlin 1997; Fig. 1). However, there is regular movement of Steller sea lions, especially juveniles and males outside the breeding season, between the Western DPS (males and females equally) and the Eastern DPS (almost exclusively males) across the DPS boundary (Jemison et al. 2013, 2018; Hastings et al. 2019). In this report, the Western DPS is equivalent to the Western stock and the Eastern DPS is equivalent to the Eastern stock.

Mixing of mostly breeding females occurred between Prince William Sound and northern Southeast Alaska, beginning in the 1990s (Gelatt et al. 2007; Jemison et al. 2013, 2018; O'Corry-Crowe et al. 2014; Rehberg et al. 2018). In 1998 a single Steller sea lion pup was observed on Graves Rock just north of Cross Sound in Southeast Alaska, and within 15 years (2013) pup counts increased to 551 (DeMaster 2014). Movements of animals marked as pups in both stocks corroborate the extensive genetic research findings for a strong separation between the two currently recognized stocks (Jemison et al. 2013, 2018). Mitochondrial and microsatellite analysis of pup tissue samples collected at Graves Rock in 2002 revealed that approximately 70% of the pups had mtDNA haplotypes that were consistent with those found in the Western stock (Gelatt et al. 2007). Similarly, a rookery to the south on the White Sisters Islands, where pups were first noted in 1990, was also sampled in 2002 and approximately 45% of those pups had Western stock haplotypes (O'Corry-Crowe et al. 2014). Hastings et al. (2019) estimated that a minimum of 38% and 13% of animals in the North Outer Coast-Glacier Bay and Lynn Canal-Frederick Sound regions in northern Southeast Alaska, respectively, carry genetic information unique to the Western stock. Collectively, this information demonstrates that these two most recently established rookeries in northern Southeast Alaska were partially to predominately established by Western stock females (Jemison et al. 2013, 2018; O'Corry-Crowe et al. 2014; Rehberg et al. 2018; Hastings et al. 2019).

O'Corry-Crowe et al. (2014) concluded that the results of their study of the genetic characteristics of pups born on these new rookeries "demonstrates that resource limitation may trigger an exodus of breeding animals from declining populations, with substantial impacts on distribution and patterns of genetic variation." Jemison et al. (2018) also found that movement of Prince William Sound females east to these rookeries was negatively correlated with density: the population's declines prior to the early 2000s likely spurred these animals to move east in search of better foraging opportunities. This movement also revealed that this event is rare because colonists dispersed across an evolutionary boundary, suggesting that the causative factors behind recent declines are unusual or of larger magnitude than normally occur (O'Corry-Crowe et al. 2014). Thus, although recent colonization events in the northern part of the Eastern stock indicate movement of Western sea lions (especially adult females) into this area, the mixed part of the range remains geographically distinct (Jemison et al. 2013, 2018), and the discreteness between the Eastern and Western stocks remains. Movement of Western stock sea lions south of these rookeries and Eastern stock sea lions moving to the west is less common (Jemison et al. 2013, O'Corry-Crowe et al 2014).

Hybridization among subspecies and species along a contact zone such as a stock boundary is not unexpected as the ability to interbreed is an ancestral condition, whereas reproductive isolation would be considered a recently derived condition. As stated by NMFS and the U.S. Fish and Wildlife Service (USFWS) in a 1996 response to a previous comment regarding stock discreteness policy (61 FR 47222), "The Services do not consider it appropriate to require absolute reproductive isolation as a prerequisite to recognizing a distinct population segment" or stock. The fundamental concept overlying this distinctiveness is the collection of morphological, ecological, behavioral, and genetic evidence for stock differences initially described by Bickham et al. (1996) and Loughlin (1997) and supported by Baker et al. (2005), Harlin-Cognato et al. (2006), Hoffman et al. (2006, 2009), O'Corry-Crowe et al. (2006), and Phillips et al. (2009, 2011).

Steller sea lions that breed in Asia are considered part of the Western stock in the 2008 Steller sea lion Recovery Plan (NMFS 2008). Steller sea lions seasonally inhabit coastal waters of Japan in the winter and breeding rookeries of Western stock animals outside of the U.S. are currently only located in Russia (Burkanov and Loughlin 2005). Analyses of genetic data differ in their interpretation of separation between Asian and Alaska sea lions. Based on analysis of mitochondrial DNA, Baker et al. (2005) found evidence of a genetic split between the Commander Islands (Russia) and Kamchatka that would include Commander Island sea lions within the Western U.S. stock and animals west of there in an Asian stock. However, Hoffman et al. (2006) did not support an Asian/Western stock split based on their analysis of nuclear microsatellite markers indicating high rates of male gene flow. Berta and Churchill (2012) concluded that a putative Asian stock is "not substantiated by microsatellite data since the Asian stock groups with the Western stock." All genetic analyses (Baker et al. 2005; Harlin-Cognato et al. 2006; Hoffman et al. 2006, 2009; O'Corry-Crowe et al. 2006) confirm a strong separation between Western and Eastern stocks, and O'Corry-Crowe et al. (2006) identified structure at the level of different oceanic regions within the Aleutian Islands. There may be sufficient morphological differentiation to support elevating the two recognized stocks to subspecies (Phillips et al. 2009), although a review by Berta and Churchill (2012) characterized the status of these subspecies assignments as "tentative" and requiring further attention before their status can be determined. Work by Phillips et al. (2011) addressed the effect of climate change, in the form of glacial events, on the evolution of Steller sea lions and reported that the effective population size at the time of the event determines the impact of change on the population. The results suggested that during historic glacial periods, dispersal events were correlated with historically low effective population sizes, whereas range fragmentation type events were correlated with larger effective population sizes. This work again reinforced the stock delineation concept by noting that ancient population subdivision likely led to the sequestering of most mtDNA haplotypes as stock or subspecies-specific (Phillips et al. 2011).

POPULATION SIZE

The Western stock of Steller sea lions decreased from 220,000 to 265,000 animals in the late 1970s to less than 50,000 in 2000 (Loughlin et al. 1984, Loughlin and York 2000, Burkanov and Loughlin 2005). Since 2003, the abundance of the Western stock has increased, but there has been considerable regional variation in trend (Sease and Gudmundson 2002; Burkanov and Loughlin 2005; Fritz et al. 2013, 2016). Abundance surveys to count Steller sea lions are conducted in late June through mid-July starting approximately 10 days after the mean pup birth dates in the survey area (4-14 June) after approximately 95% of all pups are born (Pitcher et al. 2001, Kuhn et al. 2017). Modeled counts and trends are reported for the total Western stock in Alaska and the six regions (eastern, central, and western Gulf of Alaska and eastern, central, and western Aleutian Islands) that compose this geographic range. The boundaries for the six regions were identified based on metapopulation analysis of survey count data collected from 1976 to 1994 (York et al. 1996).

NMFS uses raw counts collected during the period from 1978 through 2019 to model counts and annual rates of change of non-pups and pups for regional aggregations using agTrend (Johnson and Fritz 2014). Using this model produces two types of count estimates: predicted and realized counts. Predicted counts are used to estimate trends and account for both observation and process errors. Realized counts use the standardized variance of raw counts at each site throughout the time series to estimate survey counts we could expect to collect if we had completely surveyed all sites. Therefore, the more complete the survey, the more similar raw counts are to realized counts, which is evident by smaller confidence intervals. Modeled counts, like raw counts, do not account for animals at sea; however, pup counts are considered a census of live pups as they are generally not in the water during the survey period.

Demographic multipliers (e.g., pup production multiplied by 4.5) and corrections for proportions of each age-sex class that are hauled out during the day in the breeding season (when aerial surveys are conducted) have been proposed as methods to estimate total population size from pup and/or non-pup counts (Calkins and Pitcher 1982, Higgins et al. 1988, Milette and Trites 2003, Maniscalco et al. 2006). There are several factors which make using demographic multipliers problematic when applied to counts of Western Steller sea lions in Alaska, including the lack of vital (survival and reproductive) rate information for the western and central Aleutian Islands, the large variability in abundance trends across the range (see Current Population Trend section below and Pitcher et al. 2007), and the large uncertainties related to reproductive status and foraging conditions that affect proportions hauled out (see review in Holmes et al. 2007).

The most recent comprehensive aerial photographic and land-based surveys of Western Steller sea lions in Alaska were conducted during the 2018 (Aleutian Islands west of Shumagin Islands) and 2019 (Southeast Alaska and Gulf of Alaska east of Shumagin Islands) breeding seasons (Sweeney et al. 2018, 2019). The Western Steller sea lion pup and non-pup model-predicted counts in Alaska in 2019 were 12,581 (95% credible interval of 11,308-14,051) and 40,351 (35,886-44,884), respectively.

Methods used to survey Steller sea lions in Russia differ from those used in Alaska, with less use of aerial photography and more use of skiff surveys and cliff counts for non-pups and ground counts for pups (Burkanov 2018a). Since 2015, the use of drones has allowed more survey effort to collect aerial imagery, similar to survey methods used for the Alaska range (Burkanov 2018a). The most recent total count of live pups on rookeries in Russia is available from counts conducted in 2016 and 2017, which totaled 5,629 pups, about 11% more than the 5,073 pups counted in 2013 and 2015 (Burkanov 2018b). Rookery pup counts represent more than 95% of pup counts at all sites (including haulouts) but are underestimates of total pup production. Modeled counts and trends are reported for non-pups only (there are not robust data available to model pup counts) for the six regions (Commander Islands, east Kamchatka, Kuril Islands, northern part of Sea of Okhotsk, Sakhalin Island, and western Bering Sea) that compose the geographic range in Russia (Fig. 2). In 2017, the non-pup count was modeled to be 13,691 (95% credible interval of 12,225-15,133) in Russia (Burkanov 2017, Johnson 2018).

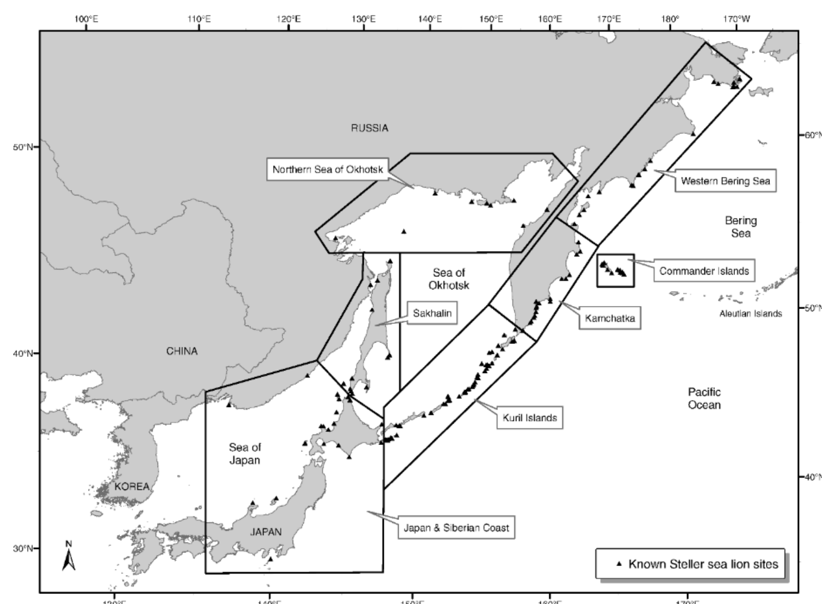

Figure 2. Steller sea lion survey regions along the Asian coast (Burkanov and Loughlin 2005).

Minimum Population Estimate

The minimum population estimate (N_{MIN}) can be defined by the 20th percentile of a log-normal distribution based on a population abundance estimate for the stock (Wade 1994). Because current population size (N) and a pup multiplier to estimate N are not known we cannot produce an abundance estimate. With agTrend we can produce a sum of non-pup and pup modeled counts, which don't account for non-pups at sea, or animals that are born or die after the survey. Therefore, the summed count estimate is lower than an abundance estimate and we should not use the 20th percentile of this number. We use the best estimate of the total count of Western Steller sea lions in Alaska as the minimum population estimate (N_{MIN}). The agTrend model (Johnson and Fritz 2014) was used to estimate Western Steller sea lion pup and non-pup counts of 12,581 and 40,351, respectively, in Alaska in 2019 (Sweeney et al. 2019). These sum to 52,932, which will be used as the N_{MIN} for the U.S. portion of the Western stock of Steller sea lions (NMFS 2016).

Current Population Trend

The first reported trend counts (sums of counts at consistently surveyed, large sites used to examine population trends) of Steller sea lions in Alaska were made in 1956-1960. Those counts indicated that there were at least 140,000 (no correction factor applied) sea lions in the Gulf of Alaska and Aleutian Islands (Merrick et al. 1987). Subsequent surveys indicated a major population decrease, first detected in the eastern Aleutian Islands in the mid-1970s (Braham et al. 1980). Counts from 1976 to 1979 totaled about 110,000 sea lions (no correction factor applied). The decline appears to have spread eastward to Kodiak Island during the late 1970s and early 1980s, and then westward to the central and western Aleutian Islands during the early and mid-1980s (Merrick et al. 1987, Byrd 1989). During the late 1980s, counts in Alaska overall declined at approximately 15% per year (NMFS 2008) which prompted the listing (in 1990) of the species as threatened range-wide under the Endangered Species Act (ESA). Continued declines in counts of Western Steller sea lions in Alaska in the 1990s (Sease et al. 2001) led NMFS to change the ESA listing status to endangered in 1997 (NMFS 2008). Surveys in Alaska in 2002, however, were the first to note an increase in counts, which suggested that the overall decline of Western Steller sea lions stopped in the early 2000s (Sease and Gudmundson 2002).

Johnson and Fritz's (2014) agTrend model estimated regional and overall trends in counts of pups and non-pups in Alaska using data collected at all sites with at least two non-zero counts, rather than relying solely on counts at "trend" sites (also see Fritz et al. 2013, 2016). Using agTrend, modeled count data from 1978 to 2019 were used to produce trends for the total Western DPS in Alaska, east of Samalga Pass, and the central, western, and eastern Gulf of Alaska regions.

Model results indicated that pup and non-pup counts of Western stock Steller sea lions in Alaska were at their lowest levels in 2002 and have increased at 1.63% y^{-1} and 1.82% y^{-1}, respectively, between 2002 and 2019 (Table 1; Fig. 3; Sweeney et al. 2019). However, there are strong regional differences across the range in Alaska, with positive trends in the Gulf of Alaska and the eastern Aleutian Islands region, including eastern Bering Sea (east of Samalga Pass, ~170°W), and generally negative trends to the west of Samalga Pass, in the central and western Aleutian Islands (Table 1; Figs. 4 and 5).

Table 1. Trends (annual rates of change expressed as % y^{-1} with 95% credible interval) in counts of Western Steller sea lion pups and non-pups (adults and juveniles) in Alaska, by regional areas. The rates reported for the Western DPS in Alaska; east of Samalga Pass; and eastern, central, and western Gulf of Alaska were calculated for the period from 2002 to 2019 (Sweeney et al. 2019). The rates reported for west of Samalga Pass and eastern, central, and western Aleutian Islands were calculated for the period from 2002 (when the Western DPS as a whole began to rebound) to 2018 (Sweeney et al. 2018).

Region	**Latitude Range**	**Pups**				**Non-pups**		
		Trend	**-95%**	**+95%**		**Trend**	**-95%**	**+95%**
Western DPS in Alaska	144°W-172°E	1.63	1.12	2.16		1.82	1.29	2.38
East of Samalga Pass	144°-170°W	2.90	2.37	3.53		2.71	2.05	3.35
Eastern Gulf of Alaska	144°-150°W	2.68	1.08	4.36		3.32	1.42	5.24
Central Gulf of Alaska	150°-158°W	3.08	1.76	4.35		3.40	2.53	4.32
Western Gulf of Alaska	158°-163°W	3.37	2.25	4.52		2.77	1.47	4.01
Eastern Aleutian Islands	163°-170°W	2.54	1.67	3.46		1.76	0.50	3.07
West of Samalga Pass	170°W-172°E	-2.08	-3.13	-0.79		-1.22	-2.20	-0.25
Central Aleutian Islands	170°W-177°E	-1.60	-2.75	-0.21		-0.53	-1.64	0.50
Western Aleutian Islands	172°-177°E	-6.47	-7.42	-5.57		-6.47	-7.81	-5.21

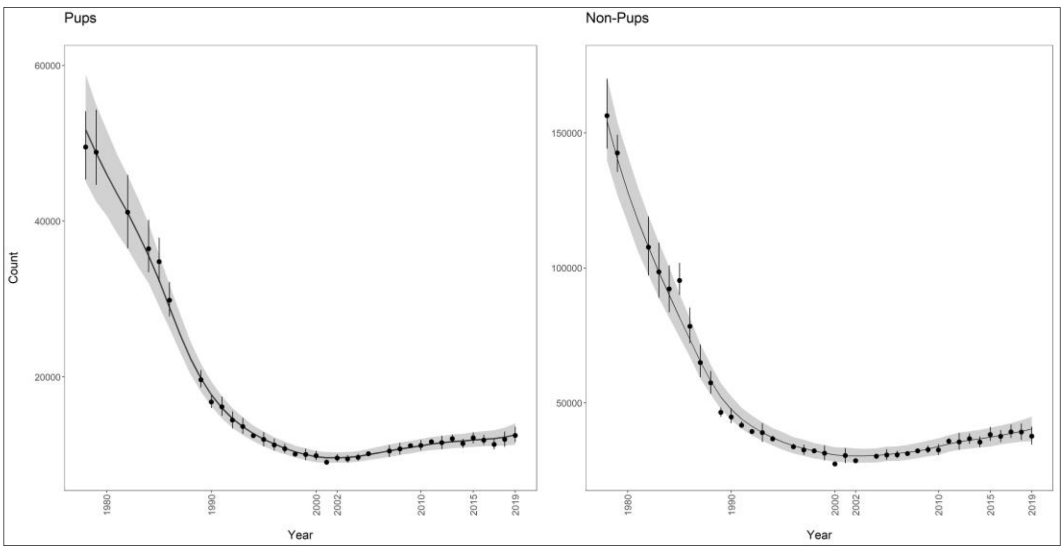

Figure 3. Realized and predicted counts of Western Steller sea lion pups (left) and non-pups (right) in Alaska, from 1978 to 2019. Realized counts are represented by points and vertical lines (95% credible intervals). Predicted counts are represented by the black line surrounded by the gray 95% credible interval.

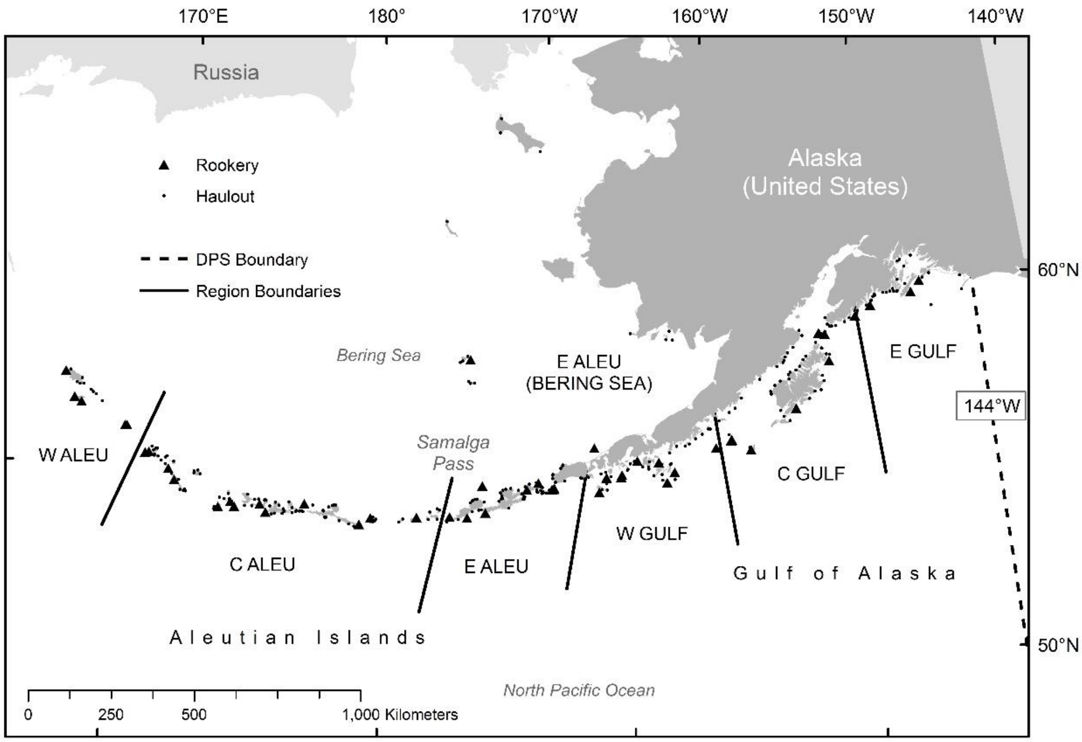

Figure 4. Regions of Alaska used for Western Steller sea lion population trend estimation. E GULF, C GULF, and W GULF are eastern, central, and western Gulf of Alaska regions, respectively. E ALEU, C ALEU, and W ALEU are eastern, central, and western Aleutian Islands regions, respectively (AFSC-MML-Alaska Ecosystems Program 2016).

In 2019, Western DPS survey effort was focused in the Gulf of Alaska (Sweeney et al. 2019). Between 2015 and 2017, pup counts declined in the eastern (-33%) and central (-18%) Gulf of Alaska, counter to the continuous increases observed in both regions since 2002 (Sweeney et al. 2017). These declines may have been due to changes in availability of prey associated with warm ocean temperatures that occurred in the northern Gulf of Alaska from 2014 to 2016 (Bond et al. 2015, Peterson et al. 2016, von Biela et al. 2019, Yang et al. 2019). There was also a movement of approximately 1,000 non-pups from the eastern to the central Gulf of Alaska regions, although the combined non-pup count in these two regions remained relatively stable between 2015 and 2017 (western Gulf of Alaska did not appear to change; Sweeney et al. 2017). In 2019, pup counts rebounded to 2015 levels; however, there was a decline in non-pup counts in the eastern, central, and western Gulf of Alaska regions (Sweeney et al. 2019).

No new data were collected for the Aleutian Islands in the 2019 survey, but the 2020 survey effort will be focused in this area. In 2018, survey effort was focused in the Aleutian Islands with some opportunistic surveys in the Gulf of Alaska (Sweeney et al. 2018). The area west of Samalga Pass was significantly declining, especially in the western Aleutian Islands region. The eastern Aleutian Islands region pups and non-pups have showed signs of recovery and have been increasing since the early 2000s.

Since part of the Western stock began to recover in the early 2000s, net movement between the Eastern and Western stocks appears to be small during the breeding season (Jemison et al. 2018). For example, there was an estimated net 75 sea lions that moved from east to west in 2016 (Jemison et al. 2013, Fritz et al. 2016). Very few females moved from Southeast Alaska to the Western stock, while approximately 500 were estimated to move from west to east (net increase in the east). Males moved in both directions, but with a net increase in the west. As a result, trends in counts estimated from breeding season surveys should be relatively insensitive, at a stock level, to inter-stock movements.

Burkanov and Loughlin (2005) estimated the Russian Steller sea lion population (pups and non-pups) declined approximately 52% from the 1970s to the 1990s. Johnson (2018) estimated the non-pup count in Russia declined 1.3% y^{-1} between 2002 and 2017; however, just as in the U.S. portion of the Western stock, there were significant regional differences in population trend in Russia (Table 2; Fig. 6; Burkanov 2018a, Johnson 2018). The significant decline in non-pup counts appears to be primarily driven by the decline in the Kurils which, traditionally, represents the largest area in terms of non-pup counts (Burkanov 2018a, Johnson 2018). Moreover, it seems the statistically significant decline in the Kurils is the result of the 2015 survey, where there appeared to be a large reduction in comparison to previous years (Fig. 6; Johnson 2018). Pup production appeared to decline in most areas where breeding occurs in Russia (Kuril Islands, eastern Kamchatka, the Commander Islands, and parts of the Sea of Okhotsk-Iony rookery); only Tuleny Island (Sakhalin region) and part of the Sea of Okhotsk (Yamsky Islands rookery) had increasing pup counts between 2006 and 2017 (Burkanov 2018a, 2018b).

Table 2. Trends (annual rates of change expressed as % y^{-1} with 95% credible interval) in non-pup counts for the Asian stock (Russia) of Steller sea lions and by region, from 2002 to 2017 (Johnson 2018). See Figure 2 for regions.

Region	**Trend**	**-95%**	**+95%**
Asian stock (Russia)	-1.3	-2.6	-0.1
Commander Islands	-0.6	-2.6	1.2
Kamchatka	-0.8	-3.0	1.5
Kuril	-4.1	-5.4	-2.8
Northern Sea of Okhotsk	0.9	-2.0	4.0
Sakhalin	0.9	-2.3	5.4
Western Bering Sea	-1.1	-16.1	10.2

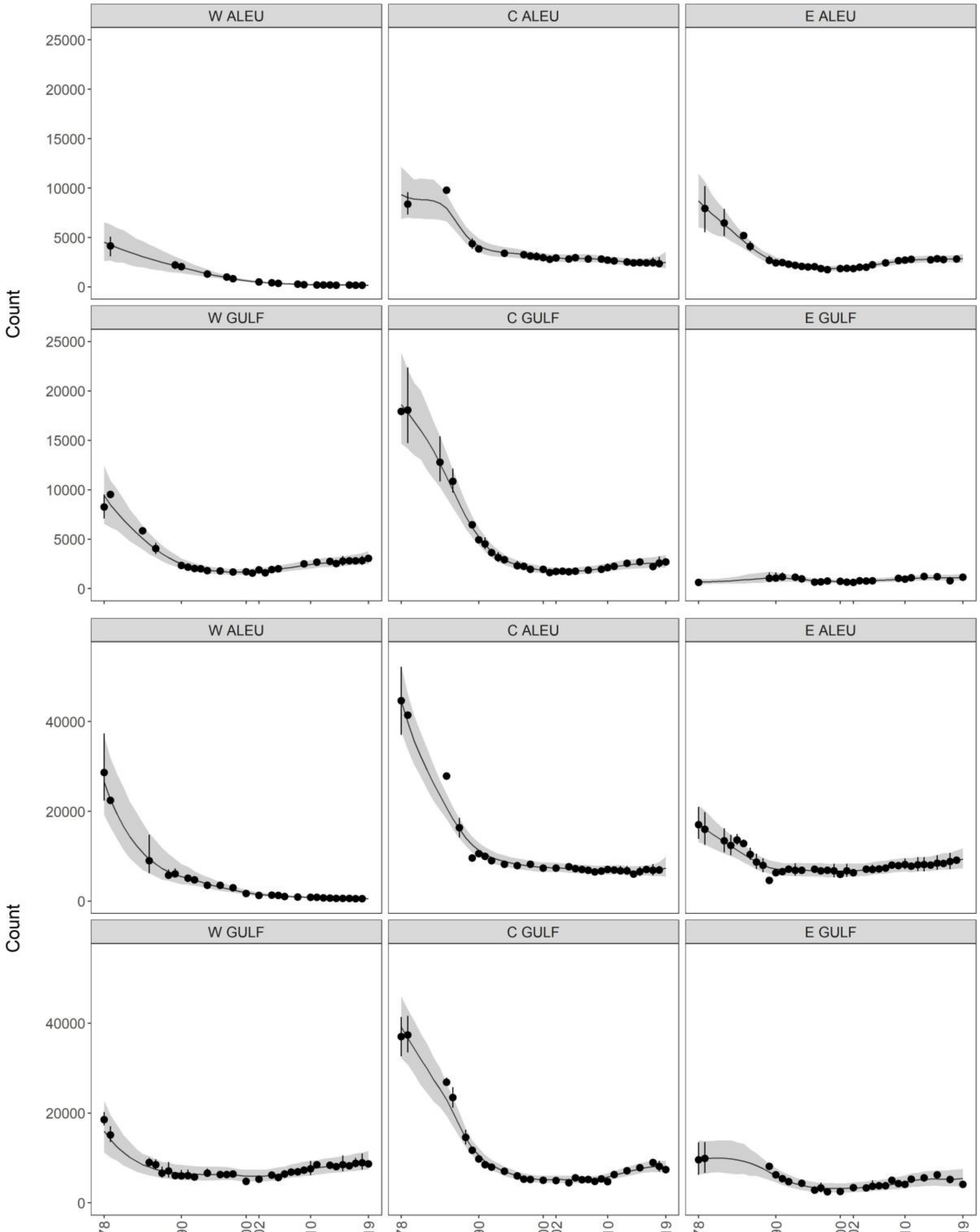

Figure 5. Realized and predicted counts of Steller sea lion pups (top) and non-pups (bottom) in the six regions that compose the Western stock in Alaska, 1978 to 2019. Realized counts are represented by points and vertical lines (95% credible intervals). Predicted counts are represented by the black line surrounded by the gray 95% credible interval (Sweeney et al. 2018, 2019).

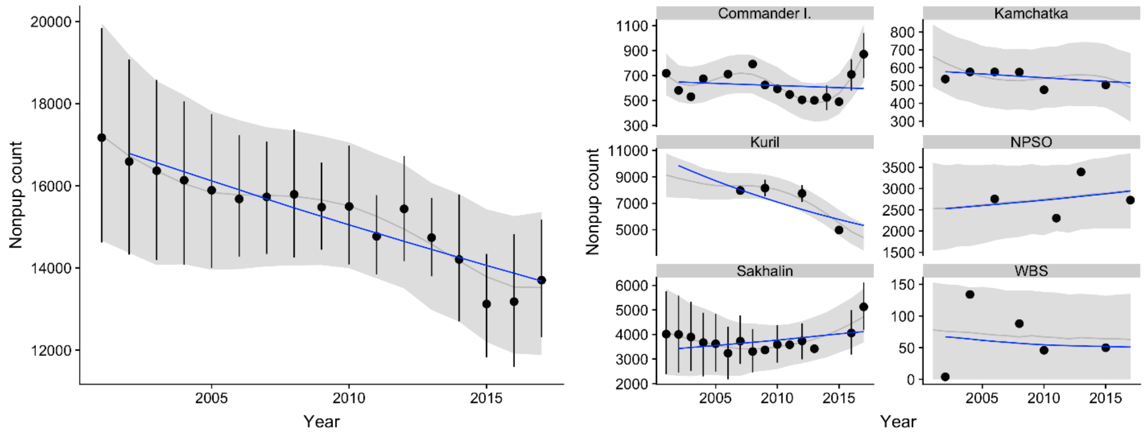

Figure 6. Realized and predicted counts of Russian Steller sea lion non-pups in Russia (left) and by region (right; Fig. 2), 2002 to 2017. Realized counts are represented by points and vertical lines (95% credible intervals). Predicted counts are represented by the black line surrounded by the gray 95% credible interval. The blue line represents the trend based on constant average growth for the entire Asian stock as a whole.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

There are no estimates of the maximum net productivity rate (R_{MAX}) for Steller sea lions. Until additional data become available, the default pinniped maximum theoretical net productivity rate of 12% will be used for this stock (NMFS 2016).

POTENTIAL BIOLOGICAL REMOVAL

Potential biological removal (PBR) is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: PBR = $N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 0.1, the default value for stocks listed as endangered under the ESA (NMFS 2016). Thus, for the U.S. portion of the Western stock of Steller sea lions, PBR is 318 sea lions (52,932 × 0.06 × 0.1).

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2014 and 2018 is listed, by marine mammal stock, in Young et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports. The minimum estimated mean annual level of human-caused mortality and serious injury for Western U.S. Steller sea lions between 2014 and 2018 is 254 sea lions: 37 in U.S. commercial fisheries, 0.8 in unknown (commercial, recreational, or subsistence) fisheries, 3.6 in marine debris, 3.6 due to other causes (illegal shooting, mortality incidental to Marine Mammal Protection Act (MMPA)-authorized research), and 209 in the Alaska Native subsistence harvest. No observers have been assigned to several fisheries that are known to interact with this stock and estimates of entanglement in fishing gear and marine debris based solely on stranding reports in areas west of 144°W longitude may underestimate the entanglement of Western stock animals that travel to parts of Southeast Alaska. Due to a lack of available resources, NMFS is not operating the Alaska Marine Mammal Observer Program (AMMOP) focused on marine mammal interactions that occur in fisheries managed by the State of Alaska. The most recent data on Steller sea lion interactions with state-managed fisheries in Alaska are from the Southeast Alaska salmon drift gillnet fishery in 2012 and 2013 (Manly 2015), a fishery in which the majority of the Steller sea lions taken are likely to be from the Eastern stock, although sea lions carrying Western genetic material could be as high as 38% (Hastings et al. 2019). Counts of annual illegal gunshot mortality in the Copper River Delta should be considered minimums as they are based solely on aerial carcass surveys from 2015 to 2018, no data are available for 2014, a cause of death for all carcasses found was not determined, and it is not likely that all carcasses are detected. Disturbance of Steller sea lion haulouts and rookeries can potentially cause disruption of reproduction, stampeding, or increased exposure to predation by marine predators (NMFS 2008; see also NMFS 1990, 1997). Effects of disturbance are highly variable and difficult to predict. Data are not available to estimate potential impacts from non-monitored activities, including disturbance near rookeries without 3-nmi no-entry buffer zones. Potential threats most likely to result in direct human-caused mortality or serious injury of this stock include subsistence harvest, incidental take, illegal shooting, disturbance at rookeries that could cause stampedes, and entanglement in fishing gear and marine debris.

Fisheries Information

Information for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is available in Appendix 3 of the Alaska Stock Assessment Reports (observer coverage) and in the NMFS List of Fisheries (LOF) and the fact sheets linked to fishery names in the LOF (observer coverage and reported incidental takes of marine mammals: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-list-fisheries, accessed December 2020).

Based on historical reports and their geographic range, Steller sea lion mortality and serious injury could occur in several fishing gear types, including trawl, gillnet, longline, and troll fisheries. However, observer data are limited. Of these fisheries, only trawl fisheries are regularly observed and gillnet fisheries have had limited observations in select areas over short time frames and with modest observer coverage. Consequently, there are little to no data on Steller sea lion mortality and serious injury in non-trawl fisheries. Therefore, the potential for fisheries-caused mortality and serious injury may be greater than is reflected in existing observer data.

Between 2014 and 2018, mortality and serious injury of Western Steller sea lions was observed in 10 of the federally-managed commercial fisheries in Alaska that are monitored for incidental mortality and serious injury by fisheries observers: Bering Sea/Aleutian Islands Atka mackerel trawl, Bering Sea/Aleutian Islands flatfish trawl, Bering Sea/Aleutian Islands Pacific cod trawl, Bering Sea/Aleutian Islands pollock trawl, Bering Sea/Aleutian Islands Pacific cod longline, Gulf of Alaska Pacific cod trawl, Gulf of Alaska Pacific cod longline, Gulf of Alaska flatfish trawl, Gulf of Alaska rockfish trawl, and Gulf of Alaska pollock trawl fisheries, resulting in a mean annual mortality and serious injury rate of 22 sea lions (Table 3; Breiwick 2013; MML, unpubl. data).

AMMOP observers monitored the Alaska State-managed Prince William Sound salmon drift gillnet fishery in 1990 and 1991, recording two incidental mortalities in 1991, extrapolated to 29 (95% CI: 1-108) for the entire fishery (Wynne et al. 1992; Table 3). No incidental mortality or serious injury was observed during 1990 for this fishery (Wynne et al. 1991), resulting in a mean annual mortality rate of 15 sea lions for 1990 and 1991. It is not known whether this incidental mortality and serious injury rate is representative of the current rate in this fishery.

Between 2014 and 2018, Steller sea lion mortality resulting from entanglements in commercial longline gear (1 in 2015) and commercial salmon seine net (1 in 2018) was reported to the NMFS Alaska Region marine mammal stranding network (Young et al. 2020), resulting in a mean annual mortality and serious injury rate of 0.4 sea lions in commercial gear (Table 4). This mortality and serious injury estimate results from an actual count of verified human-caused deaths and serious injuries and is a minimum because not all entangled animals strand nor are all stranded animals found, reported, or have the cause of death determined.

The minimum estimated mean annual mortality and serious injury rate in U.S. commercial fisheries between 2014 and 2018 is 37 Steller sea lions from this stock (37 from observer data + 0.4 from stranding data) (Tables 3 and 4). No observers have been assigned to several fisheries that are known to interact with this stock, thus, the estimated mortality and serious injury is likely an underestimate of the actual level.

Table 3. Summary of incidental mortality and serious injury of Western U.S. Steller sea lions due to U.S. commercial fisheries between 2014 and 2018 (or the most recent data available) and calculation of the mean annual mortality and serious injury rate (Wynne et al. 1991, 1992; Breiwick 2013; MML, unpubl. data). N/A indicates that data are not available. Methods for calculating percent observer coverage are described in Appendix 3 of the Alaska Stock Assessment Reports.

Fishery name	Years	Data type	Percent observer coverage	Observed mortality	Estimated mortality (CV)	Mean estimated annual mortality
Bering Sea/Aleutian Is. Atka mackerel trawl	2014	obs data	100	0	0	1.2 (CV = 0.07)
	2015		100	0	0	
	2016		98	0	0	
	2017		100	1	1 (0.06)	
	2018		100	5	5.1 (0.08)	
Bering Sea/Aleutian Is. flatfish trawl	2014	obs data	100	5	5.0 (0.02)	8.2 (CV = 0.01)
	2015		100	6	6.0 (0.02)	
	2016		99	9	9.0 (0.02)	
	2017		100	13	13 (0.01)	
	2018		100	8	8.0 (0.02)	
Bering Sea/Aleutian Is. Pacific cod trawl	2014	obs data	80	0	0	0.4 (CV = 0)
	2015		72	0	0	
	2016		68	0	0	
	2017		68	1	1 (0)	
	2018		73	1	1 (0)	
Bering Sea/Aleutian Is. pollock trawl	2014	obs data	98	2	2.0 (0.1)	5.7 (CV = 0.02)
	2015		99	1	1 (0.07)	
	2016		99	13	13 (0.03)	
	2017		99	6	6.1 (0.05)	
	2018		99	6	6.1 (0.04)	
Bering Sea/Aleutian Is. pollock trawl	2017	obs data	99	1[a]	N/A	0.2 (CV = N/A)
Bering Sea/Aleutian Is. Pacific cod longline	2014	obs data	64	1	1.7 (0.63)	1.6 (CV = 0.28)
	2015		62	3	4.9 (0.36)	
	2016		57	0	0	
	2017		58	1	1.6 (0.6)	
	2018		55	0	0	
Gulf of Alaska Pacific cod longline	2014	obs data	31	0	0	0.3 (CV = 0.5)
	2015		36	1	1.3 (0.5)	
	2016		30	0	0	
	2017		40	0	0	
	2018		29	0	0	
Gulf of Alaska Pacific cod trawl	2014	obs data	12	0	0	2.0 (CV = 0.9)
	2015		13	0	0	
	2016		13	1	10 (0.9)	
	2017		11	0	0	
	2018		25	0	0	
Gulf of Alaska flatfish trawl	2014	obs data	47	0	0	0 (+0.2)[d] (CV = N/A)
	2015		54	0 (+1)[b]	0 (+1)[c]	
	2016		39	0	0	
	2017		56	0	0	
	2018		34	0	0	

Fishery name	Years	Data type	Percent observer coverage	Observed mortality	Estimated mortality (CV)	Mean estimated annual mortality
Gulf of Alaska rockfish trawl	2014	obs data	96	0	0	0 (+0.2)[d] (CV = N/A)
	2015		93	0 (+1)[b]	0 (+1)[c]	
	2016		98	0	0	
	2017		98	0	0	
	2018		95	0	0	
Gulf of Alaska pollock trawl	2014	obs data	14	0	0	1.0 (+1)[g] (CV = 0.89)
	2015		23	0 (+5)[e]	0 (+5)[f]	
	2016		27	1	4.8 (0.89)	
	2017		19	0	0	
	2018		21	0	0	
Prince William Sound salmon drift gillnet	1990	obs data	4	0	0	15 (CV = 1.0)
	1991		5	2	29	
Minimum total estimated annual mortality						37 (CV = 0.43)

[a]This animal was discovered during a vessel offload. Because it could not be associated with a haul number, it was not included in the bycatch estimate for the fishery.
[b]Total mortality and serious injury observed in 2015: 0 sea lions in sampled hauls + 1 sea lion in an unsampled haul.
[c]Total estimate of mortality and serious injury in 2015: 0 sea lions (extrapolated estimate from 0 sea lions observed in sampled hauls) + 1 sea lion (1 sea lion observed in an unsampled haul).
[d]Mean annual mortality and serious injury for fishery: 0 sea lions (mean of extrapolated estimates from sampled hauls) + 0.2 sea lions (mean of number observed in unsampled hauls).
[e]Total mortality and serious injury observed in 2015: 0 sea lions in sampled hauls + 5 sea lions in unsampled hauls.
[f]Total estimate of mortality and serious injury in 2015: 0 sea lions (extrapolated estimate from 0 sea lions observed in sampled hauls) + 5 sea lions (5 sea lions observed in unsampled hauls).
[g]Mean annual mortality and serious injury for fishery: 1.0 sea lion (mean of extrapolated estimates from sampled hauls) + 1 sea lion (mean of number observed in unsampled hauls).

Reports to the NMFS Alaska Region marine mammal stranding network of Steller sea lions entangled in fishing gear or with injuries caused by interactions with gear are another source of mortality and serious injury data (Table 4; Young et al. 2020). From 2014 to 2018, there were three reports of Steller sea lion interactions with salmon hook and line gear, in which an animal in poor body condition had a flasher lure hanging from its mouth and was believed to have ingested the hook, and one report of an animal that was entangled in unidentified hook and line gear, resulting in a mean annual mortality and serious injury rate of 0.8 sea lions in these unknown (commercial, recreational, or subsistence) fisheries (Table 4). This mortality and serious injury estimate results from an actual count of verified human-caused deaths and serious injuries and is a minimum because not all entangled animals strand nor are all stranded animals found, reported, or have the cause of death determined. Additionally, since Steller sea lions from parts of the Western stock are known to regularly occur in parts of Southeast Alaska (Jemison et al. 2013, 2018; NMFS 2013), and higher rates of entanglement of Steller sea lions have been observed in this area (e.g., Raum-Suryan et al. 2009), estimates based solely on stranding reports in areas west of 144°W longitude may underestimate the total entanglement of Western stock sea lions in fishery-related gear and marine debris.

Table 4. Summary of Western U.S. Steller sea lion mortality and serious injury, by year and type, reported to the NMFS Alaska Region marine mammal stranding network and Alaska Department of Fish and Game between 2014 and 2018 (Young et al. 2020). N/A indicates that data are not available.

Cause of injury	2014	2015	2016	2017	2018	Mean annual mortality
Entangled in commercial Kodiak salmon seine net	0	0	0	0	1	0.2
Entangled in commercial longline gear	0	1	0	0	0	0.2
Hooked by salmon hook and line gear*	1	0	0	1	1	0.6
Entangled in unknown hook and line gear*	1	0	0	0	0	0.2
Entangled in marine debris	3	6	1	3	5	3.6
Illegally shot	N/A	8	1	0	0	3[a]
Incidental to MMPA-authorized research	0	1	2	0	0	0.6
Total in commercial fisheries						0.4
*Total in unknown (commercial, recreational, or subsistence) fisheries						0.8
Total in marine debris						3.6
Total due to other causes (illegally shot, incidental to MMPA-authorized research)						3.6

[a]Dedicated effort to survey the Copper River Delta for stranded marine mammals began in 2015 in response to a high number of reported strandings, some of which were later determined to be human-caused (illegally shot). Dedicated surveys were also conducted in 2016, 2017, and 2018. Because similar data are not available for 2014 and survey effort was limited in 2018, the data were averaged over 3 years of survey effort (2015-2017) for a more informed estimate of mean annual mortality.

The minimum mean annual mortality and serious injury rate for all fisheries between 2014 and 2018, based on observer data and stranding data for U.S. commercial fisheries (37 sea lions) and on stranding data for unknown (commercial, recreational, or subsistence) fisheries (0.8 sea lions), is 38 Western Steller sea lions.

Alaska Native Subsistence/Harvest Information

NMFS signed agreements with the Tribal Government of St. Paul Island (2000) and the Traditional Council of St. George Island (2001) to co-manage Steller sea lions and northern fur seals. NMFS also signed an agreement with the Aleut Marine Mammal Commission (2006) for the conservation and management of all marine mammal subsistence species, with particular focus on Steller sea lions and harbor seals. These co-management agreements promote full and equal participation by Alaska Natives in decisions affecting the subsistence management of Steller sea lions (to the maximum extent allowed by law) as a tool for conserving Steller sea lion populations in Alaska (https://www.fisheries.noaa.gov/alaska/marine-mammal-protection/co-management-marine-mammals-alaska, accessed December 2020).

Information on the subsistence harvest of Steller sea lions comes via three sources: the Alaska Department of Fish and Game (ADF&G), the Ecosystem Conservation Office of the Aleut Community of St. Paul Island, and the Kayumixtax Eco-Office of the Aleut Community of St. George Island. The ADF&G conducted systematic interviews with hunters and users of marine mammals in approximately 2,100 households in about 60 coastal communities within the geographic range of the Steller sea lion in Alaska (Wolfe et al. 2005, 2006, 2008, 2009a, 2009b). The interviews were conducted once per year in the winter (January to March) and covered hunter activities for the previous calendar year. As of 2009, annual statewide data on community subsistence harvests are no longer being consistently collected. Data are being collected periodically in subareas. Data were collected on the Alaska Native harvest of Western U.S. Steller sea lions for 7 communities on Kodiak Island in 2011 and for 15 communities in Southcentral Alaska in 2014. The Alaska Native Harbor Seal Commission (ANHSC) and ADF&G estimated a total of 20 adult sea lions were harvested on Kodiak Island in 2011, with a 95% confidence range between 15 and 28 animals (Wolfe et al. 2012), and 7.9 sea lions (CI = 6-15.3) were harvested in Southcentral Alaska in 2014, with adults comprising 84% of the harvest (ANHSC 2015). These estimates do not represent a comprehensive statewide estimate; therefore, the best available statewide subsistence harvest estimates for a 5-year period are those from 2004 to 2008. Thus, the most recent 5 years of data available from the ADF&G (2004-2008) will be used for calculating an annual mortality and serious injury estimate for all areas except St. Paul, St. George, and Atka Islands (Wolfe et al. 2005, 2006, 2008, 2009a, 2009b; NMFS, unpubl. data) (Table 5). Harvest data are collected in near real-time on St. Paul Island (e.g., Melovidov 2013) and St. George Island (e.g., Kashevarof 2015) and recorded within 36 hours of the harvest. The most recent 5 years of data from St. Paul (Melovidov 2013, 2014, 2015, 2016; NMFS, unpubl. data) and St. George (Kashevarof 2015; NMFS, unpubl. data) are for 2014 to 2018 (Table 5).

The mean annual subsistence harvest from this stock for all areas except St. Paul, St. George, and Atka Island between 2004 and 2008 (172) combined with the mean annual harvest for St. Paul (30), St. George (1.4), and Atka (6) Islands between 2014 and 2018 is 209 Western Steller sea lions (Table 5).

Table 5. Summary of the subsistence harvest data for Western U.S. Steller sea lions. As of 2009, data on community subsistence harvests are no longer being consistently collected. Therefore, the most recent 5 years of data (2004 to 2008) will be used for calculating an annual mortality and serious injury estimate for all areas except St. Paul, St. George, and Atka Islands. Data from St. Paul, St. George, and Atka Islands are still being collected and the most recent 5 years of data available (2014 to 2018) will be used. N/A indicates that data are not available.

	All areas except St. Paul Island			St. Paul Island	St. George Island	Atka Island
Year	**Number harvested**	**Number struck and lost**	**Total**	**Number harvested + Number struck and lost**	**Number harvested + Number struck and lost**	**Number harvested + Number struck and lost**
2004	136.8	49.1	185.9[a]			
2005	153.2	27.6	180.8[b]			
2006	114.3	33.1	147.4[c]			
2007	165.7	45.2	210.9[d]			
2008	114.7	21.6	136.3[e]			
2014	N/A	N/A	N/A	35[h]	1[g]	N/A
2015	N/A	N/A	N/A	24[i]	3[g]	N/A
2016	N/A	N/A	N/A	31[j]	2[j]	N/A
2017	N/A	N/A	N/A	30[j]	0[j]	N/A
2018	N/A	N/A	N/A	28[j]	1[j]	6
Mean annual harvest	137	35	172	30	1.4	6

[a]Wolfe et al. (2005); [b]Wolfe et al. (2006); [c]Wolfe et al. (2008); [d]Wolfe et al. (2009a); [e]Wolfe et al. (2009b); [h]Melovidov (2015); [i]Melovidov (2016); [j]NMFS, unpubl. data.

Other Mortality

Reports to the NMFS Alaska Region marine mammal stranding network of Steller sea lions entangled in marine debris or with injuries caused by other types of human interaction are another source of mortality and serious injury data. These mortality and serious injury estimates result from an actual count of verified human-caused deaths and serious injuries and are minimums because not all entangled animals strand nor are all stranded animals found, reported, or have the cause of death determined. Between 2014 and 2018, reports to the stranding network resulted in mean annual mortality and serious injury rates of three Steller sea lions illegally shot in the Copper River Delta (3-year average) and 3.6 observed entangled in marine debris (Table 4; Young et al. 2020). Additional reports of Steller sea lion mortality due to gunshot wounds are not included in the estimate of the mean annual mortality and serious injury rate for 2014 to 2018 because it could not be confirmed that the animals were illegally shot rather than struck and lost in the Alaska Native subsistence harvest.

Mortality and serious injury may occasionally occur incidental to marine mammal research activities authorized under MMPA permits issued to a variety of government, academic, and other research organizations. Between 2014 and 2018, there were three reports (one in 2015 and two in 2016) of mortality incidental to research on the Western U.S. stock of Steller sea lions (Table 4; Young et al. 2020), resulting in a mean annual mortality and serious injury rate of 0.6 sea lions from this stock.

STATUS OF STOCK

The minimum estimated mean annual U.S. commercial fishery-related mortality and serious injury rate (37 sea lions) is more than 10% of the PBR (10% of PBR = 32) and, therefore, cannot be considered insignificant and approaching a zero mortality and serious injury rate. Based on available data, the minimum estimated mean annual level of human-caused mortality and serious injury (254 sea lions) is below the PBR level (318) for this stock. The Western U.S. stock of Steller sea lions is currently listed as endangered under the ESA and, therefore, designated as depleted under the MMPA. As a result, the stock is classified as a strategic stock. The population previously declined for unknown reasons that are not explained by the documented level of direct human-caused mortality and serious injury.

There are key uncertainties in the assessment of the Western U.S. stock of Steller sea lions. Some genetic studies support the separation of Steller sea lions in western Alaska from those in Russia; population numbers in this assessment are only from the U.S. to be consistent with the geographic range of information on mortality and serious injury. We provide data for the Russian population for context for the entire Western DPS. There is some overlap in range between animals in the Western and Eastern stocks in northern Southeast Alaska. The population abundance is based on counts of visible animals; the calculated N_{MIN} and PBR levels are conservative because there are no data available to correct for animals not visible during the visual surveys. There are multiple nearshore commercial fisheries that are not observed; thus, there is likely to be unreported fishery-related mortality and serious injury of Steller sea lions. Estimates of human-caused mortality and serious injury from stranding data are underestimates because not all animals strand nor are all stranded animals found, reported, or have the cause of death determined. Several factors may have been important drivers of the decline of the stock. However, there is uncertainty about threats currently impeding their recovery, particularly in the Aleutian Islands.

HABITAT CONCERNS

Many factors have been suggested as causes of the steep decline in abundance of Western Steller sea lions observed in the 1980s, including competitive effects of fishing, environmental change, disease, contaminants, killer whale predation, incidental take, and illegal and legal shooting (Atkinson et al. 2008, NMFS 2008). A number of management actions have been implemented since 1990 to promote the recovery of the Western U.S. stock of Steller sea lions, including 3-nmi no-entry zones around rookeries, prohibition of shooting at or near sea lions, and regulation of fisheries for sea lion prey species (e.g., walleye pollock, Pacific cod, and Atka mackerel; see reviews by Fritz et al. 1995, McBeath 2004, Atkinson et al. 2008, NMFS 2008). Additionally, potentially deleterious events, such as harmful algal blooms (Lefebvre et al. 2016) and disease transmission across the Arctic (VanWormer et al. 2019) that have been associated with warming waters, could lead to potentially negative population-level impacts on Steller sea lions. Metal and contaminant exposure remains a focus of ongoing investigation. Total mercury concentrations measured in hair samples collected from pups in the western-central Aleutian Islands are the highest measured for this species and at levels that in other species cause neurological and reproductive effects (Rea et al. 2013), and organochlorine burdens were detected in tissue samples from across the range but were highest in pups sampled from the Aleutian Islands (Beckmen et al. 2016, Keogh et al. 2020).

The area of greatest (continued) decline in the U.S. remains in the western Aleutian Islands (west of Samalga Pass). Pacific cod and Atka mackerel are two of the primary prey species of Steller sea lions in the central and western Aleutian Islands (Sinclair et al. 2013, Tollit et al. 2017). In the increasing eastern Aleutian Islands region, Rand et al. (2019) reported dense and consistent aggregations of Atka mackerel. However, in the western Aleutian Islands region, this important prey species was more spread out over a larger area during the non-breeding (i.e., "winter") season (Fritz et al. 2019, Rand et al. 2019). Prey availability over winter is thought to be a key factor in energy budgets of sea lions, especially for pregnant females and especially those supporting a pup and/or juvenile (NMFS 2010, Boyd 2000, Malavaer 2002, Winship et al. 2002, Williams 2005). This could result in increases in energy expenditures by Steller sea lions associated with finding and capturing prey, as evident by increased frequency and duration of foraging trips observed in juvenile Steller sea lions in this region (Lander et al. 2010). Prey species (e.g., Atka mackerel, Pacific cod, and walleye pollock) are likely to have lower overall abundance, less predictable spatial distributions, and altered demographics in fished versus unfished habitats (Hsieh et al. 2006, Barbeaux et al. 2013, Fritz et al. 2019). In 2011, the Pacific cod and Atka mackerel fisheries were closed and then re-opened in 2014. In the western Aleutian Islands region, modeled realized counts exhibited stability from 2014 to 2016 (and potentially an increase in pup counts), followed by continued declines since 2016 (Sweeney et al. 2016, 2017, 2018). Fritz et al. (2019) suggested that if nutrition is a driver of the decline, then it appears that other factors (than diet diversity, species mix, and energy density) may be acting. The literature does not prove (or disprove) a correlation between fisheries, sea lion population trends, and prey availability in the Aleutian Islands, and this hypothesis is an important area of investigation for Steller sea lions, especially in the Aleutian Islands.

The Pacific marine heatwave that occurred from 2014 to 2016, and subsequent warm waters in the north Pacific, especially the Gulf of Alaska, has been linked to large declines in productivity and impacts on groundfish populations (von Biela et al. 2019, Yang et al. 2019). In fact, the concomitant decline in pup productivity in the eastern and central Gulf of Alaska regions observed from 2015 and 2017 may be related to the reduction of available prey in the area (Sweeney et al. 2017). In 2019, pup production in these regions rebounded to 2015 levels; however, there was a decline in non-pups that spanned all the Gulf of Alaska regions (Sweeney et al. 2019). These declines are concerning given that prior to 2017, these regions were showing relatively consistent and steady increases in counts (Sweeney et al. 2019). As Alaska waters, especially the Gulf of Alaska, continue to warm, it seems evident from NOAA Fisheries sea lion surveys that this could continue to impact the Western stock in the U.S. It is also possible that changes in foraging ability could affect sea lion movements between and within the stocks (Jemison et al. 2018).

CITATIONS

AFSC/MML/Alaska Ecosystems Program. 2016. Steller sea lion haulout and rookery locations in the United States for 2016-05-14 (NCEI Accession 0129877). NOAA National Centers for Environmental Information Dataset. DOI: dx.doi.org/10.7289/V58C9T7V .

Alaska Native Harbor Seal Commission (ANHSC). 2015. 2014 estimate of the subsistence harvest of harbor seals and sea lions by Alaska Natives in southcentral Alaska: summary of study findings. Alaska Native Harbor Seal Commission and Alaska Department of Fish & Game, Division of Subsistence. 15 p.

Atkinson, S., D. P. DeMaster, and D. G. Calkins. 2008. Anthropogenic causes of the western Steller sea lion *Eumetopias jubatus* population decline and their threat to recovery. Mammal Rev. 38(1):1-18.

Baker, A. R., T. R. Loughlin, V. Burkanov, C. W. Matson, T. G. Trujillo, D. G. Calkins, J. K. Wickliffe, and J. W. Bickham. 2005. Variation of mitochondrial control region sequences of Steller sea lions: the three-stock hypothesis. J. Mammal. 86:1075-1084.

Barbeaux, S. J., J. K. Horne, and M. W. Dorn. 2013. Characterizing walleye pollock (*Theragra chalcogramma*) winter distribution from opportunistic data. ICES J. Mar. Sci. 70(6):1162-1173. DOI: dx.doi.org/10.1093/icesjms/fst052 .

Beckmen, K. B., M. J. Keogh, K. A. Burek-Huntington, G. M. Ylitalo, B. S. Fadely, and K. W Pitcher. 2016. Organochlorine contaminant concentrations in multiple tissues of free-ranging Steller sea lions (*Eumetopias jubatus*) in Alaska. Science of the Total Environment 542:441-452. DOI: dx.doi.org/10.1016/j.scitotenv.2015.10.119 .

Berta, A., and M. Churchill. 2012. Pinniped taxonomy: review of currently recognized species and subspecies, and evidence used for their description. Mammal Rev. 42(2):207-234.

Bickham, J. W., J. C. Patton, and T. R. Loughlin. 1996. High variability for control-region sequences in a marine mammal: implications for conservation and biogeography of Steller sea lions (*Eumetopias jubatus*). J. Mammal. 77:95-108.

Bond, N. A., M. F. Cronin, H. Freeland, and N. Mantua. 2015. Causes and impacts of the 2014 warm anomaly in the NE Pacific. Geophys. Res. Lett. 42(9):3414-3420. DOI: dx.doi.org/10.1002/2015GL063306 .

Boyd, I. L. 2000. State-dependent fertility in pinnipeds: contrasting capital and income breeders. Functional Ecology 14(5):623-630.

Braham, H. W., R. D. Everitt, and D. J. Rugh. 1980. Northern sea lion decline in the eastern Aleutian Islands. J. Wildl. Manage. 44:25-33.

Breiwick, J. M. 2013. North Pacific marine mammal bycatch estimation methodology and results, 2007-2011. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-260, 40 p.

Burkanov, V. 2017. Results of breeding season Steller sea lion pup surveys in Russia, 2011-2016. Memorandum to T. Gelatt and J. Bengtson, April 6, 2017. Available from Marine Mammal Laboratory, AFSC, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Burkanov, V. 2018a. Brief results on the most recent and complete Steller sea lion counts in Russia. Memorandum to T. Gelatt and J. Bengtson. Available from Marine Mammal Laboratory, AFSC, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115. 35 p.

Burkanov, V. 2018b. Current Steller sea lion pup production along Asian coast, 2016-2017. Memorandum to T. Gelatt and J. Bengtson. Available from Marine Mammal Laboratory, AFSC, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115. 3 p.

Burkanov, V., and T. R. Loughlin. 2005. Distribution and abundance of Steller sea lions on the Asian coast, 1720's–2005. Mar. Fish. Rev. 67(2):1-62.

Byrd, G. V. 1989. Observations of northern sea lions at Ugamak, Buldir, and Agattu Islands, Alaska in 1989. Unpubl. report, U.S. Fish and Wildlife Service, Alaska Maritime National Wildlife Refuge, P.O. Box 5251, NSA Adak, FPO Seattle, WA 98791.

Calkins, D. G., and K. W. Pitcher. 1982. Population assessment, ecology and trophic relationships of Steller sea lions in the Gulf of Alaska. Environmental Assessment of the Alaskan Continental Shelf. Final Reports 19:455-546.

DeMaster, D. P. 2014. Results of Steller sea lion surveys in Alaska, June-July 2013. Memorandum to J. Balsiger, J. Kurland, B. Gerke, and L. Rotterman, NMFS Alaska Regional Office, Juneau, AK, January 30, 2014. Available from Marine Mammal Laboratory, AFSC, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Dizon, A. E., C. Lockyer, W. F. Perrin, D. P. DeMaster, and J. Sisson. 1992. Rethinking the stock concept: a phylogeographic approach. Conserv. Biol. 6:24-36.

Fritz, L. W., R. C. Ferrero, and R. J. Berg. 1995. The threatened status of Steller sea lions, *Eumetopias jubatus*, under the Endangered Species Act: effects on Alaska groundfish fisheries management. Mar. Fish. Rev. 57(2):14-27.

Fritz, L., K. Sweeney, D. Johnson, M. Lynn, and J. Gilpatrick. 2013. Aerial and ship-based surveys of Steller sea lions (*Eumetopias jubatus*) conducted in Alaska in June-July 2008 through 2012, and an update on the status and trend of the western stock in Alaska. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-251, 91 p.

Fritz, L., K. Sweeney, R. Towell, and T. Gelatt. 2016. Aerial and ship-based surveys of Steller sea lions (*Eumetopias jubatus*) conducted in Alaska in June-July 2013 through 2015, and an update on the status and trend of the western distinct population segment in Alaska. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-321, 72 p.

Fritz, L., B. Brost, E. Laman, K. Luxa, K. Sweeney, J. Thomason, D. Tollit, W. Walker, and T. Zeppelin. 2019. A re-examination of the relationship between Steller sea lion (*Eumetopias jubatus*) diet and population trend using data from the Aleutian Islands. Can. J. Zool. 97:1137-1155. DOI: dx.doi.org/10.1139/cjz-2018-0329 .

Gelatt, T. S., A. W. Trites, K. Hastings, L. Jemison, K. Pitcher, and G. O'Corry-Crowe. 2007. Population trends, diet, genetics, and observations of Steller sea lions in Glacier Bay National Park, p. 145-149. *In* J. F. Piatt, and S. M. Gende (eds.), Proceedings of the Fourth Glacier Bay Science Symposium, October 26–28, 2004: U.S. Geological Survey Scientific Investigations Report 2007-5047.

Harlin-Cognato, A., J. W. Bickham, T. R. Loughlin, and R. L. Honeycutt. 2006. Glacial refugia and the phylogeography of Steller's sea lion (*Eumetopias jubatus*) in the North Pacific. J. Evol. Biol. 19:955-969. DOI: dx.doi.org/10.1111/j.1420-9101.2005.01052.x .

Hastings, K. K., L. A. Jemison, G. W. Pendleton, K. L. Raum-Suryan, and K. W. Pitcher. 2017. Natal and breeding philopatry of female Steller sea lions in southeastern Alaska. PLoS ONE 13(4):e0196412. DOI: dx.doi.org/10.1371/journal.pone.0176840 .

Hastings, K. K., M. J Rehberg, G. M. O'Corry-Crowe, G. W. Pendleton, L. A. Jemison, and T. S. Gelatt. 2019. Demographic consequences and characteristics of recent population mixing and colonization in Steller sea lions, *Eumetopias jubatus*. J. Mammal. DOI: dx.doi.org/10.1093/jmammal/gyz192 .

Higgins, L. V., D. P. Costa, A. C. Huntley, and B. J. Le Boeuf. 1988. Behavioral and physiological measurements of maternal investment in the Steller sea lion, *Eumetopias jubatus*. Mar. Mammal Sci. 4:44-58.

Hoffman, J. I., C. W. Matson, W. Amos, T. R. Loughlin, and J. W. Bickham. 2006. Deep genetic subdivision within a continuously distributed and highly vagile marine mammal, the Steller's sea lion (*Eumetopias jubatus*). Mol. Ecol. 15:2821-2832.

Hoffman, J. I., K. K. Dasmahapatra, W. Amos, C. D. Phillips, T. S. Gelatt, and J. W Bickham. 2009. Contrasting patterns of genetic diversity at three different genetic markers in a marine mammal metapopulation. Mol. Ecol. 18(14):2961-2978.

Holmes, E. E., L. W. Fritz, A. E. York, and K. Sweeney. 2007. Age-structured modeling provides evidence for a 28-year decline in the birth rate of western Steller sea lions. Ecol. Appl. 17(8):2214-2232.

Hsieh, C. H., C. S. Reiss, J. R. Hunter, J. R. Beddington, R. M. May, and G. Sugihara. 2006. Fishing elevates variability in the abundance of exploited species. Nature 443:859-862. DOI: dx.doi.org/10.1038/nature05232 .

Jemison, L. A., G. W. Pendleton, L. W. Fritz, K. K. Hastings, J. M. Maniscalco, A. W. Trites, and T. S. Gelatt. 2013. Inter-population movements of Steller sea lions in Alaska, with implications for population separation. PLoS ONE 8(8):e70167.

Jemison, L. A., G. W. Pendleton, K. K. Hastings, J. M. Maniscalco, and L. W. Fritz. 2018. Spatial distribution, movements, and geographic range of Steller sea lions (*Eumetopias jubatus*) in Alaska. PLoS ONE 13(12):e0208093. DOI: dx.doi.org/10.1371/journal.pone.0208093 .

Johnson, D. 2018. Trends of nonpup survey counts of Russian Steller sea lions. Memorandum for T. Gelatt and J. Bengtson, June 6, 2018. Available from NMFS Alaska Region, Office of Protected Resources, 709 West 9th Street, Juneau, AK 99802-1668.

Johnson, D. S., and L. W. Fritz. 2014. agTrend: a Bayesian approach for estimating trends of aggregated abundance. Methods Ecol. Evol. 5:1110-1115. DOI: dx.doi.org/10.1111/2041-210X.12231 .

Kashevarof, H. 2015. St. George co-management comprehensive report. St. George Island Traditional Council Kayumixtax Eco-Office, St. George Island, AK 99591.

Keogh, M. J., B. Taras, K. B. Beckmen, K. A. Burek-Huntington, G. M. Ylitalo, B. S. Fadely, L. D. Rea, and K. W. Pitcher. 2020. Organochlorine contaminant concentrations in blubber of young Steller sea lion (*Eumetopias jubatus*) are influenced by region, age, sex and lipid stores. Science of the Total Environment 698:134183. DOI: dx.doi.org/10.1016/j.scitotenv.2019.134183 .

Kuhn, C. E., K. Chumbley, D. Johnson, and L. Fritz. 2017. A re-examination of the timing of pupping for Steller sea lions *Eumetopias jubatus* breeding on two islands in Alaska. Endang. Species Res. 32:213-222. DOI: dx.doi.org/10.3354/esr00796 .

Lander, M. E., T. R. Loughlin, M. G. Logsdon, G. R. VanBlaricom, and B. S. Fadely. 2010. Foraging effort of juvenile Steller sea lions *Eumetopias jubatus* with respect to heterogeneity of sea surface temperature. Endang. Species Res. 10:145-158. DOI: dx.doi.org/ 10.3354/esr00260 .

Lefebvre, K. A., L. Quakenbush, E. Frame, K. Burek Huntington, G. Sheffield, R. Stimmelmayr, A. Bryan, P. Kendrick, H. Ziel, T. Goldstein, J. A. Snyder, T. Gelatt, F. Gulland, B. Dickerson, and V. Gil. 2016. Prevalence of algal toxins in Alaskan marine mammals foraging in a changing arctic and subarctic environment. Harmful Algae 55:13-24. DOI: dx.doi.org/10.1016/j.hal.2016.01.007 .

Loughlin, T. R. 1997. Using the phylogeographic method to identify Steller sea lion stocks, p. 329-341. *In* A. Dizon, S. J. Chivers, and W. Perrin (eds.), Molecular genetics of marine mammals, incorporating the proceedings of a workshop on the analysis of genetic data to address problems of stock identity as related to management of marine mammals. Soc. Mar. Mammal., Spec. Rep. No. 3.

Loughlin, T. R., and A. E. York. 2000. An accounting of the sources of Steller sea lion mortality. Mar. Fish. Rev. 62(4):40-45.

Loughlin, T. R., D. J. Rugh, and C. H. Fiscus. 1984. Northern sea lion distribution and abundance: 1956-1980. J. Wildl. Manage. 48:729-740.

Malavaer, M. Y. G. 2002. Modeling the energetics of Steller sea lions (*Eumetopias jubatus*) along the Oregon Coast. M.S. Thesis, Oregon State University, Corvallis, Oregon.

Maniscalco, J. M., P. Parker, and S. Atkinson. 2006. Interseasonal and interannual measures of maternal care among individual Steller sea lions (*Eumetopias jubatus*). J. Mammal. 87:304-311.

Manly, B. F. J. 2015. Incidental takes and interactions of marine mammals and birds in districts 6, 7, and 8 of the Southeast Alaska salmon drift gillnet fishery, 2012 and 2013. Final Report to NMFS Alaska Region. 52 p.

McBeath, J. 2004. Greenpeace v. National Marine Fisheries Service: Steller sea lions and commercial fisheries in the North Pacific. Alaska Law Rev. 21:1-42.

Melovidov, P. I. 2013. 2012 subsistence hunting of Steller sea lions on St. Paul Island. Memorandum for the Record, February 25, 2013, Aleut Community of St. Paul, Tribal Government, Ecosystem Conservation Office, St. Paul Island, Pribilof Islands, AK.

Melovidov, P. I. 2014. 2013 subsistence hunting of Steller sea lions on St. Paul Island. Memorandum for the Record, February 28, 2014, Aleut Community of St. Paul, Tribal Government, Ecosystem Conservation Office, St. Paul Island, Pribilof Islands, AK.

Melovidov, P. I. 2015. 2014 subsistence hunting of Steller sea lions on St. Paul Island. Memorandum for the Record, February 20, 2015, Aleut Community of St. Paul, Tribal Government, Ecosystem Conservation Office, St. Paul Island, Pribilof Islands, AK.

Melovidov, P. I. 2016. 2015 subsistence hunting of Steller sea lions on St. Paul Island. Memorandum for the Record, February 23, 2016, Aleut Community of St. Paul, Tribal Government, Ecosystem Conservation Office, St. Paul Island, Pribilof Islands, AK.

Merrick, R. L., T. R. Loughlin, and D. G. Calkins. 1987. Decline in abundance of the northern sea lion, *Eumetopias jubatus*, in 1956-86. Fish. Bull., U.S. 85:351-365.

Merrick, R. L., R. Brown, D. G. Calkins, and T. R. Loughlin. 1995. A comparison of Steller sea lion, *Eumetopias jubatus*, pup masses between rookeries with increasing and decreasing populations. Fish. Bull., U.S. 93:753-758.

Milette, L. L., and A. W. Trites. 2003. Maternal attendance patterns of Steller sea lions (*Eumetopias jubatus*) from stable and declining populations in Alaska. Can. J. Zool. 81:340-348.

National Marine Fisheries Service (NMFS). 1990. Final rule. Listing of Steller Sea Lions as Threatened Under the Endangered Species Act. 55 FR 24345, 26 November 1990.

National Marine Fisheries Service (NMFS). 1997. Final rule. Change in Listing Status of Steller Sea Lions Under the Endangered Species Act. 62 FR 24345, 5 May 1997.

National Marine Fisheries Service (NMFS). 2008. Recovery Plan for the Steller sea lion (*Eumetopias jubatus*). Revision. National Marine Fisheries Service, Silver Spring, MD. 325 p.

National Marine Fisheries Service (NMFS). 2010. Endangered Species Act - Section 7 Consultation Biological Opinion: Authorization of groundfish fisheries under the Fishery Management Plan for Groundfish of the Bering Sea and Aleutian Islands Management Area; Authorization of groundfish fisheries under the Fishery Management Plan for Groundfish of the Gulf of Alaska, State of Alaska parallel groundfish fisheries. Available online: https://www.fisheries.noaa.gov/resource/document/biological-opinion-authorization-alaska-groundfish-fisheries . Accessed December 2020.

National Marine Fisheries Service (NMFS). 2013. Occurrence of Western Distinct Population Segment Steller sea lions east of 144° W longitude. December 18, 2013. NMFS Alaska Region, Protected Resources Division, Juneau, AK. 3 p.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2020.

O'Corry-Crowe, G., B. L. Taylor, and T. Gelatt. 2006. Demographic independence along ecosystem boundaries in Steller sea lions revealed by mtDNA analysis: implications for management of an endangered species. Can. J. Zool. 84(12):1796-1809.

O'Corry-Crowe, G., T. Gelatt, L. Rea, C. Bonin, and M. Rehberg. 2014. Crossing to safety: dispersal, colonization and mate choice in evolutionarily distinct populations of Steller sea lions, *Eumetopias jubatus*. Mol. Ecol. 23(22):5415-5434.

Olesiuk, P. F. 2008. Abundance of Steller sea lions (*Eumetopias jubatus*) in British Columbia. Department of Fisheries and Oceans Canada, Canadian Science Advisory Secretariat Research Document 2008/063. 29 p. Available online: https://waves-vagues.dfo-mpo.gc.ca/Library/336057.pdf . Accessed December 2020.

Peterson, W., N. Bond, and M. Robert. 2016. The blob (part three): going, going, gone? PICES Press 24(1):46-48. Available online: https://search.proquest.com/docview/1785278412?accountid=28257 . Accessed December 2020.

Phillips, C. D., J. W. Bickham, J. C. Patton, and T. S. Gelatt. 2009. Systematics of Steller sea lions (*Eumetopias jubatus*): subspecies recognition based on concordance of genetics and morphometrics. Museum of Texas Tech University Occasional Papers 283:1-15.

Phillips, C. D., T. S. Gelatt, J. C. Patton, and J. W. Bickham. 2011. Phylogeography of Steller sea lions: relationships among climate change, effective population size, and genetic diversity. J. Mammal. 92(5):1091-1104.

Pitcher, K. W., V. N. Burkanov, D. G. Calkins, B. J. Le Boeuf, E. G. Mamaev, R. L. Merrick, and G. W. Pendleton. 2001. Spatial and temporal variation in the timing of births of Steller sea lions. J. Mammal. 82(4):1047-1053.

Pitcher, K. W., P. F. Olesiuk, R. F. Brown, M. S. Lowry, S. J. Jeffries, J. L. Sease, W. L. Perryman, C. E. Stinchcomb, and L. F. Lowry. 2007. Abundance and distribution of the eastern North Pacific Steller sea lion (*Eumetopias jubatus*) population. Fish. Bull., U.S. 105(1):102-115.

Rand, K., S. McDermott, E. Logerwell, M. E. Matta, M. Levine, D. R. Bryan, I. B. Spies, and T. Loomis. 2019. Higher aggregation of key prey species associated with diet and abundance of the Steller sea lion *Eumetopias jubatus* across the Aleutian Islands. Marine and Coastal Fisheries 11(6):472-486. DOI: dx.doi.org/10.1002/mcf2.10096 .

Raum-Suryan, K. L, K. W. Pitcher, D. G. Calkins, J. L. Sease, and T. R. Loughlin. 2002. Dispersal, rookery fidelity, and metapopulation structure of Steller sea lions (*Eumetopias jubatus*) in an increasing and a decreasing population in Alaska. Mar. Mammal Sci. 18(3):746-764. DOI: dx.doi.org/10.1111/j.1748-7692.2002.tb01071.x .

Raum-Suryan, K. L., L. A. Jemison, and K. W. Pitcher. 2009. Entanglement of Steller sea lions (*Eumetopias jubatus*) in marine debris: identifying causes and finding solutions. Mar. Pollut. Bull. 58:1487-1495.

Rea, L. D., J. M. Castellini, L. Correa, B. S. Fadely, and T. M. O'Hara. 2013. Maternal Steller sea lion diets elevate fetal mercury concentrations in an area of population decline. Science of the Total Environment 454-455:277-282. DOI: dx.doi.org/10.1016/j.scitotenv.2013.02.095 .

Rehberg, M., L. Jemison, J. N. Womble, and G. O'Corry-Crowe. 2018. Winter movements and long-term dispersal of Steller sea lions in the Glacier Bay region of Southeast Alaska. Endang. Species Res. 37:11-24. DOI: dx.doi.org/10.3354/esr00909 .

Sease, J. L., and C. J. Gudmundson. 2002. Aerial and land-based surveys of Steller sea lions (*Eumetopias jubatus*) from the western stock in Alaska, June and July 2001 and 2002. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-131, 54 p.

Sease, J. L., and A. E. York. 2003. Seasonal distribution of Steller's sea lions at rookeries and haul-out sites in Alaska. Mar. Mammal Sci. 19(4):745-763.

Sease, J. L., W. P. Taylor, T. R. Loughlin, and K. W. Pitcher. 2001. Aerial and land-based surveys of Steller sea lions (*Eumetopias jubatus*) in Alaska, June and July 1999 and 2000. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-122, 52 p.

Sinclair, E. H., D. S. Johnson, T. K. Zeppelin, and T. S. Gelatt. 2013. Decadal variation in the diet of Western stock Steller sea lions (*Eumetopias jubatus*). U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-248, 67 p.

Sweeney, K., L. Fritz, R. Towell, and T. Gelatt. 2016. Results of Steller sea lion surveys in Alaska, June-July 2016. Memorandum to D. DeMaster, J. Bengtson, J. Balsiger, J. Kurland, and L. Rotterman, December 5, 2016. Available from Marine Mammal Laboratory, AFSC, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Sweeney, K., L. Fritz, R. Towell, and T. Gelatt. 2017. Results of Steller sea lion surveys in Alaska, June-July 2017. Memorandum to the Record, December 5, 2017. Available from Marine Mammal Laboratory, AFSC, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Sweeney, K., R. Towell, and T. Gelatt. 2018. Results of Steller sea lion surveys in Alaska, June-July 2018. Memorandum to the Record, December 5, 2018. Available from Marine Mammal Laboratory, AFSC, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Sweeney, K., K. Luxa, B. Birkemeier, and T. Gelatt. 2019. Results of Steller sea lion surveys in Alaska, June-July 2019. Memorandum to the Record, December 6, 2019. Available from Marine Mammal Laboratory, AFSC, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Tollit, D., L. Fritz, R. Joy, K. Miller, A. Schulze, J. Thomason, W. Walker, T. Zeppelin, and T. Gelatt. 2017. Diet of endangered Steller sea lions (*Eumetopias jubatus*) in the Aleutian Islands: new insights from DNA detections and bioenergetics reconstructions. Can. J. Zool. 95:853-868. DOI: dx.doi.org/10.1139/cjz-2016-0253 .

VanWormer, E., J. A. K. Mazet, A. Hall, V. A. Gill, P. L. Boveng, J. M. London, T. Gelatt, B. S. Fadely, M. E. Lander, J. Sterling, V. N. Burkanov, R. R. Ream, P. M. Brock, L. D. Rea, B. R. Smith, A. Jeffers, M. Henstock, M. J. Rehberg, K. A. Burek-Huntington, S. L. Cosby, J. A. Hammond, and T. Goldstein. 2019. Viral emergence in marine mammals in the North Pacific may be linked to Arctic sea ice reduction. Scientific Reports 9, 15569. DOI: dx.doi.org/10.1038/s41598-019-51699-4 .

von Biela, V. R., M. L. Arimitsu, J. F. Piatt, B. Heflin, S. K. Schoen, J. L. Trowbridge, and C. M. Clawson. 2019. Extreme reduction in nutritional value of a key forage fish during the Pacific marine heatwave of 2014-2016. Mar. Ecol. Prog. Series 613:171-182. DOI: dx.doi.org/10.3354/meps12891 .

Wade, P. R. 1994. Managing populations under the Marine Mammal Protection Act of 1994: a strategy for selecting values for N_{MIN}, the minimum abundance estimate, and F_R, the recovery factor. Southwest Fisheries Science Center Administrative Report LJ-94-19, 26 p. Available from SWFSC, NMFS, 8901 La Jolla Shores Drive, La Jolla, CA 92037.

Williams, T. 2005. Reproductive energetics of sea lions: implications for the size of protected areas around Steller sea lion rookeries, p. 83-89. *In* T. R. Loughlin, D. G. Calkins, and S. Atkinson (eds.), Synopsis of research on Steller sea lions, 2001–2005. Alaska SeaLife Center, Seward, Alaska.

Winship, A. J., A. W. Trites, and D. A. S. Rosen. 2002. A bioenergetic model for estimating the food requirements of Steller sea lions *Eumetopias jubatus* in Alaska, USA. Mar. Ecol. Prog. Ser. 229:291-312.

Wolfe, R. J., J. A. Fall, and R. T. Stanek. 2005. The subsistence harvest of harbor seals and sea lions by Alaska Natives in 2004. Alaska Department of Fish and Game, Division of Subsistence, Technical Paper No. 303, Juneau, AK.

Wolfe, R. J., J. A. Fall, and R. T. Stanek. 2006. The subsistence harvest of harbor seals and sea lions by Alaska Natives in 2005. Alaska Department of Fish and Game, Division of Subsistence, Technical Paper No. 319, Juneau, AK.

Wolfe, R. J., J. A. Fall, and M. Riedel. 2008. The subsistence harvest of harbor seals and sea lions by Alaska Natives in 2006. Alaska Department of Fish and Game, Division of Subsistence, Technical Paper No. 339, Juneau, AK.

Wolfe, R. J., J. A. Fall, and M. Riedel. 2009a. The subsistence harvest of harbor seals and sea lions by Alaska Natives in 2007. Alaska Department of Fish and Game, Division of Subsistence, Technical Paper No. 345, Juneau, AK.

Wolfe, R. J., J. A. Fall, and M. Riedel. 2009b. The subsistence harvest of harbor seals and sea lions by Alaska Natives in 2008. Alaska Department of Fish and Game, Division of Subsistence, Technical Paper No. 347, Juneau, AK.

Wolfe, R. J., L. Hutchinson-Scarbrough, and M. Riedel. 2012. The subsistence harvest of harbor seals and sea lions on Kodiak Island in 2011. Alaska Department of Fish and Game, Division of Subsistence, Technical Paper No. 374, Anchorage, AK.

Wynne, K. M., D. Hicks, and N. Munro. 1991. 1990 salmon gillnet fisheries observer programs in Prince William Sound and South Unimak Alaska. Annual Report NMFS/NOAA Contract 50ABNF000036. 65 p. Available from NMFS Alaska Region, Office of Marine Mammals, P.O. Box 21668, Juneau, AK 99802.

Wynne, K. M., D. Hicks, and N. Munro. 1992. 1991 marine mammal observer program for the salmon driftnet fishery of Prince William Sound Alaska. Annual Report NMFS/NOAA Contract 50ABNF000036. 53 p. Available from NMFS Alaska Region, Office of Marine Mammals, P.O. Box 21668, Juneau, AK 99802.

Yang, Q., E. D. Cokelet, P. J. Stabeno, L. Li, A. B. Hollowed, W. A. Palsson, N. A. Bond, and S. Barbeaux. 2019. How "The Blob" affected groundfish distributions in the Gulf of Alaska. Fish. Oceanography 28(4):434-453. DOI: dx.doi.org/10.1111/fog.12422 .

York, A. E., R. L. Merrick, and T. R. Loughlin. 1996. An analysis of the Steller sea lion metapopulation in Alaska, Chapter 12, p. 259-292. *In* D. R. McCullough (ed.), Metapopulations and Wildlife Conservation. Island Press, Covelo, CA.

Young, N. C., B. J. Delean, V. T. Helker, J. C. Freed, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2014-2018. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-413, 142 p.

STELLER SEA LION (*Eumetopias jubatus*): Eastern U.S. Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

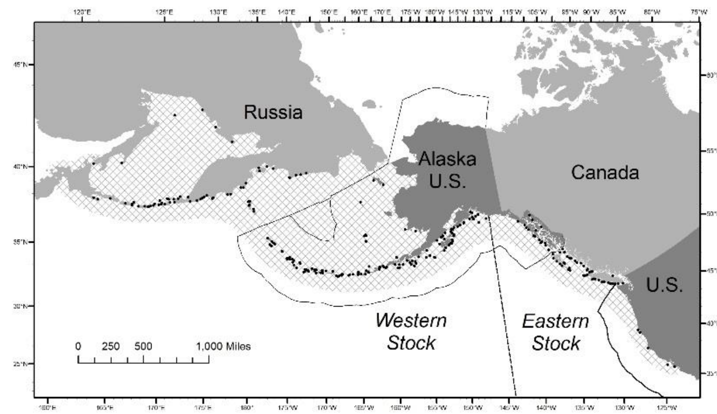

Figure 1. Generalized distribution (crosshatched area) of Steller sea lions in the North Pacific and major U.S. haulouts and rookeries (50 CFR 226.202, 27 August 1993), as well as active Asian and Canadian (British Columbia) haulouts and rookeries (points: Burkanov and Loughlin 2005; S. Majewski, Fisheries and Oceans Canada, pers. comm.). A black dashed line (144°W) indicates the stock boundary (Loughlin 1997) and a black line delineates the U.S. Exclusive Economic Zone.

Steller sea lions range along the North Pacific Rim from northern Japan to California (Loughlin et al. 1984) (Fig. 1). Large numbers of individuals disperse widely outside of the breeding season (late May to July), probably to access seasonally important prey resources. This results in marked seasonal patterns of abundance in some parts of the range and potential for intermixing in foraging areas of animals that were born in different areas (Sease and York 2003). There is an exchange of sea lions across the stock boundary (144°W; dashed line in Fig. 1), especially due to the wide-ranging seasonal movements of juveniles and adult males (Baker et al. 2005; Jemison et al. 2013, 2018). During the breeding season, sea lions, especially adult females, typically return to their natal rookery or a nearby breeding rookery to breed and pup (Raum-Suryan et al. 2002, Hastings et al. 2017). However, mixing of mostly breeding females from Prince William Sound to Southeast Alaska began in the 1990s and two new, mixed-stock rookeries were established (Gelatt et al. 2007; Jemison et al. 2013, 2018; O'Corry-Crowe et al. 2014).

Loughlin (1997) considered the following information when classifying stock structure based on the phylogeographic approach of Dizon et al. (1992): 1) Distributional data: geographic distribution continuous, yet a high degree of natal site fidelity and low (<10%) exchange rate of breeding animals among rookeries; 2) Population response data: substantial differences in population dynamics (York et al. 1996); 3) Phenotypic data: differences in pup mass (Merrick et al. 1995, Loughlin 1997); and 4) Genotypic data: substantial differences in mitochondrial DNA (Bickham et al. 1996). Based on this information, two separate stocks of Steller sea lions were recognized within U.S. waters: an Eastern U.S. stock, which includes animals born east of Cape Suckling, Alaska (144°W), and a Western U.S. stock, which includes animals born at and west of Cape Suckling (Loughlin 1997; Fig. 1). However, Jemison et al. (2013, 2018) determined there is regular movement of Steller sea lions from the western Distinct Population Segment (DPS) (males and females equally) and eastern DPS (almost exclusively males) across the DPS boundary. In this report, the western DPS is equivalent to the western stock and the eastern DPS is equivalent to the eastern stock.

All genetic analyses (Baker et al. 2005; Harlin-Cognato et al. 2006; Hoffman et al. 2006, 2009; O'Corry-Crowe et al. 2006) confirm a strong separation between western and eastern stocks, and there may be sufficient morphological differentiation to support elevating the two recognized stocks to subspecies (Phillips et al. 2009), although a review by Berta and Churchill (2012) characterized the status of these subspecies assignments as "tentative" and requiring further attention before their status can be determined. Work by Phillips et al. (2011) addressed the effect of climate change, in the form of glacial events, on the evolution of Steller sea lions and reported that the effective population size at the time of the event determines the impact of change on the population. The results suggested that during historic glacial periods, dispersal events were correlated with historically low effective population sizes, whereas range fragmentation type events were correlated with larger effective population sizes. This work again reinforced the stock delineation concept by noting that ancient population subdivision likely led to the sequestering of most mtDNA haplotypes as stock or subspecies-specific (Phillips et al. 2011).

In 1998 a single Steller sea lion pup was observed on Graves Rock just north of Cross Sound in Southeast Alaska, and within 15 years (2013) pup counts had increased to 551 (DeMaster 2014). Mitochondrial and microsatellite analysis of pup tissue samples collected in 2002 revealed that approximately 70% of the pups had mtDNA haplotypes that were consistent with those found in the western stock (Gelatt et al. 2007). Similarly, a rookery to the south on the White Sisters Islands, where pups were first noted in 1990, was also sampled in 2002 and approximately 45% of those pups had western stock haplotypes (O'Corry-Crowe et al. 2014). Collectively, this information demonstrates that these two most recently established rookeries in northern Southeast Alaska have been partially to predominately established by western stock females (Jemison et al. 2013, 2018; Rehberg et al. 2018). While movements of animals marked as pups in both stocks support these genetic results (Jemison et al. 2013, 2018), overall the observations of marked sea lion movements corroborate the extensive genetic research findings for a strong separation between the two currently recognized stocks. O'Corry-Crowe et al. (2014) concluded that the results of their study of the genetic characteristics of pups born on these new rookeries "demonstrates that resource limitation may trigger an exodus of breeding animals from declining populations, with substantial impacts on distribution and patterns of genetic variation. It also revealed that this event is rare because colonists dispersed across an evolutionary boundary, suggesting that the causative factors behind recent declines are unusual or of larger magnitude than normally occur." Thus, although recent colonization events in the northern part of the eastern stock indicate movement of western sea lions (especially adult females) into this area, the mixed part of the range remains geographically distinct (Jemison et al. 2013), and the overall discreteness of the eastern from the western stock remains distinct. Movement of western stock sea lions south of these rookeries and eastern stock sea lions moving to the west is less common (Jemison et al. 2013, O'Corry-Crowe et al 2014). Hybridization among subspecies and species along a contact zone such as now occurs near the stock boundary is not unexpected as the ability to interbreed is a primitive condition whereas reproductive isolation would be derived. In fact, as stated by NMFS and the U.S. Fish and Wildlife Service (USFWS) in a 1996 response to a previous comment regarding stock discreteness policy (61 FR 47222), "*The Services do not consider it appropriate to require absolute reproductive isolation as a prerequisite to recognizing a distinct population segment*" or stock. The fundamental concept overlying this distinctiveness is the collection of morphological, ecological, behavioral, and genetic evidence for stock differences initially described by Bickham et al. (1996) and Loughlin (1997) and supported by Baker et al. (2005), Harlin-Cognato et al. (2006), Hoffman et al. (2006, 2009), O'Corry-Crowe et al. (2006), and Phillips et al. (2009, 2011).

POPULATION SIZE

The eastern stock of Steller sea lions has historically bred on rookeries located in Southeast Alaska, British Columbia, Oregon, and California. However, within the last several years a new rookery has become established on the outer Washington coast (at the Carroll Island and Sea Lion Rock complex), with >100 pups born there in 2015 (R. DeLong and P. Gearin, NMFS-AFSC-MML, pers. comm.). Counts of pups on rookeries conducted near the end of the birthing season are nearly complete counts of pup production. The dates of the most recent aerial photographic and land-based surveys of eastern Steller sea lions have varied by region. Southeast Alaska was surveyed in June and July 2017 (Sweeney et al. 2017; NMFS, unpubl. data), while counts used in population analyses for the contiguous U.S. are from 2014 surveys in Washington (NMFS and Washington Department of Fish and Wildlife, unpubl. data) and 2017 surveys of Oregon and California (NMFS and Oregon Department of Fish and Game, unpubl. data). Counts from Canada (i.e., British Columbia) are from 2013 surveys (Olesiuk 2018; Fisheries and Oceans Canada, unpubl. data). For trend and population estimates, agTrend (an R package; Johnson and Fritz 2014) was used to augment missing counts in order to estimate 2017 counts. The 2017 estimated total eastern stock pup count is 18,450 (95% credible interval of 15,030-22,253). The 2017 estimated total eastern stock non-pup count is 58,699 (95% credible interval of 50,312-68,052). These estimates cannot be used to represent a total population abundance estimate as they do not account for animals at sea.

Minimum Population Estimate

Because current population size (N) and a pup multiplier to estimate N are not known, the best modeled estimates of the total count of eastern Steller sea lions is used as the minimum population estimate (N_{MIN}). These counts are considered minimum estimates of population size because they have not been corrected for animals that are at sea during, or pups born after, the surveys. The agTrend (Johnson and Fritz 2014) total count estimate of pups and non-pups for the entire eastern stock of Steller sea lions (including Canada; Olesiuk 2018) in 2017 is 77,149 (58,699 non-pups plus 18,450 pups). The total count estimate of pups and non-pups for the U.S. portion of the eastern stock of Steller sea lions (excluding Canada) is 43,201 (32,510 non-pups plus 10,691 pups) and is considered to be N_{MIN}.

Current Population Trend

Using agTrend, count data from 1971 to 2017 were modeled to estimate annual trends from 1987 to 2017 (30-year period). This model indicates the eastern stock of Steller sea lions increased at a rate of 4.25% per year (95% credible intervals of 3.77-4.72%) between 1987 and 2017 based on an analysis of pup counts in California, Oregon, Washington, British Columbia, and Southeast Alaska (Table 1, Figs. 2 and 3). A similar analysis of non-pup counts in the same regions yielded an estimate of population increase of 3.22% per year (95% credible intervals of 2.82-3.65%: Table 1). Pitcher et al. (2007) reported that the Eastern U.S. stock increased at a rate of 3.1% per year during a 25-year time period from 1977 to 2002; however, they used a slightly different method to estimate population growth than the methods reported in NMFS (2013). The Eastern U.S. stock increase has been driven by growth in pup counts in all regions (NMFS 2013).

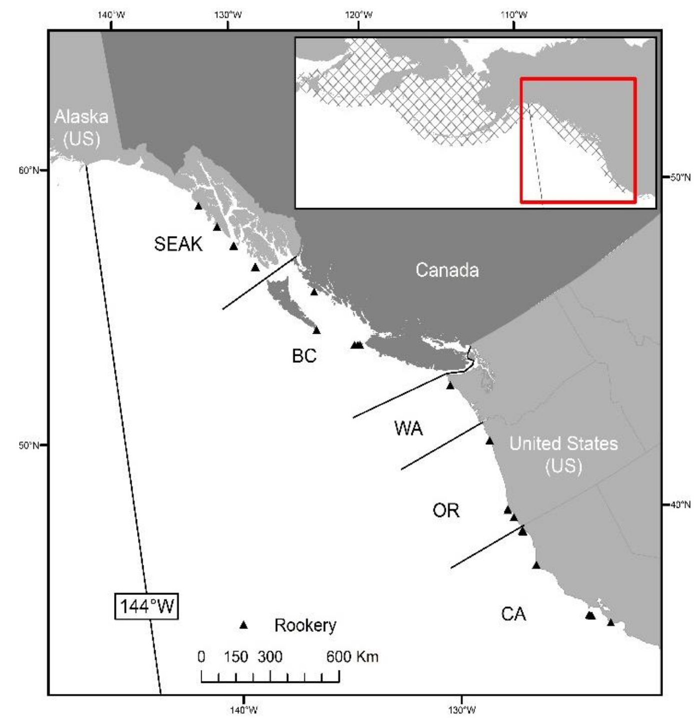

Figure 2. The eastern Steller sea lion rookery sites by region: Southeast Alaska (SEAK), British Columbia, Canada (BC), Washington State (WA), Oregon State (OR), and California State (CA).

Table 1. Trends (annual rates of change expressed as % y^{-1} with 95% credible interval) of eastern Steller sea lion non-pups (adults and juveniles) and pups, by region and total population, for 1987-2017 (Johnson and Fritz 2014, Sweeney et al. 2017).

	Non-Pup				**Pup**		
Region	**Trend**	**-95%**	**+95%**		**Trend**	**-95%**	**+95%**
California, U.S.	2.01	0.83	3.22		3.44	2.38	4.55
Oregon, U.S.	2.50	1.58	3.41		3.72	2.83	4.48
Washington, U.S.*	9.12	6.06	11.96				
British Columbia, Canada	4.18	3.47	4.96		6.91	5.89	7.91
Southeast Alaska, U.S.	2.45	1.85	3.08		3.04	2.49	3.60
Total Eastern Stock	3.22	2.82	3.65		4.25	3.77	4.72

*NMFS had not observed Steller sea lion pups born on known sites in Washington until a new rookery was established on the outer Washington coast (at the Carroll Island and Sea Lion Rock complex), with a confirmed count of 45 pups in 2013 and >100 pups in 2015 (R. DeLong and P. Gearin, NMFS-AFSC-MML, pers. comm.).

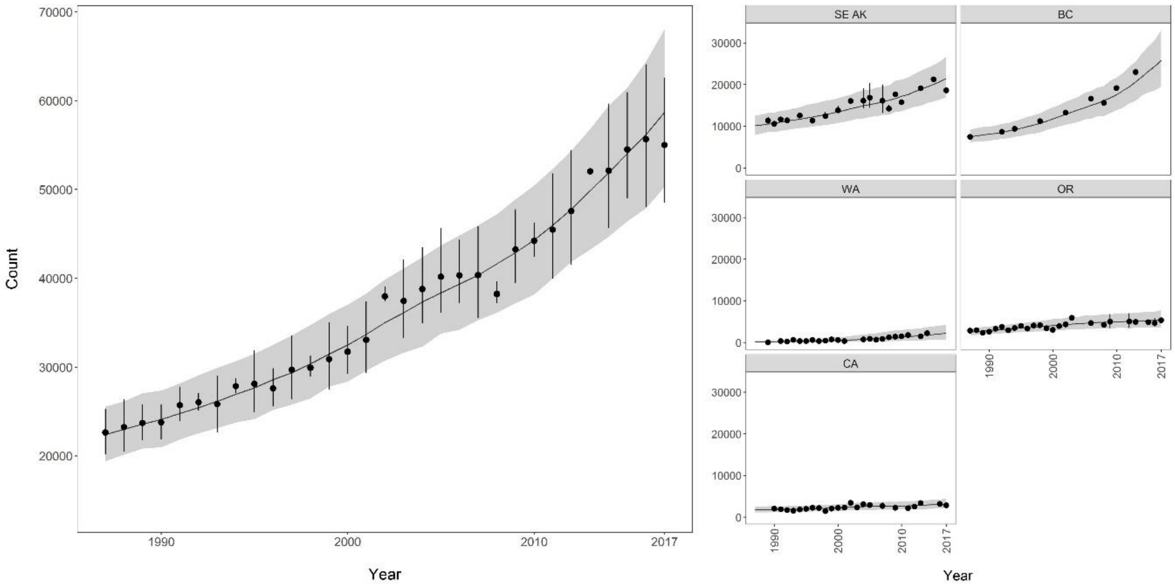

Figure 3. Estimated counts (modeled with agTrend) of Steller sea lion non-pups (adults and juveniles) for the eastern stock and the five regions: Southeast Alaska (SEAK), British Columbia, Canada (BC), Washington (WA), Oregon (OR), and California (CA) for 1987-2017 (Johnson and Fritz 2014, Sweeney et al. 2017).

While the eastern stock of Steller sea lions has been increasing in all regions from 1990 to 2017, the most significant growth has been observed in Southeast Alaska and British Columbia, Canada (Fig. 3). These two regions comprise almost 81% of the total eastern stock count. Non-pups in Oregon and Washington have been increasing since 1990, though at a lower rate. Non-pup counts in California ranged between 4,000 and 6,000 with no apparent trend from 1927 to 1947 but subsequently declined. At Año Nuevo Island off central California, a steady decline in abundance began in 1970 and there was an 85% reduction in the breeding population by 1987 (Le Boeuf et al. 1991). Non-pup counts increased slightly from 1989 to 2017, ranging from approximately 2,000 to 3,100.

Net movement between the eastern and western stocks appears to be small during the breeding season, with an estimated net 75 sea lions moving from east to west in 2016 (Jemison et al. 2013, Fritz et al. 2016). As a result, trends in counts estimated from breeding season surveys should be relatively insensitive to inter-stock movements. Very few females move from Southeast Alaska to the western stock while approximately 500 were estimated to move from west to east (net increase in the east). Males move in both directions but with a net increase in the west. This pattern of movement is supported by mitochondrial DNA evidence that indicated that the newest rookeries in northern Southeast Alaska (eastern stock) were colonized in part by western females (Gelatt et al. 2007, O'Corry-Crowe et al. 2014).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

There are no estimates of the maximum net productivity rate (R_{MAX}) for Steller sea lions. Pitcher et al. (2007) observed a rate of population increase of 3.1% per year for the eastern stock but concluded this rate did not represent a maximum rate of increase. NMFS (2013) estimated that the eastern stock increased at rates of 4.18% per year using pup counts and 2.99% per year using non-pup counts between 1979 and 2009. Here, we estimated that counts of pups and non-pups increased at rates of 4.25% and 3.22% per year, respectively, between 1987 and 2017 (Table 1). Until additional data become available, the maximum theoretical net productivity rate for pinnipeds of 12% will be used for this stock (NMFS 2016).

POTENTIAL BIOLOGICAL REMOVAL

Potential biological removal (PBR) is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: $PBR = N_{MIN} \times 0.5R_{MAX} \times F_R$. On 4 December 2013, the eastern DPS of Steller sea lions was removed from the list of threatened species under the Endangered Species Act (ESA; 78 FR 66140, 4 November 2013). NMFS' decision to delist this population was based on the information presented in the Status Review (NMFS 2013), the factors for delisting in section 4(a)(1) of the ESA, the biological and threats-based recovery criteria in the 2008 Recovery Plan (NMFS 2008), the continuing efforts to protect the species, and information received during public comment and peer review. NMFS' consideration of this information led to a determination that the eastern DPS has recovered and no longer meets the definition of a threatened species under the ESA. As recently noted within the humpback whale ESA listing final rule (81 FR 62259, 8 September 2016), in the case of a species or stock that achieved its depleted status solely on the basis of its ESA status, such as the eastern stock of Steller sea lions, the species or stock would cease to qualify as depleted under the terms of the definition set forth in MMPA Section 3(1) if the species or stock is no longer listed as threatened or endangered. Therefore, NMFS considers this stock not to be depleted; the recovery factor is 1.0 (recovery factor for a stock of unknown status that is known to be increasing), and the PBR = 2,592 (43,201 × 0.06 × 1.0).

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2013 and 2017 is listed, by marine mammal stock, in Delean et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports. The minimum estimated mean annual level of human-caused mortality and serious injury for Eastern U.S. Steller sea lions between 2013 and 2017 is 112 sea lions: 24 in U.S. commercial fisheries, 1.2 in recreational fisheries, 0.2 in subsistence fisheries, 32 in unknown (commercial, recreational, or subsistence) fisheries, 31 in marine debris, 13 due to other causes (illegally shot, explosives, ship strike, and incidental mortality during direct removals of California sea lions under authorization of MMPA Section 120 in response to their predation on endangered salmon and steelhead stocks in the Columbia River), and 11 in the Alaska Native subsistence harvest (from the 2005 to 2008 and 2012 data, which are the most recent data available). Additional potential threats most likely to result in direct human-caused mortality or serious injury of this stock include incidental take in unmonitored fisheries, unreported entanglement in marine debris, and disturbance at rookeries that could cause stampedes.

Fisheries Information

Information (including observer programs, observer coverage, and observed incidental takes of marine mammals) for federally-managed and state-managed U.S. commercial fisheries is presented in Appendices 3-6 of the Alaska Stock Assessment Reports (for fisheries in Alaska waters) and Appendix 1 of the U.S. Pacific Stock Assessment Reports (for fisheries in Washington, Oregon, and California waters).

Between 2013 and 2017, incidental mortality and serious injury of eastern Steller sea lions was observed in two of the federally-managed U.S. commercial fisheries in Alaska that are monitored for incidental mortality and serious injury by fisheries observers: the Gulf of Alaska halibut longline and Gulf of Alaska sablefish longline fisheries (Table 2; Breiwick 2013; MML, unpubl. data).

Mortality and serious injury of eastern Steller sea lions was also observed in six of the federally-managed U.S. commercial fisheries monitored by U.S. West Coast groundfish fisheries observers in 2012-2016: the Washington/Oregon/California (WA/OR/CA) groundfish bottom trawl (catch shares), WA/OR/CA groundfish midwater trawl (shoreside hake sector), WA/OR/CA groundfish midwater trawl (at-sea hake catcher-processor sector), WA/OR/CA groundfish midwater trawl (at-sea hake mothership catcher vessel sector), WA/OR/CA sablefish hook and line (limited entry), and California halibut bottom trawl (open access) fisheries (Table 2; Jannot et al. 2018; NWFSC, unpubl. data).

Mortality and serious injury of eastern Steller sea lions due to entanglement in Southeast Alaska commercial salmon drift gillnet (one in 2014) and interactions with Southeast Alaska commercial salmon troll gear (three in 2017) was reported by Marine Mammal Authorization Program (MMAP) fisherman self-reports and reports to the NMFS Alaska Region stranding network, respectively, between 2013 and 2017 (Table 3; Delean et al. 2020). Because observer data are not available for the Southeast Alaska commercial salmon drift gillnet and Southeast Alaska commercial salmon troll fisheries, this mortality and serious injury is used to calculate minimum mean annual mortality and serious injury rates of 0.2 and 0.6 eastern Steller sea lions, respectively, for these fisheries (Table 3). These mortality and serious injury estimates result from an actual count of verified human-caused deaths and serious injuries and are minimums because not all entangled animals strand or are self-reported nor are all stranded animals found, reported, or have the cause of death determined.

The minimum estimated mean annual mortality and serious injury rate incidental to U.S. commercial fisheries between 2013 and 2017 is 24 eastern Steller sea lions, based on observer data and stranding data (Tables 2 and 3). Due to limited observer program coverage, no data exist on the mortality of marine mammals incidental to Canadian commercial fisheries (i.e., those similar to U.S. fisheries known to take Steller sea lions). As a result, the number of Steller sea lions taken in Canadian waters is not known.

Table 2. Summary of incidental mortality and serious injury of Eastern U.S. Steller sea lions due to U.S. commercial fisheries between 2013 and 2017 (or the most recent data available) and calculation of the mean annual mortality and serious injury rate for Alaska fisheries (Breiwick 2013; MML, unpubl. data) and WA/OR/CA fisheries (Jannot et al. 2018; NWFSC, unpubl. data).

Fishery name	Years	Data type	Percent observer coverage	Observed mortality	Estimated mortality	Mean estimated annual mortality
Gulf of Alaska halibut longline	2013 2014 2015 2016 2017	obs data	4.2 11 9.4 9.5 4.6	0 0 1 0 0	0 0 12 0 0	2.4 (CV = 0.96)
Gulf of Alaska sablefish longline	2013 2014 2015 2016 2017	obs data	14 19 20 14 12	0 0 1 0 1	0 0 6.9 0 11	3.5 (CV = 0.69)
WA/OR/CA groundfish (bottom trawl - catch shares)	2012 2013 2014 2015 2016	obs data	99 100 100 100 100	8 6 5 8 0	8 6 5 8 0	5.4
WA/OR/CA groundfish (midwater trawl - shoreside hake sector)	2012 2013 2014 2015 2016	obs data	100 100 100 100 100	0 0 1 0 0	0 0 1 0 0	0.2
WA/OR/CA groundfish (midwater trawl - at-sea hake catcher-processor sector)	2012 2013 2014 2015 2016	obs data	100 100 100 100 100	1 2 3 0 21	1 2 3 0 21	5.4
WA/OR/CA groundfish (midwater trawl - at-sea hake mothership catcher vessel sector)	2012 2013 2014 2015 2016	obs data	98 100 100 100 100	0 0 1 0 2	0 0 1 0 2	0.6
WA/OR/CA sablefish (hook and line - limited entry)	2012 2013 2014 2015 2016	obs data	22 22 27 42 33	0 0 0 0 2	0.5 0.4 0.4 0.3 2.4	0.8
California halibut (bottom trawl - open access)	2012 2013 2014 2015 2016	obs data	6 6 22 33 30	0 0 0 3 3	2.7 3.4 3.2 6.1 6.1	4.3
Minimum total estimated annual mortality						23 (CV = 0.56)

Entanglement in marine debris and interactions with fisheries are a contributing factor in Steller sea lion injury and mortality (Raum-Suryan et al. 2009). Reports to the NMFS Alaska Region stranding network and the Alaska Department of Fish and Game (ADF&G) of Steller sea lions entangled in fishing gear or with injuries caused by interactions with gear provide additional information on fishery-related mortality and serious injury (Table 3;

Delean et al. 2020). Between 2013 and 2017, reports of Steller sea lion interactions with Southeast Alaska recreational salmon troll and Southeast Alaska recreational hook and line fisheries resulted in a minimum mean annual mortality and serious injury rate of 1.2 Steller sea lions in recreational fisheries. One mortality reported in a subsistence halibut longline fishery in 2017 resulted in a mean annual mortality and serious injury rate of 0.2 Steller sea lions in subsistence fisheries between 2013 and 2017. Steller sea lion interactions with troll fisheries between 2013 and 2017 resulted in mean annual mortality and serious injury rates of 3.4 sea lions in the Southeast Alaska salmon troll fishery and 27 in unidentified troll fisheries, including the dependent pup of a seriously injured animal. In all but one case (in which the animal was entangled in gear), the sea lions had either ingested troll gear or were hooked in the mouth; however, it is not clear whether these interactions involved recreational or commercial components of the fisheries. Other fishery-related mortality and serious injury of eastern Steller sea lions between 2013 and 2017 (and the resulting mean annual mortality and serious injury rates) was due to interactions with trawl gear (0.4) and hook and line gear (1.2). The minimum mean annual mortality and serious injury rate due to all non-commercial fishery interactions reported to the NMFS Alaska Region and ADF&G between 2013 and 2017 is 33 eastern Steller sea lions: 1.2 in recreational fisheries + 0.2 in subsistence fisheries + 32 in unknown (commercial, recreational, or subsistence) fisheries (Table 3; Delean et al. 2020). These mortality and serious injury estimates result from an actual count of verified human-caused deaths and serious injuries and are minimums because not all entangled animals strand or are self-reported nor are all stranded animals found, reported, or have the cause of death determined.

An additional eight Steller sea lions initially considered seriously injured in marine debris (one in 2014, one in 2015, and four in 2017), hook and line gear (one in 2016), and Southeast Alaska salmon troll gear (one in 2017) were disentangled and released with non-serious injuries in Alaska waters, and one Steller sea lion pup with serious injuries caused by human harassment was rehabilitated and released with non-serious injuries in Washington waters in 2014 (Delean et al. 2020). None of these animals were included in the average annual mortality and serious injury rate for 2013 to 2017.

Table 3. Summary of Eastern U.S. Steller sea lion mortality and serious injury, by year and type, reported to the NMFS Alaska Region marine mammal stranding network and ADF&G, and by fishermen self-reports, between 2013 and 2017 (Delean et al. 2020).

Cause of injury	**2013**	**2014**	**2015**	**2016**	**2017**	**Mean annual mortality**
Entangled in Southeast Alaska commercial salmon drift gillnet	0	1[a]	0	0	0	0.2
Hooked by Southeast Alaska commercial salmon troll gear	0	0	0	0	3	0.6
Hooked by SE Alaska recreational salmon troll gear	0	1	0	0	4	1
Hooked by Southeast Alaska recreational hook and line gear	0	0	1	0	0	0.2
Hooked by subsistence halibut longline gear	0	0	0	0	1	0.2
Hooked by Southeast Alaska salmon troll gear*	3	8	6	0	0	3.4
Hooked by troll gear*	3	41	26	42	17	26
Dependent pup of animal seriously injured (hooked) by troll gear*	0	0	0	1	0	0.2
Entangled in troll gear*	0	0	0	1	1	0.4
Entangled in trawl gear*	0	1	0	0	1	0.4
Hooked by hook and line gear*	0	0	0	2	2	0.8
Entangled in hook and line gear*	0	0	1	1	0	0.4
Entangled in marine debris	-	26	26	34	28	29[b]
Dependent pup of animal seriously injured by marine debris	-	3	2	2	0	1.8[b]
Illegally shot[c]	17	13	15	13	1	12

Cause of injury	2013	2014	2015	2016	2017	Mean annual mortality
Dependent pup of animal illegally shot[c]	0	1	0	0	0	0.2
Explosives	0	1	0	0	0	0.2
Ship strike	0	0	0	1	0	0.2
Incidental mortality during direct removals of California sea lions	0	0	1	0	0	0.2
Total in commercial fisheries						0.8
Total in recreational fisheries						1.2
Total in subsistence fisheries						0.2
*Total in unknown (commercial, recreational, or subsistence) fisheries						32
Total in marine debris						31
Total due to other sources (illegally shot, explosives, ship strike, incidental mortality during direct removals of California sea lions)						13

[a]Marine Mammal Authorization Program (MMAP) fisherman self report.
[b]A 4-year average (using 2014 to 2017 data) was calculated for this category, since we did not receive data on mortality and serious injury due to marine debris entanglement from the ADF&G in 2013.
[c]Only animals reported to the NMFS West Coast Region are included in this table because animals reported to the NMFS Alaska Region are likely accounted for as "struck and lost" in the Alaska Native harvest.

The minimum estimated mean annual mortality and serious injury rate incidental to all fisheries between 2013 and 2017 is 57 Steller sea lions: 24 in U.S. commercial fisheries + 1.2 in recreational fisheries + 0.2 in subsistence fisheries + 32 in unknown (commercial, recreational, or subsistence) fisheries.

Alaska Native Subsistence/Harvest Information

Information on the subsistence harvest of Steller sea lions is provided by the ADF&G. The ADF&G conducted systematic interviews with hunters and users of marine mammals in approximately 2,100 households in about 60 coastal communities within the geographic range of the Steller sea lion in Alaska in 2005-2008 (Wolfe et al. 2006, 2008, 2009a, 2009b). The interviews were conducted once per year in the winter (January to March) and covered hunter activities for the previous calendar year. Approximately 16 of the interviewed communities lie within the range of the Eastern U.S. stock. As of 2009, annual statewide data on community subsistence harvests are no longer being consistently collected. Data are being collected periodically in subareas. Between 2010 and 2017, monitoring occurred only in 2012 (Wolfe et al. 2013), when one animal was landed and eight animals were struck and lost. Therefore, the most recent 5 years of data (2005 to 2008 and 2012) will be used for calculating an annual mortality and serious injury estimate. The average number of animals harvested plus struck and lost is 11 animals per year during this 5-year period (Table 4).

An unknown number of Steller sea lions from this stock are harvested by subsistence hunters in Canada. The magnitude of the Canadian subsistence harvest is believed to be small (Fisheries and Oceans Canada 2010). Alaska Native subsistence hunters have initiated discussions with Canadian hunters to quantify their respective subsistence harvests, and to identify any effect these harvests may have on management of the stock.

Table 4. Summary of the subsistence harvest data for Eastern U.S. Steller sea lions from 2005 to 2008 and in 2012. As of 2009, data on community subsistence harvests are no longer being consistently collected at a statewide level. Therefore, the most recent 5 years of data (2005 to 2008 and 2012) will be used for calculating an annual mortality and serious injury estimate.

Year	Number harvested	Number struck and lost	Estimated total number taken
2005	0	19	19[a]
2006	2.5	10.1	12.6[b]
2007	0	6.1	6.1[c]
2008	1.7	8.0	9.7[d]
2012	1	8	9[e]
Mean annual take (2005-2008 and 2012)	1.0	10	11

[a]Wolfe et al. (2006); [b]Wolfe et al. (2008); [c]Wolfe et al. (2009a); [d]Wolfe et al. (2009b); [e]Wolfe et al. (2013).

Other Mortality

Steller sea lions were taken in British Columbia during commercial salmon farming operations. Preliminary figures from the British Columbia Aquaculture Predator Control Program indicated a mean annual mortality of 45.8 Steller sea lions from this stock from 1999 to 2003 (Olesiuk 2004). Starting in 2004, aquaculture facilities were no longer permitted to shoot Steller sea lions (P. Olesiuk, Pacific Biological Station, BC, Canada, pers. comm.). However, Fisheries and Oceans Canada (2010) summarized that "illegal and undocumented killing of Steller Sea Lions is likely to occur in B.C." and reported "[s]everal cases of illegal kills have been documented (Fisheries and Oceans Canada, unpubl. data), and mortality may also occur outside of the legal parameters assigned to permit holders (e.g., for predator control or subsistence harvest)" but "…data on these activities are currently lacking."

Illegal shooting of sea lions in U.S. waters was thought to be a potentially significant source of mortality prior to the listing of sea lions as threatened under the ESA in 1990. Steller sea lion mortality and serious injury caused by gunshot wounds is reported to the NMFS Alaska Region and the NMFS West Coast Region stranding networks. Between 2013 and 2017, 59 animals with gunshot wounds were reported to the NMFS West Coast Region stranding network, resulting in a minimum mean annual mortality and serious injury rate of 12 Steller sea lions illegally shot from this stock plus 0.2 dependent pups of seriously injured animals (Table 3; Delean et al. 2020). An additional two Steller sea lions with gunshot wounds were reported to the NMFS Alaska Region stranding network between 2013 and 2017 (one each in 2016 and 2017). Although it is likely that illegal shooting does occur in Alaska, these events are not included in the estimate of the average annual mortality and serious injury rate because it could not be confirmed that the deaths were due to illegal shooting and were not already accounted for in the estimate of animals struck and lost in the Alaska Native subsistence harvest. Other non-fishery human-caused mortality and serious injury of Steller sea lions reported to the NMFS Alaska Region stranding network between 2013 and 2017 (and the resulting minimum mean annual mortality and serious injury rates) were due to entanglement in marine debris (29), dependent pups of animals seriously injured by marine debris (1.8), explosives (0.2), ship strikes (0.2), and incidental mortality (0.2) during direct removals of California sea lions under authorization of MMPA Section 120 in response to their predation on endangered salmon and steelhead stocks in the Columbia River (Table 3; Delean et al. 2020). These estimates result from an actual count of verified human-caused deaths and serious injuries and are minimums because not all animals strand or are self-reported nor are all stranded animals found, reported, or have the cause of death determined (via necropsy by trained personnel), and human-related stranding data are not available for British Columbia.

STATUS OF STOCK

Based on currently available data, the minimum estimated mean annual U.S. commercial fishery-related mortality and serious injury rate for this stock (24 sea lions) is less than 10% of the calculated PBR (10% of PBR = 259) and, therefore, can be considered to be insignificant and approaching a zero mortality and serious injury rate. The minimum estimated mean annual level of human-caused mortality and serious injury (112 sea lions) does not exceed the PBR (2,592) for this stock. The Eastern U.S. stock of Steller sea lions is not listed under the ESA and is not considered depleted under the MMPA. This stock is classified as a non-strategic stock. Because the counts of eastern Steller sea lions have steadily increased over a 30+ year period, this stock is likely within its Optimum Sustainable Population (OSP); however, no determination of its status relative to OSP has been made.

There are key uncertainties in the assessment of the Eastern U.S. stock of Steller sea lions. There is some overlap in range between animals in the western and eastern stocks in northern Southeast Alaska. The population is based on counts of visible animals; the calculated N_{MIN} and PBR levels are conservative because there are no data available to correct for animals not visible during the visual surveys. There are multiple nearshore commercial fisheries which are not observed; thus, there is likely to be unreported fishery-related mortality and serious injury of Steller sea lions. Estimates of human-caused mortality and serious injury from stranding data are underestimates because not all animals strand nor are all stranded animals found, reported, or have the cause of death determined.

HABITAT CONCERNS

Unlike the Western U.S. stock of Steller sea lions, there has been a sustained and robust increase in abundance of the Eastern U.S. stock throughout its breeding range. In the southern end of its range (Channel Islands in southern California), it has declined considerably since the late 1930s and several rookeries and haulouts south of Año Nuevo Island have been abandoned. Changes in the ocean environment, particularly warmer temperatures, may be factors that have favored California sea lions over Steller sea lions in the southern portion of the Steller sea lion's range (NMFS 2008). The risk of oil spills to this stock may increase in the next several decades due to increased shipping, including tanker traffic, from ports in British Columbia and possibly Washington State (COSEWIC 2013, NMFS 2013, Wiles 2014) and LNG facility and pipeline construction (COSEWIC 2013).

CITATIONS

Baker, A. R., T. R. Loughlin, V. Burkanov, C. W. Matson, T. G. Trujillo, D. G. Calkins, J. K. Wickliffe, and J. W. Bickham. 2005. Variation of mitochondrial control region sequences of Steller sea lions: the three-stock hypothesis. J. Mammal. 86:1075-1084.

Berta, A., and M. Churchill. 2012. Pinniped taxonomy: review of currently recognized species and subspecies, and evidence used for their description. Mammal Rev. 42(2):207-234.

Bickham, J. W., J. C. Patton, and T. R. Loughlin. 1996. High variability for control-region sequences in a marine mammal: implications for conservation and biogeography of Steller sea lions (*Eumetopias jubatus*). J. Mammal. 77:95-108.

Breiwick, J. M. 2013. North Pacific marine mammal bycatch estimation methodology and results, 2007-2011. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-260, 40 p.

Burkanov, V., and T. R. Loughlin. 2005. Distribution and abundance of Steller sea lions on the Asian coast, 1720's–2005. Mar. Fish. Rev. 67(2):1-62.

Committee on the Status of Endangered Wildlife in Canada (COSEWIC). 2013. COSEWIC assessment and status report on the Steller sea lion *Eumetopias jubatus* in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa, Canada. xi + 54 p. Available online: https://www.sararegistry.gc.ca/virtual_sara/files/cosewic/sr_Steller%20Sea%20Lion_2013_e.pdf . Accessed December 2019.

Delean, B. J., V. T. Helker, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, J. Jannot, and N. C. Young. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2013-2017. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-401, 86 p.

DeMaster, D. 2014. Results of Steller sea lion surveys in Alaska, June-July 2013. Memorandum to J. Balsiger, J. Kurland, B. Gerke, and L. Rotterman, January 27, 2014, NMFS Alaska Regional Office, Juneau AK. Available from Alaska Fisheries Science Center, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Dizon, A. E., C. Lockyer, W. F. Perrin, D. P. DeMaster, and J. Sisson. 1992. Rethinking the stock concept: a phylogeographic approach. Conserv. Biol. 6:24-36.

Fisheries and Oceans Canada. 2010. Management Plan for the Steller Sea Lion (*Eumetopias jubatus*) in Canada [Final]. Species at Risk Act Management Plan Series. Fisheries and Oceans Canada, Ottawa. vi + 69 p.

Fritz, L., K. Sweeney, R. Towell, and T. Gelatt. 2016. Aerial and ship-based surveys of Steller sea lions (*Eumetopias jubatus*) conducted in Alaska in June-July 2013 through 2015, and an update on the status and trend of the western distinct population segment in Alaska. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-321, 72 p.

Gelatt, T., A. W. Trites, K. Hastings, L. Jemison, K. Pitcher, and G. O'Corry-Crowe. 2007. Population trends, diet, genetics, and observations of Steller sea lions in Glacier Bay National Park, p. 145-149. *In* J. F. Piatt and S. M. Gende (eds.), Proceedings of the Fourth Glacier Bay Science Symposium, October 26-28, 2004: U.S. Geological Survey Scientific Investigations Report 2007-5047.

Harlin-Cognato, A., J. W. Bickham, T. R. Loughlin, and R. L. Honeycutt. 2006. Glacial refugia and the phylogeography of Steller's sea lion (*Eumetopias jubatus*) in the North Pacific. J. Evol. Biol. 19:955-969. DOI: dx.doi.org/10.1111/j.1420-9101.2005.01052.x .

Hastings, K. K., L. A. Jemison, G. W. Pendleton, K. L. Raum-Suryan, and K. W. Pitcher. 2017. Natal and breeding philopatry of female Steller sea lions in southeastern Alaska. PLoS ONE 13(4):e0196412. DOI: dx.doi.org/10.1371/journal.pone.0176840 .

Hoffman, J. I., C. W. Matson, W. Amos, T. R. Loughlin, and J. W. Bickham. 2006. Deep genetic subdivision within a continuously distributed and highly vagile marine mammal, the Steller's sea lion (*Eumetopias jubatus*). Mol. Ecol. 15:2821-2832.

Hoffman, J. I., K. K. Dasmahapatra, W. Amos, C. D. Phillips, T. S. Gelatt, and J. W Bickham. 2009. Contrasting patterns of genetic diversity at three different genetic markers in a marine mammal metapopulation. Mol. Ecol. 18(14):2961-2978.

Jannot, J. E., K. A. Somers, V. Tuttle, J. McVeigh, J. V. Carretta, and V. Helker. 2018. Observed and estimated marine mammal bycatch in U.S. west coast groundfish fisheries, 2002–16. U.S. Dep. Commer., NWFSC Processed Report 2018-03, 36 p. DOI: dx.doi.org/10.25923/fkf8-0x49 .

Jemison, L. A., G. W. Pendleton, L. W. Fritz, K. K. Hastings, J. M. Maniscalco, A. W. Trites, and T. S. Gelatt. 2013. Inter-population movements of Steller sea lions in Alaska with implications for population separation. PLoS ONE 8(8):e70167.

Jemison, L. A., G. W. Pendleton, K. K. Hastings, J. M. Maniscalco, and L. W. Fritz. 2018. Spatial distribution, movements, and geographic range of Steller sea lions (*Eumetopias jubatus*) in Alaska. PLoS ONE 13:e0208093.

Johnson, D. S., and L. W. Fritz. 2014. agTrend: a Bayesian approach for estimating trends of aggregated abundance. Methods Ecol. Evol. 5:1110-1115. DOI: dx.doi.org/10.1111/2041-210X.12231 .

Le Boeuf, B. J., K. Ono, and J. Reiter. 1991. History of the Steller sea lion population at Año Nuevo Island, 1961-1991. Southwest Fisheries Science Center Admin. Rep. LJ-91-45C. 9 p. + tables + figs. Available from Southwest Fisheries Science Center, 8901 La Jolla Shores Drive, La Jolla, CA 92037.

Loughlin, T. R. 1997. Using the phylogeographic method to identify Steller sea lion stocks, p. 329-341. *In* A. Dizon, S. J. Chivers, and W. Perrin (eds.), Molecular genetics of marine mammals, incorporating the proceedings of a workshop on the analysis of genetic data to address problems of stock identity as related to management of marine mammals. Soc. Mar. Mammal., Spec. Rep. No. 3.

Loughlin, T. R., D. J. Rugh, and C. H. Fiscus. 1984. Northern sea lion distribution and abundance: 1956-1980. J. Wildl. Manage. 48:729-740.

Merrick, R. L., R. Brown, D. G. Calkins, and T. R. Loughlin. 1995. A comparison of Steller sea lion, *Eumetopias jubatus*, pup masses between rookeries with increasing and decreasing populations. Fish. Bull., U.S. 93:753-758.

National Marine Fisheries Service (NMFS). 2008. Recovery Plan for the Steller sea lion (*Eumetopias jubatus*). Revision. National Marine Fisheries Service, Silver Spring, MD. 325 p.

National Marine Fisheries Service (NMFS). 2013. Status review of the eastern Distinct Population Segment of Steller sea lion (*Eumetopias jubatus*). 144 p. + appendices. Protected Resources Division, Alaska Region, NMFS, 709 West 9th Street, Juneau, AK 99802.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2019.

O'Corry-Crowe, G., B. L. Taylor, and T. Gelatt. 2006. Demographic independence along ecosystem boundaries in Steller sea lions revealed by mtDNA analysis: implications for management of an endangered species. Can. J. Zool. 84(12):1796-1809.

O'Corry-Crowe, G., T. Gelatt, L. Rea, C. Bonin, and M. Rehberg. 2014. Crossing to safety: dispersal, colonization and mate choice in evolutionarily distinct populations of Steller sea lions, *Eumetopias jubatus*. Mol. Ecol. 23(22):5415-5434.

Olesiuk, P. F. 2004. Status of sea lions (*Eumetopias jubatus* and *Zalophus californianus*) wintering off southern Vancouver Island. NMMRC Working Paper No. 2004-03 (DRAFT).

Olesiuk, P. F. 2018. Recent trends in abundance of Steller sea lions (*Eumetopias jubatus*) in British Columbia. DFO Can. Sci. Advis. Sec. Res. Doc. 2018/006. v + 67 p.

Phillips, C. D., J. W. Bickham, J. C. Patton, and T. S. Gelatt. 2009. Systematics of Steller sea lions (*Eumetopias jubatus*): subspecies recognition based on concordance of genetics and morphometrics. Museum of Texas Tech University Occasional Papers 283:1-15.

Phillips, C. D., T. S. Gelatt, J. C. Patton, and J. W. Bickham. 2011. Phylogeography of Steller sea lions: relationships among climate change, effective population size, and genetic diversity. J. Mammal. 92(5):1091-1104.

Pitcher, K. W., P. F. Olesiuk, R. F. Brown, M. S. Lowry, S. J. Jeffries, J. L. Sease, W. L. Perryman, C. E. Stinchcomb, and L. F. Lowry. 2007. Abundance and distribution of the eastern North Pacific Steller sea lion (*Eumetopias jubatus*) population. Fish. Bull., U.S. 105(1):102-115.

Raum-Suryan, K. L., K. W. Pitcher, D. G. Calkins, J. L. Sease, and T. R. Loughlin. 2002. Dispersal, rookery fidelity, and metapopulation structure of Steller sea lions (*Eumetopias jubatus*) in an increasing and a decreasing population in Alaska. Mar. Mammal Sci. 18(3):746-764. DOI: dx.doi.org/10.1111/j.1748-7692.2002.tb01071.x .

Raum-Suryan, K. L., L. A. Jemison, and K. W. Pitcher. 2009. Entanglement of Steller sea lions (*Eumetopias jubatus*) in marine debris: identifying causes and finding solutions. Mar. Pollut. Bull. 58:1487-1495.

Rehberg, M., L. Jemison, J. N. Womble, and G. O'Corry-Crowe. 2018. Winter movements and long-term dispersal of Steller sea lions in the Glacier Bay region of Southeast Alaska. Endang. Species Res. 37:11-24. DOI: dx.doi.org/10.3354/esr00909 .

Sease, J. L., and A. E. York. 2003. Seasonal distribution of Steller's sea lions at rookeries and haul-out sites in Alaska. Mar. Mammal Sci. 19(4):745-763.

Sweeney, K., L. Fritz, R. Towell, and T. Gelatt. 2017. Results of Steller sea lion surveys in Alaska, June-July 2017. Memorandum to the Record, December 5, 2017. Available from Marine Mammal Laboratory, AFSC, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Wiles, G. J. 2014. Draft Washington State periodic status review for the Steller sea lion. Washington Department of Fish and Wildlife, Olympia, WA. 35 p.

Wolfe, R. J., J. A. Fall, and R. T. Stanek. 2006. The subsistence harvest of harbor seals and sea lions by Alaska Natives in 2005. Alaska Department of Fish and Game, Division of Subsistence, Technical Paper No. 319, Juneau, AK.

Wolfe, R. J., J. A. Fall, and M. Riedel. 2008. The subsistence harvest of harbor seals and sea lions by Alaska Natives in 2006. Alaska Department of Fish and Game, Division of Subsistence, Technical Paper No. 339, Juneau, AK.

Wolfe, R. J., J. A. Fall, and M. Riedel. 2009a. The subsistence harvest of harbor seals and sea lions by Alaska Natives in 2007. Alaska Department of Fish and Game, Division of Subsistence, Technical Paper No. 345, Juneau, AK.

Wolfe, R. J., J. A. Fall, and M. Riedel. 2009b. The subsistence harvest of harbor seals and sea lions by Alaska Natives in 2008. Alaska Department of Fish and Game, Division of Subsistence, Technical Paper No. 347, Juneau, AK.

Wolfe, R. J., J. Bryant, L. Hutchinson-Scarbrough, M. Kookesh, and L. A. Sill. 2013. The subsistence harvest of harbor seals and sea lions in Southeast Alaska in 2012. Alaska Department of Fish and Game, Division of Subsistence, Technical Paper No. 383, Anchorage, AK.

York, A. E., R. L. Merrick, and T. R. Loughlin. 1996. An analysis of the Steller sea lion metapopulation in Alaska, p. 259-292. *In* D. R. McCullough (ed.), Metapopulations and Wildlife Conservation. Island Press, Covelo, CA.

NORTHERN FUR SEAL (*Callorhinus ursinus*): Eastern Pacific Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Northern fur seals occur from southern California north to the Bering Sea (Fig. 1) and west to the Sea of Okhotsk and Honshu Island, Japan. During the summer breeding season, most of the worldwide population is found on the Pribilof Islands (St. Paul Island and St. George Island) in the southern Bering Sea, with the remaining animals on rookeries in Russia, on Bogoslof Island in the southern Bering Sea, on San Miguel Island off southern California (Lander and Kajimura 1982, NMFS 1993), and on the Farallon Islands off central California. Non-breeding northern fur seals may occasionally haul out on land at other sites in Alaska, British Columbia, and on islets along the west coast of the United States (Fiscus 1983).

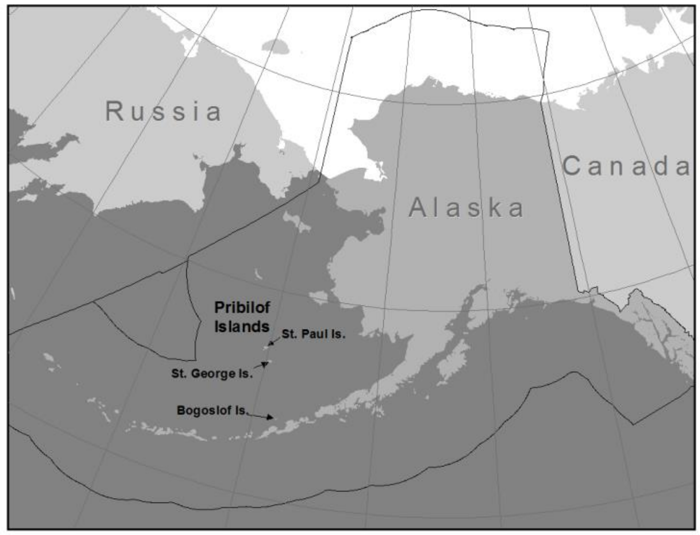

Figure 1. Approximate distribution of northern fur seals in the eastern North Pacific (dark shaded area). Eastern Pacific northern fur seal breeding colonies in U.S. waters are located on the three named islands. The U.S. Exclusive Economic Zone is delineated by a black line.

During the reproductive season, adult males usually are on shore during the 4-month period from May to August, although some may be present until November (well after giving up their territories). Adult females are ashore during a 6-month period (June-November). Following their respective times ashore, Alaska northern fur seals of both genders then move south and remain at sea until the next breeding season (Roppel 1984). Adult females and pups from the Pribilof Islands move through the Aleutian Islands into the North Pacific Ocean, often to the waters offshore of Oregon and California (Ream et al. 2005). Adult males generally move only as far south as the Gulf of Alaska in the eastern North Pacific (Kajimura 1984) and the Kuril Islands in the western North Pacific (Loughlin et al. 1999). In Alaska, pups are born during summer months and leave the rookeries in the fall, on average around mid-November but ranging from late October to early December. Alaska northern fur seal pups generally remain at sea for 22 months (Kenyon and Wilke 1953) before returning to land, usually at their rookery of birth but with considerable interchange of individuals between rookeries.

Two separate stocks of northern fur seals, an Eastern Pacific stock and a California stock, are recognized within U.S. waters based on the distribution and population response factors of the Dizon et al. (1992) phylogeographic approach: 1) Distribution: continuous during non-breeding season and discontinuous during the breeding season, high natal site fidelity (DeLong 1982, Baker et al. 1995); 2) Population response: substantial differences in population dynamics between the Pribilof Islands and San Miguel Island (DeLong 1982, DeLong and Antonelis 1991, NMFS 1993); 3) Phenotypic differentiation: unknown; and 4) Genotypic differentiation: little evidence of genetic differentiation among breeding islands (Ream 2002, Dickerson et al. 2010). The California stock is reported in the Stock Assessment Reports for the U.S. Pacific Region.

POPULATION SIZE

The population estimate for the Eastern Pacific stock of northern fur seals is calculated as the estimated number of pups born at rookeries in the eastern Bering Sea multiplied by a series of expansion factors determined from a life table analysis to estimate the number of yearlings, 2-year-olds, 3-year-olds, and animals 4 or more years old (Lander 1981). The resulting population estimate is equal to the pup production estimate multiplied by 4.47. The expansion factor is based on a sex and age distribution estimated after the harvest of juvenile males was terminated. There is no coefficient of variation (CV) for the expansion factor. Pup production is estimated at all islands using a mark-recapture method, or “shear-sampling” (Chapman and Johnson 1968, York and Kozloff 1987, Towell et al. 2006), with the exception of estimates conducted at Bogoslof Island through 1995, where the smaller population size in those years allowed direct counting of pups. As the majority of pups are born on St. Paul and St. George Islands, pup surveys are conducted biennially on these islands. Pup production estimates are available less frequently on Sea Lion Rock (adjacent to St. Paul Island) and Bogoslof Island (Table 1). Annual variation in female reproductive rates is reflected in the respective pup production estimates. Because the estimation of stock population size relies on these estimates of pup production, means of recent pup production estimates are used to account for variability in the reproductive rates over time. The most recent estimate for the number of northern fur seals in the Eastern Pacific stock, based on pup production estimates on Sea Lion Rock (2014), on St. Paul and St. George Islands (mean of 2014, 2016, and 2018), and on Bogoslof Island (2015), is 608,143 northern fur seals (4.47 × 136,050).

Table 1. Estimates and/or counts of northern fur seal pups born on the Pribilof Islands and Bogoslof Island. Standard errors for pup estimates at rookery locations and the CV for total pup production estimates are provided in parentheses (direct counts do not have standard errors). The “ symbol indicates that no new data are available for that year and, thus, the most recent prior estimate/count was used in determining total annual estimates.

	Rookery location				
Year	**St. Paul**	**Sea Lion Rock**	**St. George**	**Bogoslof**	**Total**
1994	192,104 (8,180)	12,891 (989)	22,244 (410)	1,472 (N/A)	228,711 (0.036)
1995	“	“	“	1,272 (N/A)	228,511 (0.036)
1996	170,125 (21,244)	“	27,385 (294)		211,673 (0.10)
1997	“	“	“	5,096 (33)	215,497 (0.099)
1998	179,149 (6,193)	“	22,090 (222)		219,226 (0.029)
2000	158,736 (17,284)	“	20,176 (271)	“	196,899 (0.089)
2002	145,716 (1,629)	8,262 (191)	17,593 (527)	“	176,667 (0.01)
2004	122,825 (1,290)	“	16,876 (239)	“	153,059 (0.01)
2005	“	“	“	12,631 (335)	160,594 (0.01)
2006	109, 961 (1,520)	“	17,072 (144)	“	147,900 (0.011)
2007	“	“	“	17,574 (843)	152,867 (0.011)
2008	102,674 (1,084)	6,741 (80)	18,160 (288)	“	145,149 (0.009)
2010	94,502 (1,259)	“	17,973 (323)	“	136,790 (0.011)
2011	“	“	“	22,905 (921.5)	142,121 (0.011)
2012	96,828 (1,260)	“	16,184 (155)	“	142,658 (0.011)
2014	91,737 (769)	5,250 (293)	18,937 (308)	“	138,829 (0.009)
2015	“	“	“	27,750 (228)	143,674 (0.006)
2016	80,641 (717)	“	20,490 (460)	“	134,131 (0.007)
2018	75,719 (1,008)	“	21,625 (345)	“	130,344 (0.009)

Minimum Population Estimate

A CV(N) that incorporates the variance of the correction factor is not available. Consistent with a recommendation of the Alaska Scientific Review Group (SRG) in October 1997 (DeMaster 1998) and recommendations contained in Wade and Angliss (1997), a default CV(N) of 0.2 is used in the calculation of the minimum population estimate (N_{MIN}) for this stock. N_{MIN} is calculated using Equation 1 from the potential biological removal (PBR) guidelines (NMFS 2016): $N_{MIN} = N/\exp(0.842\times[\ln(1+[CV(N)]^2)]^{½})$. Using the population estimate (N) of 608,143 and the default CV (0.2), N_{MIN} for the Eastern Pacific stock is 514,738 northern fur seals.

Current Population Trend

Estimates of the size of the Alaska population of northern fur seals increased to approximately 1.25 million in 1974. The population began to decrease in the mid-1970s, with pup production declining at a rate of 6.5-7.8% per year into the 1980s (York 1987). By 1983, the total stock estimate was 877,000 northern fur seals (Briggs and Fowler 1984). Annual pup production on St. Paul Island remained stable between 1981 and 1996 (Fig. 2; York and Fowler 1992). There has been a decline in pup production on St. Paul Island since the mid-1990s. Pup production at St. George Island had a less pronounced period of stabilization, beginning in the late-1980s, that was similarly followed by a decline. However, pup production stabilized again on St. George Island beginning around 2002 (Fig. 3). From 1998 to 2018, pup production declined 4.09% per year (SE = 0.34%; $P < 0.01$) on St. Paul Island and showed no significant trend (SE = 0.58%; $P = 0.59$) on St. George Island. The estimated pup production in 2018 was below the 1919 level (Bower 1920) on both St. Paul and St. George Islands. Northern fur seal pup production at Bogoslof Island has grown at an exponential rate since the 1990s (Towell and Ream 2012) (Fig. 4). Despite continued growth at Bogoslof Island, recent estimates of pup production indicate that the rate of increase may be slowing. Between 1997 and 2015, pup production at Bogoslof Island increased 10.1% per year. Temporary increases in the overall stock size are observed when opportunistic estimates are conducted at Bogoslof, but declines at the larger Pribilof colony (specifically St. Paul) continue to drive the overall stock estimate down over time. The current trend in pup production was fit using agTrend (Johnson and Fritz 2014). Estimated pup production for the Eastern Pacific stock has been declining 1.93% (95% CI: -2.67 to -1.24) per year from 1998 to 2018 (Fig. 5).

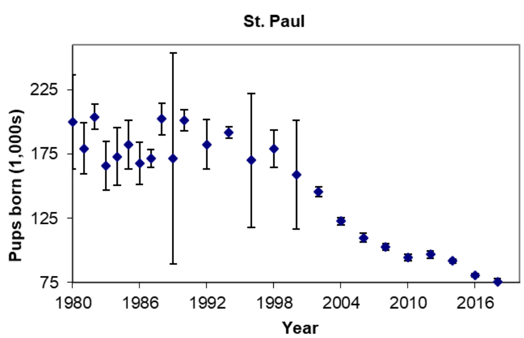

Figure 2. Estimated number of northern fur seal pups born on St. Paul Island, 1980-2018.

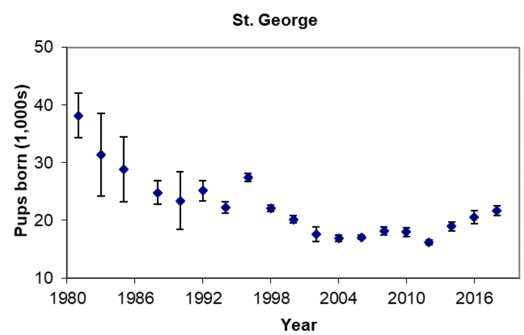

Figure 3. Estimated number of northern fur seal pups born on St. George Island, 1980-2018.

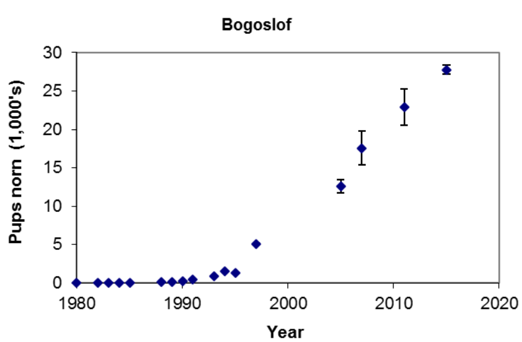

Figure 4. Estimated number of northern fur seal pups born on Bogoslof Island, 1980-2015.

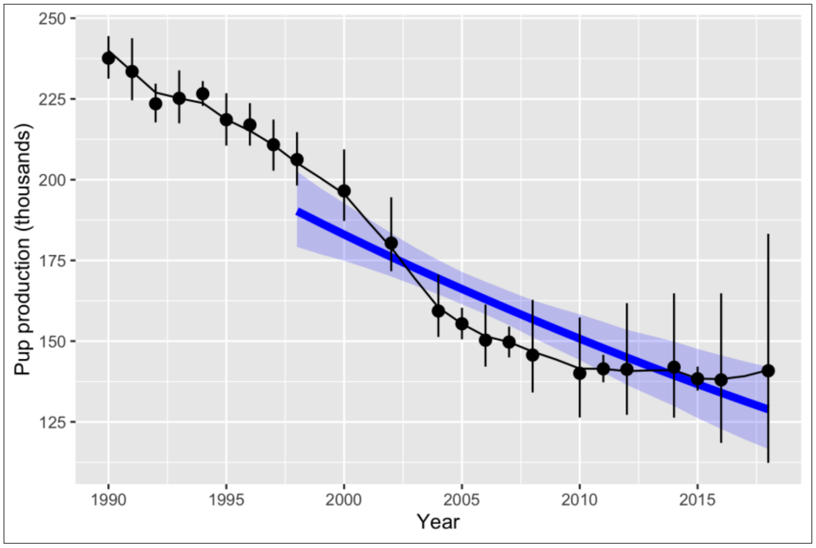

Figure 5. Estimated pup production for the Eastern Pacific stock, 1990-2018, from agTrend (dots), 95% credible interval (bars), agTrend temporal interpolation fit (black line), 1998-2018 average decline (blue line), and 95% credible interval for the fitted average decline in each year (light blue shading).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Pelagic sealing led to a decrease in the fur seal population; however, a moratorium on fur seal harvesting and termination of pelagic sealing resulted in a steady increase in the northern fur seal population from 1912 to 1924. During this period, the rate of population growth was approximately 8.6% (SE = 1.47) per year (A. York, NMFS-AFSC-MML (retired), unpubl. data), the maximum recorded for this species. This growth rate is similar and slightly higher than the 8.1% rate of increase (approximate SE = 1.29) estimated by Gerrodette et al. (1985). Though not as high as growth rates estimated for other fur seal species, the 8.6% rate of increase is considered a reliable estimate of the maximum net productivity rate (R_{MAX}) given the extremely low density of the population in the early 1900s.

POTENTIAL BIOLOGICAL REMOVAL

PBR is defined as the product of the minimum population estimate, one-half the maximum estimated net productivity rate, and a recovery factor: PBR = $N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 0.5, the value for depleted stocks under the Marine Mammal Protection Act (MMPA) (NMFS 2016). Thus, for the Eastern Pacific stock, PBR is 11,067 northern fur seals (514,738 × 0.043 × 0.5).

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2014 and 2018 is listed, by marine mammal stock, in Young et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports. The minimum estimated mean annual level of human-caused mortality and serious injury for the Eastern Pacific stock between 2014 and 2018 is 387 northern fur seals: 3.4 in U.S. commercial fisheries, 2 in unknown (commercial, recreational, or subsistence) fisheries, 7.8 in marine debris, 0.6 due to other causes (car strike, dog attack, oil/tar), and 373 in the Alaska Native subsistence harvest. These mortality and serious injury data do not reflect the total potential threat of entanglement, since additional northern fur seals initially considered seriously injured due to entanglement in fishing gear or marine debris were disentangled and released with non-serious injuries between 2014 and 2018 (see details in the text and in Young et al. 2020). Assignment of mortality and serious injury to both the Eastern Pacific and California stocks of northern fur seals, when events occur in the area and time of year where the two stocks overlap (off the U.S. west coast in December through May), may result in overestimating stock specific mortality and serious injury. Additional potential threats most likely to result in direct human-caused mortality or serious injury of this stock include the increased potential for oil spills due to an increase in vessel traffic in Alaska waters (with changes in sea-ice coverage).

Fisheries Information

Information for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is available in Appendix 3 of the Alaska Stock Assessment Reports (observer coverage) and in the NMFS List of Fisheries (LOF) and the fact sheets linked to fishery names in the LOF (observer coverage and reported incidental takes of marine mammals: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-list-fisheries, accessed December 2020).

Between 2014 and 2018, incidental mortality and serious injury of northern fur seals was observed in one of the federally-managed U.S. commercial fisheries in Alaska monitored for incidental mortality and serious injury by fisheries observers: the Bering Sea/Aleutian Islands flatfish trawl fishery (Table 2; Breiwick 2013; MML, unpubl. data). The minimum estimated mean annual mortality and serious injury rate in this fishery between 2014 and 2018 is 0.8 northern fur seals.

Observer programs for Alaska State-managed commercial fisheries have not documented any mortality or serious injury of northern fur seals.

Table 2. Summary of incidental mortality and serious injury of Eastern Pacific northern fur seals due to U.S. commercial fisheries between 2014 and 2018 and calculation of the mean annual mortality and serious injury rate (Breiwick 2013; MML, unpubl. data). Methods for calculating percent observer coverage are described in Appendix 3 of the Alaska Stock Assessment Reports.

Fishery name	**Years**	**Data type**	**Percent observer coverage**	**Observed mortality**	**Estimated mortality (CV)**	**Mean estimated annual mortality**
Bering Sea/Aleutian Is. flatfish trawl	2014 2015 2016 2017 2018	obs data	100 100 99 100 100	1 0 0 1 2	1 (0.04) 0 0 1 (0.03) 2 (0.03)	0.8 (CV = 0.02)
Minimum total estimated annual mortality						0.8 (CV = 0.02)

Entanglements of northern fur seals have been observed on St. Paul, St. George, and Bogoslof Islands. Since 2011, there has been an increased effort to include entanglement reports in the NMFS Alaska Region stranding database. A summary of entanglements in fishing gear reported between 2014 and 2018 is provided in Table 3 (Young et al. 2020). These mortality and serious injury estimates result from an actual count of verified human-caused deaths and serious injuries and are minimums because not all entangled animals strand nor are all stranded animals found, reported, or have the cause of death determined. Three northern fur seals entangled in commercial Bering Sea/Aleutian Islands halibut longline gear and six northern fur seals entangled in commercial Bering Sea/Aleutian Islands trawl gear were reported to the NMFS Alaska Region marine mammal stranding network between 2014 and 2018, resulting in minimum mean annual mortality and serious injury rates of 0.6 and 1.2 northern fur seals, respectively, in these fisheries (Table 3; Young et al. 2020).

A total of seven northern fur seals initially considered to be seriously injured due to entanglement in commercial Bering Sea/Aleutian Islands trawl gear (one in 2014), Bering Sea/Aleutian Islands trawl gear (one in 2015), unidentified trawl gear (three in 2016), and unidentified net (one each in 2016 and 2017) were disentangled and released with non-serious injuries (Young et al. 2020), therefore, they were not included in the mean annual mortality and serious injury rate for 2014 to 2018.

The total mean annual mortality and serious injury rate incidental to U.S. commercial fisheries between 2014 and 2018 is 3.4 northern fur seals (0.8 from observer data + 2.6 from stranding data).

The minimum mean annual mortality and serious injury rate due to entanglements in gillnet (0.4), unidentified fishing gear (0.2), and unidentified fishing net (0.2) in Alaska waters between 2014 and 2018 totaled 0.8 northern fur seals (Table 3; Young et al. 2020). These entanglements cannot be assigned to a specific fishery, and it is unknown whether commercial, recreational, or subsistence fisheries are the source of the fishing debris.

The Eastern Pacific northern fur seal stock can occur off the west coast of the continental U.S. in winter/spring; therefore, any mortality or serious injury of northern fur seals reported off the coasts of Washington, Oregon, or California during December through May is assigned to both the Eastern Pacific and California stocks of northern fur seals (as noted in Table 3). Reports to the NMFS West Coast Region marine mammal stranding network between 2014 and 2018 resulted in minimum mean annual mortality and serious injury rates of one northern fur seal entangled in trawl gear and 0.2 entangled in unidentified fishing net from unknown (commercial, recreational, or subsistence) fisheries off the U.S. west coast in December through May (Table 3; Young et al. 2020). These mortality and serious injury estimates result from an actual count of verified human-caused deaths and serious injuries and are minimums because not all entangled animals strand nor are all stranded animals found, reported, or have the cause of death determined.

Table 3. Summary of mortality and serious injury of Eastern Pacific northern fur seals, by year and type, reported to the NMFS Alaska Region and NMFS West Coast Region marine mammal stranding networks between 2014 and 2018 (Young et al. 2020). Animals that were disentangled and released with non-serious injuries have been excluded from this table.

Cause of injury	**2014**	**2015**	**2016**	**2017**	**2018**	**Mean annual mortality**
Entangled in commercial Bering Sea/Aleutian Is. halibut longline gear	3	0	0	0	0	0.6
Entangled in commercial Bering Sea/Aleutian Is. trawl gear	6	1	1	1	1	2
Entangled in Bering Sea/Aleutian Is. gillnet gear*	0	1	0	0	0	0.2
Entangled in Bering Sea/Aleutian Is. unidentified fishing gear*	0	1	0	0	0	0.2
Entangled in gillnet*	1	0	0	0	0	0.2
Entangled in unidentified net*	1 + 1[a]	0	0	0	0	0.2 + 0.2[a]
Entangled in trawl gear*	2[a]	0	0	3[a]	0	1[a]
Entangled in marine debris	11	0	9	13	6	7.8
Struck by car	0	1	0	0	0	0.2
Dog attack	0	0	1[a]	0	0	0.2[a]
Oil/tar	1[a]	0	0	0	0	0.2[a]
Total in commercial fisheries						2.6
*Total in unknown (commercial, recreational, or subsistence) fisheries						2
Total in marine debris						7.8
Total due to other causes (car strike, dog attack, oil/tar)						0.6

[a]The mortality or serious injury occurred off the coast of Washington, Oregon, or California in December through May and was assigned to both the Eastern Pacific and California stocks of northern fur seals.

Alaska Native Subsistence/Harvest Information

NMFS signed agreements with the Tribal Government of St. Paul Island (2000) and the Traditional Council of St. George Island (2001) to co-manage Steller sea lions and northern fur seals. These co-management agreements promote full and equal participation by Alaska Natives in decisions affecting the subsistence management of northern fur seals (to the maximum extent allowed by law) as a tool for conserving northern fur seal populations in Alaska (https://www.fisheries.noaa.gov/alaska/marine-mammal-protection/co-management-marine-mammals-alaska, accessed December 2020). Alaska Natives residing on the Pribilof Islands are allowed an annual subsistence harvest of northern fur seals, with a 3-year take range based on historical local needs. Typically, only juvenile males are taken in the subsistence harvest, which results in a much smaller impact on population growth than a harvest that includes females. However, accidental harvesting of females does occur. The accidental harvest of female northern fur seals between 2014 and 2018 included four females on St. Paul Island (Melovidov et al. 2014) and one on St. George Island (Kashevarof 2014) in 2014, two on St. Paul in 2015 (Lestenkof et al. 2015), and one on St. Paul in 2016 (Melovidov et al. 2017). The harvest of northern fur seal pups on St. George Island between 2014 and 2018, beginning with the inaugural pup harvest in 2014, included 54 pups in 2014 (Testa 2016), 57 in 2015 (Meyer 2016), 46 in 2016 (Meyer 2017), 51 in 2017 (Meyer 2018), and 26 in 2018 (Meyer 2019). Between 2014 and 2018, the average annual subsistence harvest of northern fur seals on the Pribilof Islands was 373 fur seals (Table 4).

Table 4. Summary of the Alaska Native subsistence harvest of northern fur seals on St. Paul and St. George Islands between 2014 and 2018.

Year	St. Paul	St. George	Total harvested
2014	266[a]	158[b, c]	424
2015	314[d]	118[e, f]	432
2016	309[g]	83[h, i]	392
2017	217[j]	89[k, l]	306
2018	225[m]	88[n,o]	313
Mean annual harvest			373

[a]Melovidov et al. (2014); [b]Kashevarof (2014); [c]Testa (2016); [d]Lestenkof et al. (2015); [e]Kashevarof (2016), [f]Meyer (2016); [g]Melovidov et al. (2017); [h]Testa (2018); [i]Meyer (2017); [j]NMFS, unpubl. data; [k]Lekanof (2017); [l]Meyer (2018); [m]Lestenkof et al. (2019), [n]Malavansky (2019); [o]Meyer (2019).

Other Mortality

Intentional killing of northern fur seals by commercial fishermen, sport fishermen, and others may occur, but the magnitude of that mortality is unknown.

Because the Eastern Pacific and California stocks of northern fur seals overlap off the west coast of the continental U.S. during December through May, non-fishery mortality and serious injury reported off the coast of Washington, Oregon, or California during that time is assigned to both stocks (see details in Table 3). Reports to the NMFS Alaska Region and West Coast Region stranding networks between 2014 and 2018 resulted in mean annual mortality and serious injury rates of 7.8 northern fur seals due to entanglement in marine debris in Alaska waters, 0.2 due to a car strike on St. Paul Island, and 0.2 each due to a dog attack and oil/tar in California (Table 3; Young et al. 2020). These mortality and serious injury estimates result from an actual count of verified human-caused deaths and serious injuries and are minimums because not all entangled animals strand nor are all stranded animals found, reported, or have the cause of death determined

An additional 29 northern fur seals that were initially considered seriously injured due to entanglement in marine debris (four in 2014, six in 2015, six in 2016, four in 2017, and 9 in 2018) were disentangled and released with non-serious injuries (Young et al. 2020); therefore, these animals were not included in the mean annual mortality and serious injury rate for 2014 to 2018.

STATUS OF STOCK

Based on currently available data, the minimum estimate of the mean annual U.S. commercial fishery-related mortality and serious injury rate for this stock (3.4 northern fur seals) is less than 10% of the calculated PBR (10% of PBR = 1,107 northern fur seals) and, therefore, can be considered insignificant and approaching a zero mortality and serious injury rate. The minimum estimated mean annual level of human-caused mortality and serious injury (387 northern fur seals) does not exceed the PBR (11,067) for this stock. The PBR calculation assumes mortality is evenly distributed across males, females, and each age class; but that is not the case with the subsistence harvest, which accounts for most of the known direct human-caused mortality. The subsistence harvest is almost entirely sub-adult males and male pups and, therefore, has a relatively low impact on the population due to the disproportionate importance of females to the population. Thus, non-breeding male-biased mortality up to the maximum levels authorized for subsistence use does not represent a significant risk to the Eastern Pacific northern fur seal stock. The northern fur seal was designated as depleted under the MMPA in 1988 because population levels had declined to less than 50% of levels observed in the late 1950s (1.8 million animals; 53 FR 17888, 18 May 1988). The Eastern Pacific stock of northern fur seals is classified as a strategic stock because it is designated as depleted under the MMPA.

There are key uncertainties in the assessment of the Eastern Pacific stock of northern fur seals. The abundance estimate is based on pup counts multiplied by a constant; this constant was based on northern fur seal demographic information which is now quite dated and it is unknown whether the constant is still optimum for this population. Because an estimate of variance cannot be determined, the N_{MIN} calculation uses a default CV of 0.2. At this time, the cause of the decline of this stock is unknown. Estimates of human-caused mortality and serious injury from stranding data are underestimates because not all animals strand nor are all stranded animals found, reported, or have the cause of death determined.

HABITAT CONCERNS

A number of natural and human-related factors have been suggested as contributing to the continued decline in abundance of the Eastern Pacific stock of northern fur seals, including environmental perturbation, disease, predation, contaminants, indirect effects of commercial fishing, incidental take, poaching, and the effects of human presence and development at or near fur seal rookeries (NMFS 2007). The concentration of fur seals on the breeding islands and in the surrounding waters of the Bering Sea during summer, and their broad pelagic distribution across the North Pacific Ocean over the winter, complicates the understanding of these factors and the ability to implement effective management strategies. However, the population trends at the Pribilof Islands are of significant concern, with declines in stock abundance continuing to be driven by the declines on St. Paul Island rookeries; pup production at St. George Island has stabilized (Figs. 2 and 3). The Pribilof Island communities, particularly St. Paul, have developed a fishery-based economy since the cessation of the commercial fur harvest in 1985. Harbor development and expansion from 1985 to present, and the economic growth resulting from the now well-established fisheries, has increased the potential exposure of fur seals to construction activities, vessel and vehicle traffic, seafood and municipal waste discharge, and human presence. Management measures are in place to help ameliorate some of these threats around the fur seal breeding and resting sites (e.g., regulatory closures that prohibit unauthorized human access beyond posted fur seal breeding and resting sites from 1 June to 15 October each year, establishment of Aircraft Advisory Zones and Requested Aircraft Flight Paths, and new subsistence use regulations).

Northern fur seals from each island, and even from central breeding areas within each island, may also experience dissimilar exposure to varying environmental and foraging conditions across the Bering Sea; northern fur seals from different central breeding areas consistently use different foraging habitat (Robson et al. 2004, Sterling and Ream 2004, Call et al. 2008, Kuhn et al. 2014). Climate change could alter the abundance, distribution, and makeup of available prey for northern fur seals in the Bering Sea as a result of reduced sea ice and warming temperatures. These changes could differentially impact the survival and reproduction of individuals and breeding aggregations on the three islands; however, the exact mechanisms are unknown and there are no clear management actions that could be taken to address the impacts on northern fur seals.

Commercial fisheries target fur seal prey and prey that compete with fur seals in both the Bering Sea and the North Pacific Ocean. Northern fur seals predominantly prey on walleye pollock over the Bering Sea shelf, and progressively greater proportions of oceanic fish and squid are consumed when they forage over the slope and in off-shelf waters (Zeppelin and Ream 2006). Comparison of ingested prey sizes based on scat and spew analysis indicates an overlap between sizes of pollock consumed by Pribilof Island northern fur seals and those caught by the commercial trawl fishery, suggesting possible competition between fur seals and commercial fisheries for pollock (Gudmundson et al. 2006). In contrast to northern fur seals from the Pribilof Islands, Bogoslof Island northern fur seals forage in the deeper water of the Bering Sea Basin and their diet is comprised primarily of off-shelf species (northern smoothtongue, squid, myctophids) as well as juvenile walleye pollock (Zeppelin and Orr 2010, Kuhn et al. 2014). Our understanding of the consequences of commercial fisheries removals on northern fur seal survival and productivity is highly uncertain.

CITATIONS

Baker, J. D., G. A. Antonelis, C. W. Fowler, and A. E. York. 1995. Natal site fidelity in northern fur seals, *Callorhinus ursinus*. Anim. Behav. 50(1):237-247.

Bower, W. T. 1920. Alaska fisheries and fur industries in 1919. U.S. Dep. Commer., Appendix IX to Report of U.S. Commissioner of Fisheries for 1919. Bureau of Fisheries Document No. 891. Washington Government Printing Office. 160 p.

Breiwick, J. M. 2013. North Pacific marine mammal bycatch estimation methodology and results, 2007-2011. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-260, 40 p.

Briggs, L., and C. W. Fowler. 1984. Table and figures of the basic population data for northern fur seals of the Pribilof Islands. *In* Background papers submitted by the United States to the 27th annual meeting of the Standing Scientific Committee of the North Pacific Fur Seal Commission, March 29-April 9, 1984, Moscow, U.S.S.R. Available from Marine Mammal Laboratory, AFSC, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Call, K. A., R. R. Ream, D. Johnson, J. T. Sterling, and R. G. Towell. 2008. Foraging route tactics and site fidelity of adult female northern fur seal (*Callorhinus ursinus*) around the Pribilof Islands. Deep-Sea Res. II 55:1883-1896.

Chapman, D. G., and A. M. Johnson. 1968. Estimation of fur seal pup populations by randomized sampling. Trans. Am. Fish. Soc. 97:264-270.

DeLong, R. L. 1982. Population biology of northern fur seals at San Miguel Island, California. Ph.D. Dissertation, University of California, Berkeley, CA. 185 p.

DeLong, R. L., and G. A. Antonelis. 1991. Impacts of the 1982-1983 El Niño on the northern fur seal population at San Miguel Island, California, p. 75-83. *In* F. Trillmich and K. Ono (eds.), Pinnipeds and El Niño: Responses to Environmental Stress. University of California Press, Berkeley, CA.

DeMaster, D. P. 1998. Minutes from the sixth meeting of the Alaska Scientific Review Group, 21-23 October 1997, Seattle, Washington. 40 p. Available from Alaska Fisheries Science Center, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Dickerson B. R., R. R. Ream, S. N. Vignieri, and P. Bentzen. 2010. Population structure as revealed by mtDNA and microsatellites in northern fur seals, *Callorhinus ursinus*, throughout their range. PLoS ONE 5(5):e10671. DOI: dx.doi.org/10.1371/journal.pone.0010671 .

Dizon, A. E., C. Lockyer, W. F. Perrin, D. P. DeMaster, and J. Sisson. 1992. Rethinking the stock concept: a phylogeographic approach. Conserv. Biol. 6:24-36.

Fiscus, C. F. 1983. Fur seals. *In* Background papers submitted by the United States to the 26th annual meeting of the Standing Scientific Committee of the North Pacific Fur Seal Commission, Washington, DC, 28 March-5 April, 1983. Available from Marine Mammal Laboratory, AFSC, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Gerrodette, T., D. Goodman, and J. Barlow. 1985. Confidence limits for population projections when vital rates vary randomly. Fish. Bull., U.S. 83:207-217.

Gudmundson, C. J., T. K. Zeppelin, and R. R. Ream. 2006. Application of two methods for determining diet of northern fur seals (*Callorhinus ursinus*). Fish. Bull., U.S. 104:445-455.

Johnson, D. S., and L. Fritz. 2014. agTrend: a Bayesian approach for estimating trends of aggregated abundance. Methods Ecol. Evol. 5:1110-1115. DOI: dx.doi.org/10.1111/2041-210X.12231 .

Kajimura, H. 1984. Opportunistic feeding of the northern fur seal, *Callorhinus ursinus*, in the eastern North Pacific Ocean and eastern Bering Sea. U.S. Dep. Commer., NOAA Tech. Rep. NMFS SSRF-779, 49 p.

Kashevarof, H. 2014. Northern fur seal harvests, St. George Island, AK: harvest report for the 2014 season 7.7.2014-8.8.2014. Aleut Community of St. George Island, St. George Traditional Council, Kayumixtax Eco-Office, St. George Island, Pribilof Islands, AK. 3 p.

Kashevarof, H. 2016. Northern fur seal harvests, St. George Island, AK: harvest report for the 2015 season 7.7.2015-8.7.2015. Aleut Community of St. George Island, St. George Traditional Council, Kayumixtax Eco-Office, St. George Island, Pribilof Islands, AK.

Kenyon K. W., and F. Wilke. 1953. Migration of the northern fur seal, *Callorhinus ursinus*. J. Mammal. 34(1):86-98.

Kuhn, C. E., R. R. Ream, J. T. Sterling, J. R. Thomason, and R. G. Towell. 2014. Spatial segregation and the influence of habitat on the foraging behavior of northern fur seals (*Callorhinus ursinus*). Can. J. Zool. 92:861-873.

Lander, R. H. 1981. A life table and biomass estimate for Alaskan fur seals. Fish. Res. (Amst.) 1:55-70.

Lander, R. H., and H. Kajimura. 1982. Status of northern fur seals. FAO Fisheries Series 5:319-345.

Lekanof, D. 2017. Northern fur seal harvests, St. George Island, AK: harvest report for the 2017 season 07/10/2017-08/08/2017. Aleut Community of St. George Island, St. George Traditional Council, Kayumixtax Eco-Office, St. George Island, Pribilof Islands, AK.

Lestenkof, P. M., P. I. Melovidov, and A. P. Lestenkof. 2015. The subsistence harvest of subadult northern fur seals on St. Paul Island, Alaska in 2015. Aleut Community of St. Paul Island, Tribal Government, Ecosystem Conservation Office, St. Paul Island, Pribilof Islands, AK. 16 p.

Lestenkof, P. M., L. M. Divine, P. I. Melovidov, A. P. Lestenkof, V. M. Padula, and K. M. Melovidov. 2019. The subsistence harvest of subadult laaqudan (northern fur seals) on St. Paul Island, Alaska in 2018. Aleut Community of St. Paul Island, Tribal Government, Ecosystem Conservation Office, St. Paul Island, Pribilof Islands, AK. 16 p.

Loughlin, T. R., W. J. Ingraham, Jr., N. Baba, and B. W. Robson. 1999. Use of a surface-current model and satellite telemetry to assess marine mammal movements in the Bering Sea. University of Alaska Sea Grant Press, AK-SG-99-03, Fairbanks, AK.

Malavansky, A. 2019. Northern fur seal harvests, St. George Island, AK: harvest report for the 2018 season 06/21/2018-08/08/2018. Aleut Community of St. George Island, St. George Traditional Council, Kayumixtax Eco-Office, St. George Island, Pribilof Islands, AK.

Melovidov, P. I., P. M. Lestenkof, M. Rukovishnikoff, Sr., and D. V. V. Roberts. 2014. The subsistence harvest of subadult northern fur seals on St. Paul Island, Alaska in 2014. Aleut Community of St. Paul Island, Tribal Government, Ecosystem Conservation Office, St. Paul Island, Pribilof Islands, AK. 16 p.

Melovidov, P. I., P. M. Lestenkof, A. P. Lestenkof, L. M. Divine, and R. M. Rukovishnikoff. 2017. The subsistence harvest of subadult northern fur seals on St. Paul Island, Alaska in 2016. Aleut Community of St. Paul Island, Tribal Government, Ecosystem Conservation Office, St. Paul Island, Pribilof Islands, AK. 14 p. + appendices.

Meyer, B. 2016. Harvest monitoring services, subsistence harvest of northern fur seals on St. George Island, AK: harvest report for the 2015 season September 15 to November 30, 2015. Aleut Community of St. George Island, St. George Traditional Council, Kayumixtax Eco-Office, St. George Island, Pribilof Islands, AK.

Meyer, B. 2017. Harvest monitoring services, subsistence harvest of northern fur seals on St. George Island, AK: harvest report for the 2016 season September 16 to November 30, 2016. Aleut Community of St. George Island, St. George Traditional Council, Kayumixtax Eco-Office, St. George Island, Pribilof Islands, AK.

Meyer, B. 2018. Harvest monitoring services, subsistence harvest of northern fur seals on St. George Island, AK: harvest report for the 2017 season September 15 to November 30, 2017. Aleut Community of St. George Island, St. George Traditional Council, Kayumixtax Eco-Office, St. George Island, Pribilof Islands, AK. 5 p.

Meyer, B. 2019. Young of the year subsistence harvest of northern fur seals on St. George Island, AK: harvest report for the 2018 season September 15 to November 30. Aleut Community of St. George Island, St. George Traditional Council, Kayumixtax Eco-Office, St. George Island, Pribilof Islands, AK. 8 p.

National Marine Fisheries Service (NMFS). 1993. Final conservation plan for the northern fur seal (*Callorhinus ursinus*). Prepared by the National Marine Mammal Laboratory, Alaska Fisheries Science Center, Seattle, WA, and the Office of Protected Resources, National Marine Fisheries Service, Silver Spring, MD. 80 p.

National Marine Fisheries Service (NMFS). 2007. Conservation plan for the Eastern Pacific stock of northern fur seal (*Callorhinus ursinus*). National Marine Fisheries Service, Alaska Regional Office, Juneau, AK.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2020.

Ream, R. R. 2002. Molecular ecology of northern otariids: genetic assessment of northern fur seal and Steller sea lion distributions. Ph.D. Dissertation, University of Washington, Seattle, WA. 134 p.

Ream, R. R., J. T. Sterling, and T. R. Loughlin. 2005. Oceanographic features related to northern fur seal migratory movements. Deep-Sea Res. II 52:823-843.

Robson, B. R., M. E. Goebel, J. D. Baker, R. R. Ream, T. R. Loughlin, R. C. Francis, G. A. Antonelis, and D. P. Costa. 2004. Separation of foraging habitat among breeding sites of a colonial marine predator, the northern fur seal (*Callorhinus ursinus*). Can. J. Zool. 82:20-29.

Roppel, A. Y. 1984. Management of northern fur seals on the Pribilof Islands, Alaska, 1786-1981. U.S. Dep. Commer., NOAA Tech. Rep. NMFS-4, 32 p.

Sterling, J. T., and R. R. Ream. 2004. At-sea behavior of juvenile male northern fur seals (*Callorhinus ursinus*). Can. J. Zool. 82:1621-1637.

Testa, J. W. (ed.). 2016. Fur seal investigations, 2013-2014. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-316, 124 p.

Testa, J. W. (ed.). 2018. Fur seal investigations, 2015-2016. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-375, 107 p.

Towell, R., and R. Ream. 2012. 2011 northern fur seal pup production estimate on Bogoslof Island, Alaska. Available from Marine Mammal Laboratory, AFSC, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Towell, R. G., R. R. Ream, and A. E. York. 2006. Decline in northern fur seal (*Callorhinus ursinus*) pup production on the Pribilof Islands. Mar. Mammal Sci. 22(2):486-491.

Wade, P. R., and R. Angliss. 1997. Guidelines for assessing marine mammal stocks: report of the GAMMS Workshop April 3-5, 1996, Seattle, Washington. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-OPR-12, 93 p.

York, A. E. 1987. Northern fur seal, *Callorhinus ursinus*, eastern Pacific population (Pribilof Islands, Alaska, and San Miguel Island, California), p. 9-21. *In* J. P. Croxall and R. L. Gentry (eds.), Status, biology, and ecology of fur seals. Proceedings of an international symposium and workshop, Cambridge, England, 23-27 April 1984. U.S. Dep. Commer., NOAA Tech. Rep. NMFS-51.

York, A. E., and C. W. Fowler. 1992. Population assessment, Pribilof Islands, Alaska, p. 9-26. *In* H. Kajimura and E. Sinclair (eds.), Fur seal investigations, 1990. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-2.

York, A. E., and P. Kozloff. 1987. On estimating the number of fur seal pups born on St. Paul Island, 1980-86. Fish. Bull., U.S. 85:367-375.

Young, N. C., B. J. Delean, V. T. Helker, J. C. Freed, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2014-2018. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-413, 142 p.

Zeppelin, T. K., and A. J. Orr. 2010. Stable isotope and scat analyses indicate diet and habitat partitioning in northern fur seals, *Callorhinus ursinus*, across the eastern Pacific. Mar. Ecol. Prog. Ser. 409:241-253.

Zeppelin, T. K., and R. R. Ream. 2006. Foraging habitats based on the diet of female northern fur seals (*Callorhinus ursinus*) on the Pribilof Islands, Alaska. J. Zool. 270:565-576.

HARBOR SEAL (*Phoca vitulina richardii*)

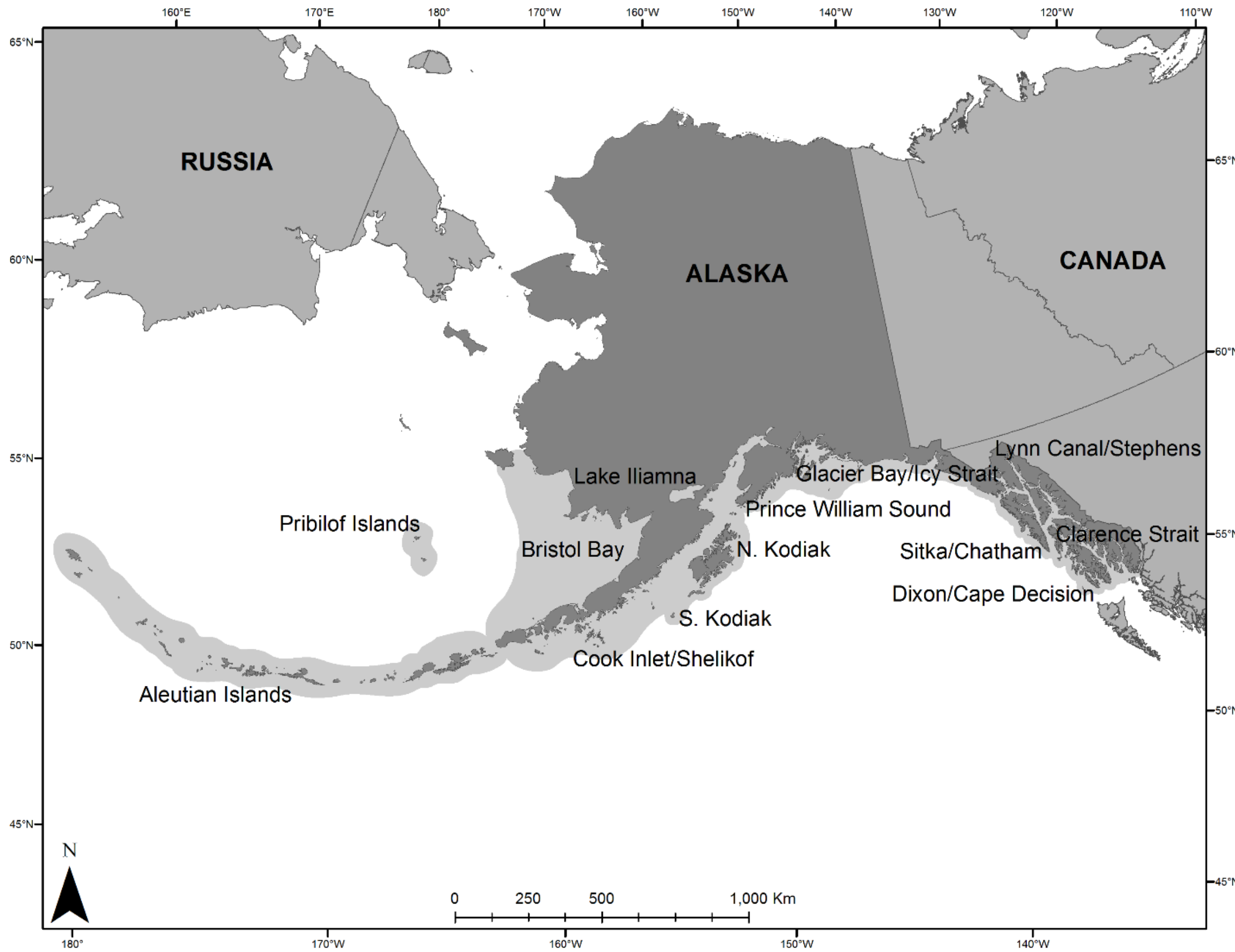

Figure 1. Approximate extent of harbor seals in Alaska waters (shaded coastline area).

STOCK DEFINITION AND GEOGRAPHIC RANGE

Harbor seals inhabit coastal and estuarine waters off Baja California, north along the western coasts of the United States, British Columbia, and Southeast Alaska, west through the Gulf of Alaska and Aleutian Islands, and in the Bering Sea north to Cape Newenham and the Pribilof Islands. They haul out on rocks, reefs, beaches, and drifting glacial ice and feed in marine, estuarine, and occasionally fresh waters. Harbor seals generally are non-migratory, with local movements associated with such factors as tides, weather, season, food availability, and reproduction (Scheffer and Slipp 1944; Fisher 1952; Bigg 1969, 1981; Hastings et al. 2004). The results of past and recent satellite-tagging studies in Southeast Alaska, Prince William Sound, Kodiak Island, and Cook Inlet are also consistent with the conclusion that harbor seals are non-migratory (Swain et al. 1996, Lowry et al. 2001, Small et al. 2003, Boveng et al. 2012). However, some long-distance movements of tagged animals in Alaska have been recorded (Pitcher and McAllister 1981, Lowry et al. 2001, Small et al. 2003, Womble 2012, Womble and Gende 2013). Strong fidelity of individuals for haul-out sites during the breeding season has been documented in several populations (Härkönen and Harding 2001), including some regions in Alaska such as Kodiak Island, Prince William Sound, Glacier Bay/Icy Strait, and Cook Inlet (Pitcher and McAllister 1981, Small et al. 2005, Boveng et al. 2012, Womble 2012, Womble and Gende 2013).

Westlake and O'Corry-Crowe's (2002) analysis of genetic information from 881 samples across 181 sites revealed population subdivisions on a scale of 600-820 km. These results suggest that genetic differences within

Alaska, and most likely over their entire North Pacific range, increase with increasing geographic distance. New information revealed substantial genetic differences indicating that female dispersal occurs at region specific spatial scales of 150-540 km. This research identified 12 demographically independent clusters within the range of Alaska harbor seals; however, significant geographic areas within the Alaska harbor seal range remain unsampled (O'Corry-Crowe et al. 2003).

In 2010, NMFS and their co-management partners, the Alaska Native Harbor Seal Commission, identified 12 separate stocks of harbor seals based largely on genetic structure; this represented a significant increase in the number of harbor seal stocks from the three stocks (Bering Sea, Gulf of Alaska, Southeast Alaska) previously recognized. Given the genetic samples were not obtained continuously throughout the range, a total evidence approach was used to consider additional factors such as population trends, observed harbor seal movements, and traditional Alaska Native use areas in the final designation of stock boundaries. The 12 stocks of harbor seals currently identified in Alaska are 1) the Aleutian Islands stock – occurring along the entire Aleutian chain from Attu Island to Ugamak Island; 2) the Pribilof Islands stock – occurring on Saint Paul and Saint George Islands, as well as on Otter and Walrus Islands; 3) the Bristol Bay stock – ranging from Nunivak Island south to the west coast of Unimak Island and extending inland to Kvichak Bay and Lake Iliamna; 4) the North Kodiak stock – ranging from approximately Middle Cape on the west coast of Kodiak Island northeast to West Amatuli Island and south to Marmot and Spruce Islands; 5) the South Kodiak stock – ranging from Middle Cape on the west coast of Kodiak Island southwest to Chirikof Island and east along the south coast of Kodiak Island to Spruce Island, including the Trinity Islands, Tugidak Island, Sitkinak Island, Sundstrom Island, Aiaktalik Island, Geese Islands, Two Headed Island, Sitkalidak Island, Ugak Island, and Long Island; 6) the Prince William Sound stock – ranging from Elizabeth Island off the southwest tip of the Kenai Peninsula to Cape Fairweather, including Prince William Sound, the Copper River Delta, Icy Bay, and Yakutat Bay; 7) the Cook Inlet/Shelikof Strait stock – ranging from the southwest tip of Unimak Island east along the southern coast of the Alaska Peninsula to Elizabeth Island off the southwest tip of the Kenai Peninsula, including Cook Inlet, Knik Arm, and Turnagain Arm; 8) the Glacier Bay/Icy Strait stock – ranging from Cape Fairweather southeast to Column Point, extending inland to Glacier Bay, Icy Strait, and from Hanus Reef south to Tenakee Inlet; 9) the Lynn Canal/Stephens Passage stock – ranging north along the east and north coast of Admiralty Island from the north end of Kupreanof Island through Lynn Canal, including Taku Inlet, Tracy Arm, and Endicott Arm; 10) the Sitka/Chatham Strait stock – ranging from Cape Bingham south to Cape Ommaney, extending inland to Table Bay on the west side of Kuiu Island and north through Chatham Strait to Cube Point off the west coast of Admiralty Island, and as far east as Cape Bendel on the northeast tip of Kupreanof Island; 11) the Dixon/Cape Decision stock – ranging from Cape Decision on the southeast side of Kuiu Island north to Point Barrie on Kupreanof Island and extending south from Port Protection to Cape Chacon along the west coast of Prince of Wales Island and west to Cape Muzon on Dall Island, including Coronation Island, Forrester Island, and all the islands off the west coast of Prince of Wales Island; and 12) the Clarence Strait stock – ranging along the east coast of Prince of Wales Island from Cape Chacon north through Clarence Strait to Point Baker and along the east coast of Mitkof and Kupreanof Islands north to Bay Point, including Ernest Sound, Behm Canal, and Pearse Canal (Fig. 1). Individual stock distributions can be seen in Figures 2a-l.

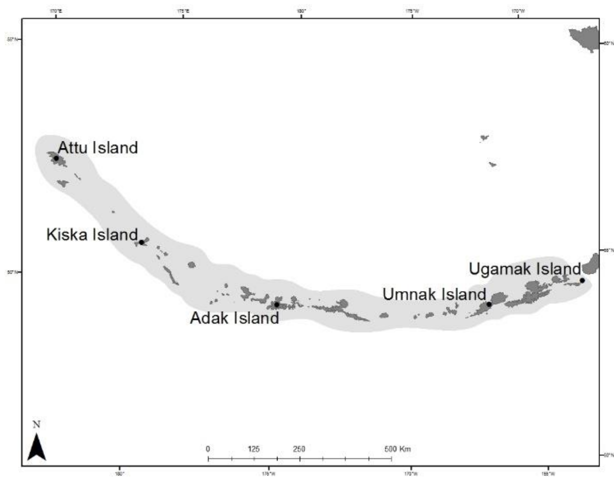

Figure 2a. Approximate extent of Aleutian Islands harbor seal stock (shaded area).

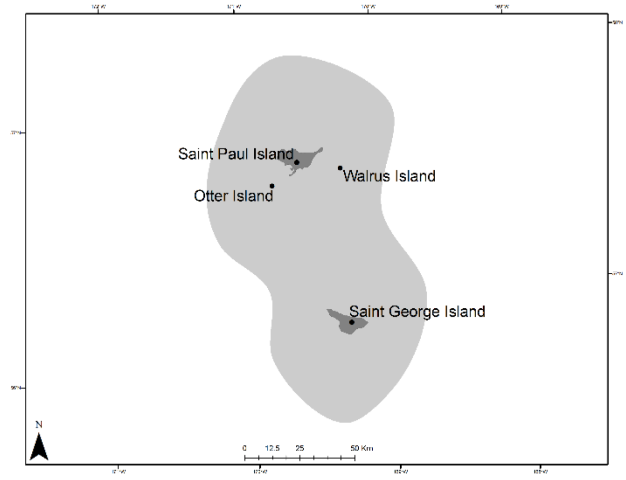

Figure 2b. Approximate extent of Pribilof Islands harbor seal stock (shaded area).

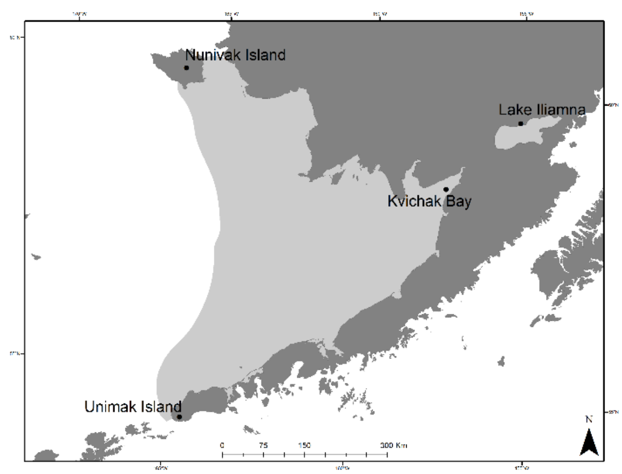

Figure 2c. Approximate extent of Bristol Bay harbor seal stock (shaded area).

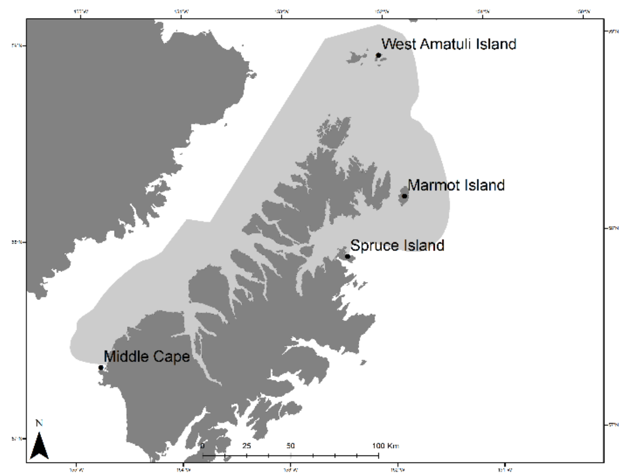

Figure 2d. Approximate extent of North Kodiak harbor seal stock (shaded area).

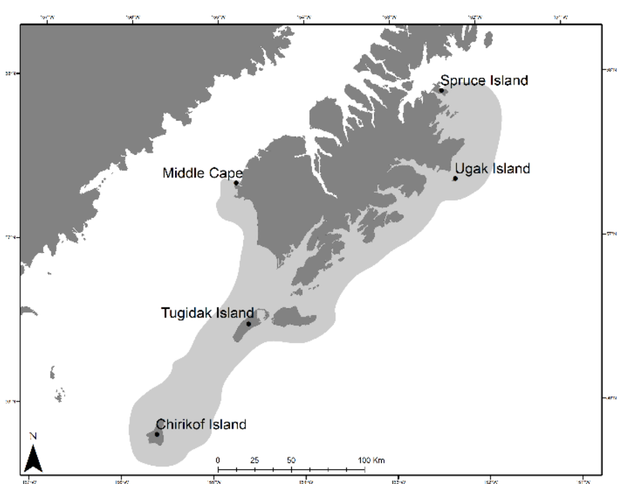

Figure 2e. Approximate extent of South Kodiak harbor seal stock (shaded area).

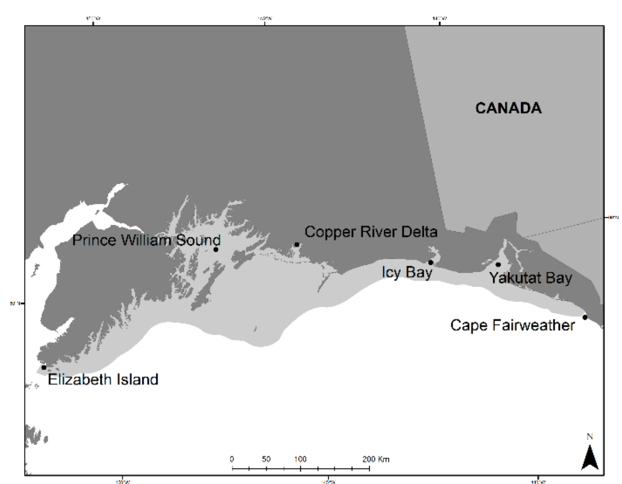

Figure 2f. Approximate extent of Prince William Sound harbor seal stock (shaded area).

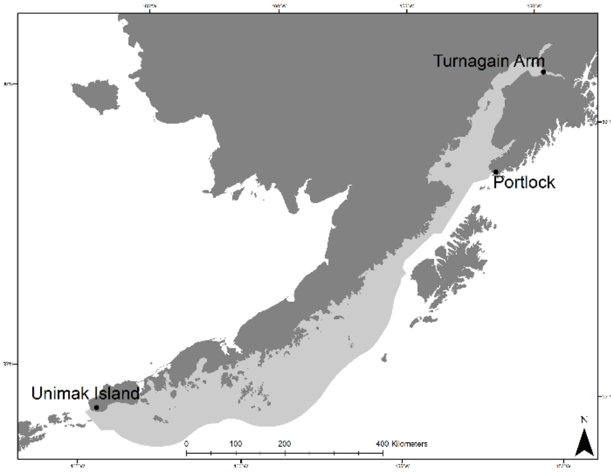

Figure 2g. Approximate extent of Cook Inlet/Shelikof Strait harbor seal stock (shaded area).

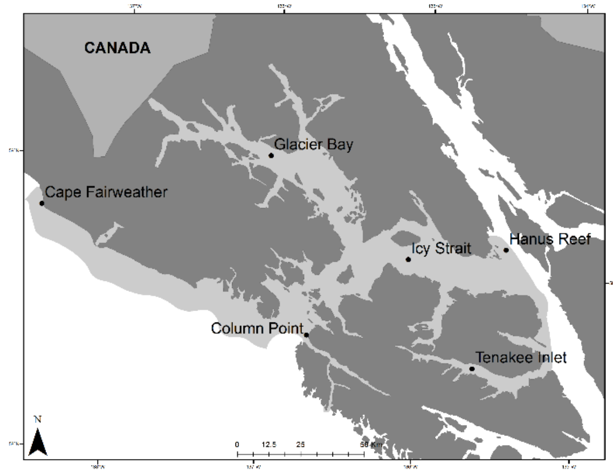

Figure 2h. Approximate extent of Glacier Bay/Icy Strait harbor seal stock (shaded area).

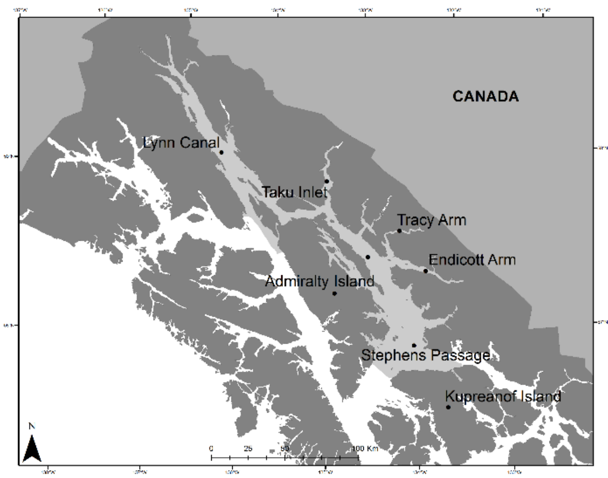

Figure 2i. Approximate extent of Lynn Canal/Stephens Passage harbor seal stock (shaded area).

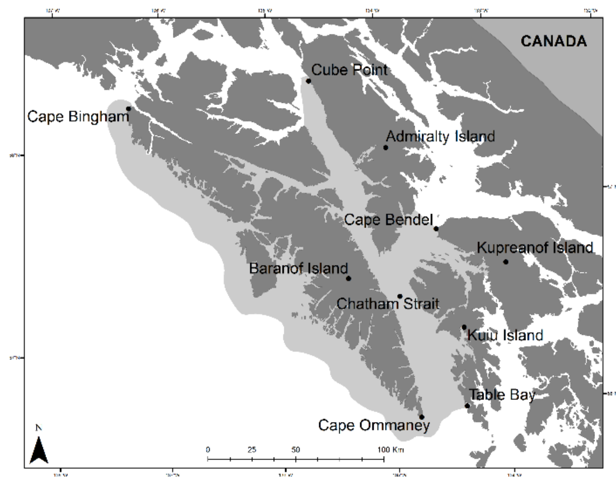

Figure 2j. Approximate extent of Sitka/Chatham Strait harbor seal stock (shaded area).

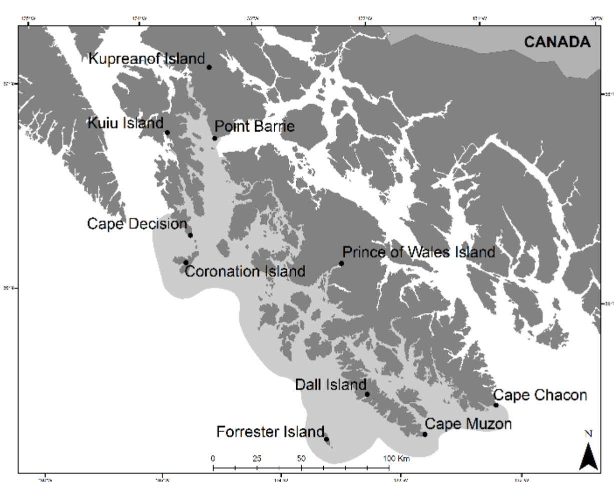

Figure 2k. Approximate extent of Dixon/Cape Decision harbor seal stock (shaded area).

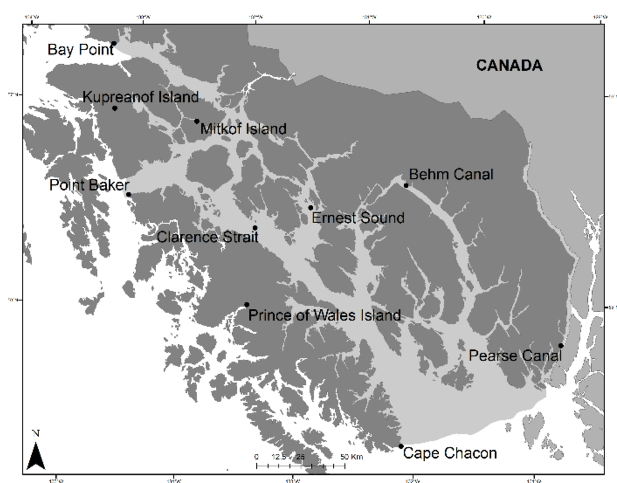

Figure 2l. Approximate extent of Clarence Strait harbor seal stock (shaded area).

POPULATION SIZE

Local or regional trends in harbor seal numbers have been monitored at various time intervals since the 1970s, revealing diverse spatial patterns in apparent population trends. Where declines have been observed, they seem, generally, to have been strongest in the late 1970s or early 1980s to the 1990s. For example, counts of harbor seals declined by about 80% at Tugidak Island in the 1970s and 1980s (Pitcher 1990), and numbers at Nanvak Bay in northern Bristol Bay also declined at about the same time (Jemison et al. 2006). In Prince William Sound, harbor seal numbers declined by about 63% overall between 1984 and 1997, including a 40% decline prior to the *Exxon Valdez* oil spill that occurred in 1989 (Frost et al. 1999, Ver Hoef and Frost 2003). Harbor seal counts in Glacier Bay National Park, where the majority of seals haul out on floating ice calved from glaciers, declined by roughly 60% between 1992 and 2001 and continued to decline through 2008 (Mathews and Pendleton 2006, Womble et al. 2010). At Aialik Bay, a site in Kenai Fjords National Park where harbor seals also haul out on ice calved from a glacier, harbor seal numbers declined by 93% from 1979 to 2009 (Hoover-Miller et al. 2011). In the Aleutian Islands, counts declined by 67% between the early 1980s and 1999, with declines of about 86% in the western Aleutians (Small et al. 2008). Although there is evidence for recent stabilization or even partial recovery of harbor seal numbers in some areas of long-term harbor seal decline, such as Tugidak Island and Nanvak Bay (Jemison et al. 2006), most have not made substantial recoveries toward historical abundances. These areas of localized declines in harbor seals contrast strongly with other large regions of Alaska where harbor seal numbers have remained stable or increased over the same period: trend monitoring regions around Ketchikan and the Kodiak area increased significantly in the 1980s and 1990s and regions around Sitka and Bristol Bay were stable (Small et al. 2003). Differences in trend across the various regions of Alaska suggest some level of independent population dynamics (O'Corry-Crowe et al. 2003, O'Corry-Crowe 2012).

The Alaska Fisheries Science Center's Marine Mammal Laboratory (MML) routinely conducts aerial surveys of harbor seals across their entire range in Alaska. Prior to 2008, Alaska was divided into five survey regions, with one region surveyed per year. In 2010, the survey sites were prioritized based on the newly defined harbor seal stock divisions, and annual aerial surveys attempt to sample the full geographic range of harbor seals in Alaska. These surveys focus, annually, on sites that make up a significant portion of each stock's population or have timely conservation interest. Sites with fewer seals are intended to be flown every 5 to 7 years. Reduced funding since 2015 has limited the scope of surveys, and efforts have been focused in regions of specific conservation interest (e.g., the Aleutian Islands).

Count data from surveys were analyzed with Bayesian hierarchical models, where true abundance per site per year was modeled with a Poisson distribution. Only a fraction of the animals could be observed, so counted seals were modeled with a binomial distribution, given the true number and a haul-out probability. The haul-out probability was modeled from bio-logging data on individual seals, using Bayesian beta regression, that accounted for date, time of day, and tide, which were also known for the counted data. The observed count data were thus adjusted for haul out by the hierarchical model. All models accounted for temporal autocorrelation, by site for count models and by seal for haul-out models, but the temporal autocorrelation parameters were pooled within stock. Models were fit with Markov chain Monte Carlo (MCMC) methods. Abundance estimates for sites were aggregated into estimates by stock, with variability in the estimates provided by the variation in the MCMC chains.

Abundance Estimates and Minimum Population Estimates

The current statewide abundance estimate for Alaska harbor seals is 243,938 (Boveng et al. 2019), based on aerial survey data collected from 1996 to 2018 (Boveng et al. 2019). See Table 1 for abundance estimates of the 12 stocks of harbor seals in Alaska. The minimum population estimate (N_{MIN}) for 11 of the 12 stocks of harbor seals in Alaska is calculated as the lower bound of the 80% credible interval obtained from the posterior distribution of abundance estimates. This approach is consistent with the definition of potential biological removal (PBR) in the current guidelines (NMFS 2016). The abundance estimate and N_{MIN} for the remaining stock, the Pribilof Islands stock, is simply the number counted in the most recent survey (2018) of this very small group.

Table 1. Abundance and 8-year trend (number of seals per year) estimates, by stock, for harbor seals in Alaska, along with respective estimates of standard error. The probability of decrease represents the proportion of the posterior probability distribution for the 8-year trend that fell below a value of 0 seals per year. N_{MIN} is the lower bound of the 80% credible interval obtained from the posterior distribution of the abundance estimates The Pribilof Islands stock abundance estimate (*) is simply the count of seals ashore during the survey and does not include a correction for seals in the water.

Stock	Year of last survey	Abundance estimate	SE	8-year trend estimate	SE	Probability of decrease	N_{MIN}
Aleutian Islands	2018	5,588	274	-131	86	0.932	5,366
Pribilof Islands	2018	229*	n/a	n/a	n/a	n/a	229
Bristol Bay	2017	44,781	7,278	1,127	1,196	0.218	38,254
North Kodiak	2017	8,677	1,335	53	236	0.409	7,609
South Kodiak	2017	26,448	5,282	1,234	1,062	0.076	22,351
Prince William Sound	2015	44,756	3,391	-200	555	0.648	41,776
Cook Inlet/Shelikof Strait	2018	28,411	1,839	-111	333	0.609	26,907
Glacier Bay/Icy Strait	2017	7,455	894	-216	147	0.904	6,680
Lynn Canal/Stephens Passage	2016	13,388	1,876	-114	262	0.73	11,867
Sitka/Chatham Strait	2015	13,289	1,734	71	277	0.41	11,883
Dixon/Cape Decision	2015	23,478	2,501	142	450	0.382	21,453
Clarence Strait	2015	27,659	3,030	138	485	0.413	24,854

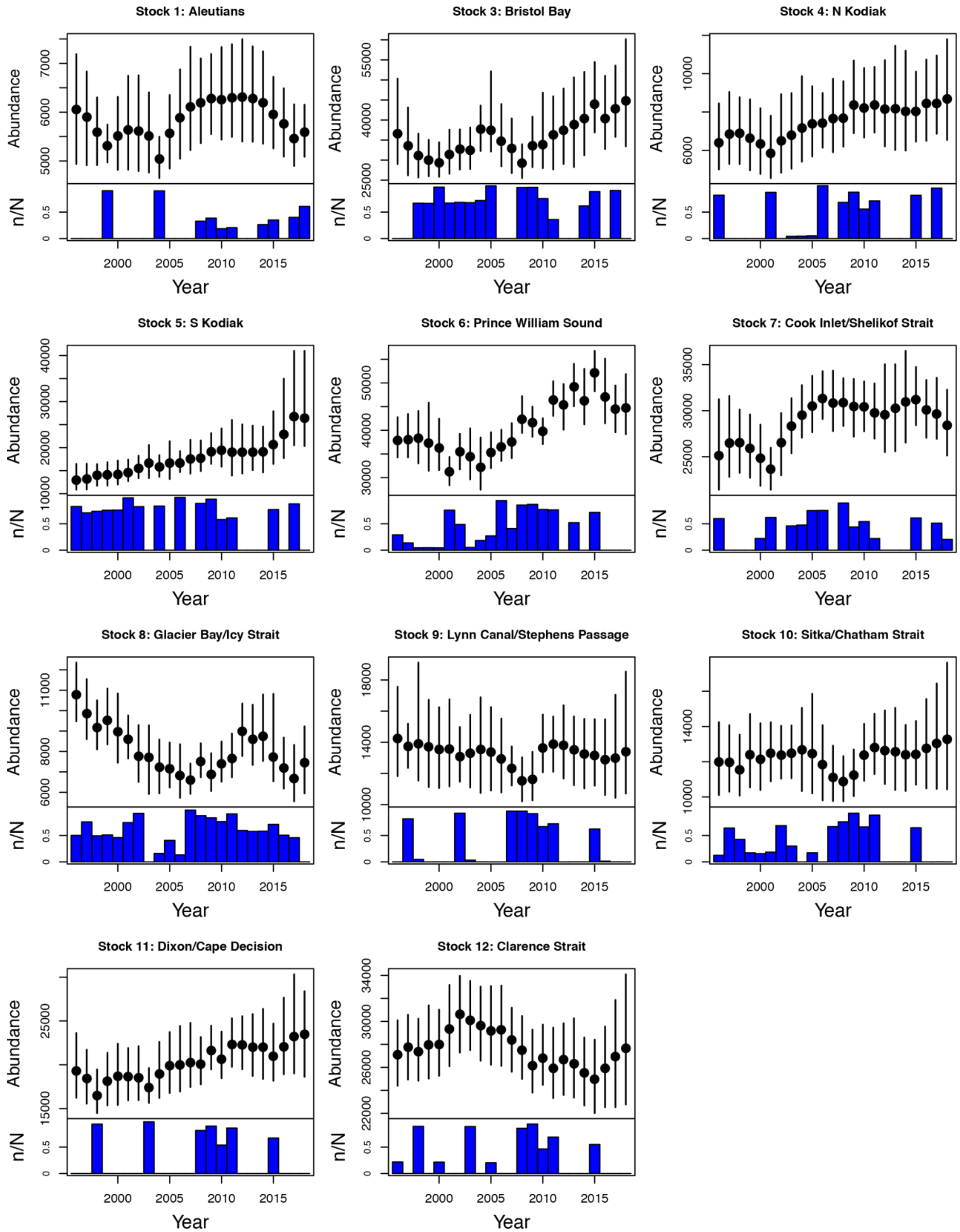

Figure 3. Annual abundance estimates (black dots) of harbor seals in Alaska for all stocks except the Pribilof Islands stock. Black lines represent the 95% credible interval. Blue bars provide a measure of survey effort and indicate the proportion of the estimated abundance likely surveyed each year.

Current Population Trend

Aerial surveys of harbor seal haul-out sites throughout Alaska have been conducted annually and provide information on trends in abundance. The most current estimates of trend (Table 1) were estimated as the means of the slopes of 1,000 simple linear regressions over the most recent eight annual estimates in each of the 1,000 MCMC samples from the posterior distributions for abundance. Thus, they are in units of seals per year, rather than the typical annual percent growth rate. There is no appropriate method for converting these estimates of trend to annual percent growth rate. As a reflection of uncertainty in trend estimates, the proportion of the posterior distribution for each stock's trend that lies below the value of 0 is used as an estimate of the probability that a stock is currently decreasing (Table 1). This allows a probabilistic determination of the qualitative trend status: a value greater than 0.5 means the evidence suggests that the stock is decreasing; a value less than 0.5 means the stock is increasing. For the estimation of trend, an 8-year time interval was used. Eight years is considered to be the approximate threshold of reliability for Marine Mammal Protection Act (MMPA) stock assessment data. One caveat of this approach is that, due to the skewness inherent in the posterior distribution, it is possible for a stock to exhibit a positive trend while also having a probability of decrease greater than 0.5. The following summarizes historical and recent information on the population trend for each of the 12 stocks.

Aleutian Islands: A partial estimate of harbor seal abundance in the Aleutian Islands was determined from skiff surveys of 106 islands from 1977 to 1982 (8,601 seals). Small et al. (2008) compared counts from the same islands during a 1999 aerial survey (2,859 seals). Counts decreased at a majority of the islands. Islands with greater than 100 seals decreased by 70%. The overall estimates showed a 67% decline during the approximate 20-year period (Small et al. 2008). Starting in 2005, the stock abundance estimates show annual increases with a peak abundance of approximately 6,500 in 2010. Since 2010, there is an apparent decline. The current estimate of the 8-year population trend in the Aleutian Islands is -131 seals per year, with a probability that the stock is decreasing of 0.932 (Table 1). Note the survey effort (as represented by n/N in Figure 3) has been consistently below 50% for the Aleutians. This stock represents the most challenging region (due to size, logistics, and weather) in Alaska for aerial surveys. Limited funds and availability of suitable aircraft have prevented greater survey coverage.

Pribilof Islands: Counts of harbor seals in the Pribilof Islands ranged from 250 to 1,224 in the 1970s. Counts in the 1980s and 1990s ranged between 119 and 232 harbor seals. Prior to July 2010, the most recent count was 202 seals in 1995. In July 2010, approximately 185 adults and 27 pups were observed on Otter Island for a maximum count of 212 harbor seals. Counts from 2010 (all ages) are nearly identical to the 1995 counts (212 vs. 202), but 2010 pup numbers were slightly less (27 vs. 42). July 2015 was the first year that counts were conducted on both Otter Island and St. George Island, resulting in a total count of 235 seals (all ages). In 2018, the Aleut Community of St. Paul and MML collaborated on a comprehensive survey of harbor seals in the Pribilof Islands using small unoccupied aircraft. The survey was conducted on the islands of Otter, St. Paul, and St. George in early September, resulting in a total of 229 seals counted across all islands (Boveng et al. 2019). For all other stocks in Alaska, the abundance and trend estimates account for the proportion of seals likely in the water during the survey. This is not done for the Pribilof Island stock because counts have typically been more opportunistic and information on environmental covariates is less standardized. It is also possible the isolated and unique nature of the habitat could lead to very different haul-out behaviors that are unknown without conducting a behavioral study. Analysis of the nearest two stocks (Aleutian Islands and Bristol Bay) estimated standardized correction factors of 1.5 and 3.0. Using the mean correction factor of 2.25 would result in approximately 515 harbor seals in the Pribilof Island region. The current population trend in the Pribilof Islands is unknown.

Bristol Bay: At Nanvak Bay, the largest haul-out location in northern Bristol Bay, harbor seals declined in abundance from 1975 to 1990 and increased from 1990 to 2000 (Jemison et al. 2006). Land-based harbor seal counts at Nanvak Bay from 1990 to 2000 increased at 9.2% per year during the pupping period and 2.1% per year during the molting period (Jemison et al. 2006). After a period of growth in the 1980s, the population in Iliamna Lake appears to be relatively stable at around 400 individuals. A population viability analysis assessing the risk of quasi-extinction in Iliamna Lake, defined as any reduction to 50 animals or below in the next 100 years, ranged from 1% to 3%, depending on the prior scenario (Boveng et al. 2018). The current 8-year estimate of the population trend in the Bristol Bay stock is +1,127 seals per year, with a probability that the stock is decreasing of 0.218 (Table 1).

North Kodiak: The current 8-year estimate of the North Kodiak population trend is +53 seals per year, with a probability that the stock is decreasing of 0.409 (Table 1). The North Kodiak stock appears to have levelled off since 2010 at approximately 8,000 seals.

South Kodiak: A significant portion of the harbor seal population within the South Kodiak stock is located at and around Tugidak Island off the southwest coast of Kodiak Island. Sharp declines in the number of seals present on Tugidak were observed between 1976 and 1998. The highest rate of decline was 21% per year between 1976 and 1979 (Pitcher 1990). While the number of seals on Tugidak has stabilized and shown some evidence of increase since the decline, the population in 2000 remained reduced by 80% compared to the levels in the 1970s (Jemison et al. 2006). The South Kodiak stock has shown a consistent, increasing trend since the low levels in the mid-1990s, with an even more noticeable increase in recent years. The current 8-year estimate of the South Kodiak population trend is +1,234 seals per year, with a probability that the stock is decreasing of 0.076 (Table 1).

Prince William Sound: The Prince William Sound stock includes harbor seals both within and adjacent to Prince William Sound proper. Within Prince William Sound proper, harbor seals declined in abundance by 63% between 1984 and 1997 (Frost et al. 1999). In Aialik Bay, adjacent to Prince William Sound proper, there has been a decline in pup production by 4.6% annually from 40 down to 32 pups born from 1994 to 2009 (Hoover-Miller et al. 2011). The current 8-year estimate of the Prince William Sound population trend is -200 seals per year, with a probability that the stock is decreasing of 0.648 (Table 1). There has been limited survey effort outside of glacial habitats in recent years and, thus, the most recent abundance estimates have larger credible intervals.

Cook Inlet/Shelikof Strait: A multi-year study of seasonal movements and abundance of harbor seals in Cook Inlet was conducted between 2004 and 2007. This study involved multiple aerial surveys throughout the year, and the data indicated a stable population of harbor seals during the August molting period (Boveng et al. 2011). Aerial surveys along the Alaska Peninsula present greater logistical challenges and have therefore been conducted less frequently. The current 8-year estimate of the Cook Inlet/Shelikof Strait population trend is -111 seals per year, with a probability that the stock is decreasing of 0.609 (Table 1).

Glacier Bay/Icy Strait: The Glacier Bay/Icy Strait stock showed a negative population trend estimate for harbor seals from 1992 to 2008 in June and August for glacial (-7.7%/yr; -8.2%/yr) and terrestrial sites (-12.4%/yr, August only) (Womble et al. 2010). Trend estimates by Mathews and Pendleton (2006) were similarly negative for both glacial and terrestrial sites. Long-term monitoring of harbor seals on glacial ice has occurred in Glacier Bay since the 1970s (Mathews and Pendleton 2006) and has shown this area to support one of the largest breeding aggregations in Alaska (Steveler 1979, Calambokidis et al. 1987). After a dramatic retreat of Muir Glacier (more than 7 km), in the East Arm of Glacier Bay, between 1973 and 1986 and the subsequent grounding and cessation of calving in 1993, floating glacial ice was greatly reduced as a haul-out substrate for harbor seals and ultimately resulted in the abandonment of upper Muir Inlet by harbor seals (Calambokidis et al. 1987, Hall et al. 1995, Mathews 1995). Prior to 1993, seal counts were up to 1,347 in the East Arm of Glacier Bay; 2008 counts were fewer than 200 (Streveler 1979, Molnia 2007). The current 8-year estimate of the Glacier Bay/Icy Strait population trend is -216 seals per year, with a probability that the stock is decreasing of 0.904 (Table 1). The majority of survey effort in recent years has been conducted by the National Park Service and focused, mostly, on glacial ice habitats. Limited surveys have been conducted in the Icy Strait portion of the stock.

Lynn Canal/Stephens Passage: The current 8-year estimate of the Lynn Canal/Stephens Passage population trend is -114 seals per year, with a probability that the stock is decreasing of 0.73 (Table 1). Outside of efforts in 2007 to 2011 and 2015, there has been limited survey effort for this stock and, thus, the recent estimates of abundance include large credible intervals.

Sitka/Chatham Strait: The current 8-year estimate of the Sitka/Chatham Strait population trend is +71 seals per year, with a probability that the stock is decreasing of 0.41 (Table 1). Outside of efforts in 2007 to 2011 and 2015, there has been limited survey effort for this stock and, thus, the recent estimates of abundance include large credible intervals.

Dixon/Cape Decision: The current 8-year estimate of the Dixon/Cape Decision population trend is +142 seals per year, with a probability that the stock is decreasing of 0.382 (Table 1). Outside of efforts in 2007 to 2011 and 2015, there has been limited survey effort for this stock and, thus, the recent estimates of abundance include large credible intervals.

Clarence Strait: The current 8-year estimate of the Clarence Strait population trend is +138 seals per year, with a probability that the stock is decreasing of 0.413 (Table 1). Outside of efforts in 2007 to 2011 and 2015, there has been limited survey effort for this stock and, thus, the recent estimates of abundance include large credible intervals.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Reliable rates of maximum net productivity have not been estimated directly from the 12 stocks of harbor seals identified in Alaska. Based on monitoring in Washington State from 1978 to 1999, Jeffries et al. (2003) estimated R_{MAX} to be 12.6% and 18.5% for harbor seals of the inland and coastal stocks, respectively. Harbor seals have been protected in British Columbia since 1970, and the monitored portion of that population responded with an annual rate of increase of approximately 12.5% through the late 1980s (Olesiuk et al. 1990), although a more recent evaluation suggested that 11.5% may be a more appropriate figure (Fisheries and Oceans Canada 2010). These empirical estimates of R_{MAX} indicate that the continued use of the pinniped maximum theoretical net productivity rate of 12% is appropriate for the Alaska stocks (NMFS 2016).

POTENTIAL BIOLOGICAL REMOVAL

Potential biological removal (PBR) is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: PBR = $N_{MIN} \times 0.5R_{MAX} \times F_R$. Marine mammal stocks such as the harbor seal stocks in Alaska that are taken by subsistence hunting may be given F_R values up to 1.0, provided they are “known to be increasing” or “not known to be decreasing” and “there have not been recent increases in the levels of takes” (NMFS 2016). For harbor seals in Alaska, these guidelines were followed by assigning all harbor seal stocks an initial, default recovery factor of 0.5. The default value was adjusted up to 0.7 if the estimated probability of decrease was less than 0.3. The value was adjusted down to 0.3 if the estimated probability of decrease was greater than 0.7. This provides a simple, balanced approach for providing a recovery factor consistent with current guidelines while incorporating results from novel statistical methods. Table 2 summarizes the PBR levels for each stock of harbor seals in Alaska based on N_{MIN} estimates, an R_{MAX} of 12%, and F_R values.

Table 2. PBR calculations by stock for harbor seals in Alaska. The N_{MIN} values are determined from the 20th percentile of the posterior distribution for stock-level abundance estimates, except for the Pribilof Islands. A default value of 0.5 was used as the recovery factor. Based on evaluation of the trend estimates and probability of decrease, the recovery factor for some stocks was increased to 0.7. For other stocks, the recovery factor was decreased to 0.3.

Stock	N_{MIN}	R_{MAX}	Recovery Factor (F_R) (default value = 0.5)	PBR
Aleutian Islands	5,366	0.12	0.3	97
Pribilof Islands	229	0.12	0.5	7
Bristol Bay	38,254	0.12	0.7	1,607
North Kodiak	7,609	0.12	0.5	228
South Kodiak	22,351	0.12	0.7	939
Prince William Sound	41,776	0.12	0.5	1,253
Cook Inlet/Shelikof Strait	26,907	0.12	0.5	807
Glacier Bay/Icy Strait	6,680	0.12	0.3	120
Lynn Canal/Stephens Passage	11,867	0.12	0.3	214
Sitka/Chatham Strait	11,883	0.12	0.5	356
Dixon/Cape Decision	21,453	0.12	0.5	644
Clarence Strait	24,854	0.12	0.5	746

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2013 and 2017 is listed, by marine mammal stock, in Delean et al. (2020);

however, only the mortality and serious injury data are included in the Stock Assessment Reports. The minimum estimated mean annual level of human-caused mortality and serious injury for all harbor seal stocks between 2013 and 2017 is 1,135 harbor seals: 32 in U.S. commercial fisheries, 0.4 in unknown (commercial, recreational, or subsistence) fisheries, 3.7 due to other causes (illegal shooting, entanglement in ADF&G research trawl gear), and 1,099 in the Alaska Native subsistence harvest. Human-caused mortality and serious injury information for individual harbor seal stocks is listed in the Status of Stock section for each stock. Additional potential threats most likely to result in direct human-caused mortality or serious injury for all stocks of harbor seals include unmonitored subsistence harvests, incidental takes in unmonitored fisheries, and illegal shooting. Disturbance by cruise vessels is an additional threat for harbor seal stocks that occur in glacial fjords (Jansen et al. 2010, 2015; Matthews et al. 2016).

Fisheries Information

Information (including observer programs, observer coverage, and observed incidental takes of marine mammals) for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is presented in Appendices 3-6 of the Alaska Stock Assessment Reports.

Observer programs have documented mortality and serious injury of harbor seals in the Bering Sea/Aleutian Islands Atka mackerel trawl, Bering Sea/Aleutian Islands flatfish trawl, Bering Sea/Aleutian Islands pollock trawl, Bering Sea/Aleutian Islands rockfish trawl, Bering Sea/Aleutian Islands Pacific cod pot, Gulf of Alaska flatfish trawl, and Gulf of Alaska halibut longline fisheries between 2013 and 2017 (Breiwick 2013; MML, unpubl. data) (Table 3).

Although a reliable estimate of the overall mortality and serious injury rate incidental to commercial fisheries is currently unavailable because of the absence of observer placements in salmon gillnet fisheries known to interact with several of these stocks, for the purposes of stock assessment, mean annual mortality and serious injury rates are assigned to the following harbor seal stocks based on the location of takes in observed fisheries between 2013 and 2017 (Table 3): Aleutian Islands stock: 0.2 from the Bering Sea/Aleutian Islands Atka mackerel trawl fishery + 0.2 from the Bering Sea/Aleutian Islands rockfish trawl fishery; Bristol Bay stock: 0.8 from the Bering Sea/Aleutian Islands flatfish trawl fishery + 0.2 from the Bering Sea/Aleutian Islands pollock trawl fishery + 2.8 from the Bering Sea/Aleutian Islands Pacific cod pot fishery; North Kodiak stock: 0.3 from the Gulf of Alaska flatfish trawl fishery; South Kodiak stock: 1.0 from the Gulf of Alaska flatfish trawl fishery; Cook Inlet/Shelikof Strait stock: 0.7 from the Gulf of Alaska flatfish trawl fishery + 1.8 from the Gulf of Alaska halibut longline fishery.

Table 3. Summary of incidental mortality and serious injury of harbor seals in Alaska due to U.S. commercial fisheries between 2013 and 2017 and calculation of the mean annual mortality and serious injury rate (Breiwick 2013; MML, unpubl. data).

Fishery name	Years	Data type	Percent observer coverage	Observed mortality	Estimated mortality	Mean estimated annual mortality
Bering Sea/Aleutian Is. Atka mackerel trawl	2013 2014 2015 2016 2017	obs	99 100 100 98 100	0 0 0 1[AI] 0	0 0 0 1.1[AI] 0	0.2[AI] (CV = 0.25)
Bering Sea/Aleutian Is. flatfish trawl	2013 2014 2015 2016 2017	obs data	100 100 100 99 100	0 1[BB] 0 0 3[BB]	0 1[BB] 0 0 3[BB]	0.8[BB] (CV = 0.02)
Bering Sea/Aleutian Is. pollock trawl	2013 2014 2015 2017 2017	obs data	98 98 99 99 99	0 1[BB] 0 0 0	0 1.0[BB] 0 0 0	0.2[BB] (CV = 0.14)

Fishery name	Years	Data type	Percent observer coverage	Observed mortality	Estimated mortality	Mean estimated annual mortality
Bering Sea/Aleutian Is. rockfish trawl	2013	obs data	100	0	0	0.2[AI] (CV = 0.05)
	2014		100	1[AI]	1[AI]	
	2015		100	0	0	
	2016		100	0	0	
	2017		100	0	0	
Bering Sea/Aleutian Is. Pacific cod pot	2013	obs data	18	0	0	2.4[BB] (+0.4[BB])[c] (CV = 0.78)
	2014		21	2[BB] (+2[BB])[a]	12[BB] (+2[BB])[b]	
	2015		27	0	0	
	2016		21	0	0	
	2017		13	0	0	
Gulf of Alaska flatfish trawl	2013	obs data	46	2[SK]	5.2[SK]	1.0[SK] + 0.3[NK] + 0.7[CI] (CV = 0.34)[d]
	2014		47	0	0	
	2015		54	0	0	
	2016		39	0	0	
	2017		56	1[NK] + 2[CI]	1.7[NK] + 3.3[CI]	
Gulf of Alaska halibut longline	2013	obs data	4.2	0	0	1.8[CI] (CV = 0.95)
	2014		11	0	0	
	2015		9.4	1[CI]	9.1[CI]	
	2016		9.5	0	0	
	2017		4.6	0	0	
Minimum total estimated annual mortality						0.4[AI] + 3.8[BB] + 0.3[NK] + 1.0[SK] + 2.5[CI] (CV = 0.34)[e]

[a]Total mortality and serious injury observed in 2014: 2 harbor seals in sampled hauls + 2 harbor seals in unsampled hauls.
[b]Total estimate of mortality and serious injury in 2014: 12 harbor seals (extrapolated estimate from 2 harbor seals observed in sampled hauls) + 2 harbor seals (2 harbor seals observed in unsampled hauls).
[c]Mean annual mortality and serious injury for fishery: 2.4 harbor seals (mean of extrapolated estimates from sampled hauls) + 0.4 harbor seals (mean of number observed in unsampled hauls).
[d]This CV is for the mean estimated annual mortality for all harbor seal stocks taken in the fishery.
[e]This CV is for the sum of the mean estimated annual mortality for all stocks.

Harbor seal stock identifications for observed mortality, estimated mortality, and mean estimated annual mortality:
[AI]Aleutian Islands stock
[BB]Bristol Bay stock
[NK]North Kodiak stock
[SK]South Kodiak stock
[CI]Cook Inlet/Shelikof Strait stock

Observer programs in Alaska State-managed salmon set gillnet and salmon drift gillnet fisheries have documented harbor seal mortality and serious injury (Table 4). The Prince William Sound salmon drift gillnet fishery is known to interact with harbor seals, although the most recent observer data available for this fishery are from 1990 and 1991 (Wynne et al. 1991, 1992). The minimum estimated average annual mortality and serious injury rate (24 seals) in this fishery will be applied to the Prince William Sound stock of harbor seals. Although the observer data are dated, they are considered the best available data on mortality and serious injury levels in this fishery.

Observers reported a South Kodiak harbor seal mortality in a federally-managed U.S. commercial Gulf of Alaska pot fishery in 2014; however, there was not enough information in the record to assign the event to a specific fishery. Therefore, the observed mortality is used to calculate a mean annual mortality and serious injury rate of 0.2 South Kodiak harbor seals in commercial Gulf of Alaska pot fisheries between 2013 and 2017 (Delean et al. 2020; Table 5).

Table 4. Summary of incidental mortality and serious injury of harbor seals in Alaska due to U.S. commercial salmon drift and set gillnet fisheries in 1990 and 1991 and calculation of the mean annual mortality and serious injury rate based on the most recent observer program data available (Wynne et al. 1991, 1992).

Fishery name	**Years**	**Data type**	**Percent observer coverage**	**Observed mortality**	**Estimated mortality**	**Mean estimated annual mortality**
Prince William Sound salmon drift gillnet	1990 1991	obs data	4 5	2 1	36 12	24 (CV = 0.50)
Minimum total estimated annual mortality						24 (CV = 0.50)

Reports to the NMFS Alaska Region stranding network of harbor seals entangled in fishing gear or with injuries caused by interactions with gear are another source of mortality and serious injury data (Delean et al. 2020). Between 2013 and 2017, there were two reports of Cook Inlet/Shelikof Strait harbor seal mortality and serious injury due to entanglements in fishing gear, including one in a Cook Inlet salmon set gillnet in 2014 and one in an unidentified net in 2017, resulting in a mean annual mortality and serious injury rate of 0.4 harbor seals from this stock due to interactions with unknown (commercial, recreational, or subsistence) fisheries (Table 5).

Table 5. Summary of harbor seal mortality and serious injury, by year, type, and harbor seal stock, reported to the NMFS Alaska Region marine mammal stranding network between 2013 and 2017 (Delean et al. 2020).

Cause of injury	**2013**	**2014**	**2015**	**2016**	**2017**	**Mean annual mortality**
Gulf of Alaska commercial pot fishery	0	1^{SK}	0	0	0	0.2^{SK}
Entangled in Cook Inlet salmon set gillnet*	0	1^{CI}	0	0	0	0.2^{CI}
Entangled in unidentified net*	0	0	0	0	1^{CI}	0.2^{CI}
Illegally shot[a]	-	-	1^{PW}	3^{PW}	3^{PW}	2.3^{PW}
Illegally shot	0	0	0	6^{BB}	0	1.2^{BB}
Entangled in ADF&G research trawl gear	0	1^{NK}	0	0	0	0.2^{NK}
Total in commercial fisheries *Total in unknown (commercial, recreational, or subsistence) fisheries Total due to other causes (illegally shot, research fisheries)						0.2^{SK} 0.4^{CI} $2.3^{PW} + 1.2^{BB} + 0.2^{NK}$

[a]Dedicated effort to survey the Copper River Delta for stranded marine mammals began in 2015 in response to a high number of reported strandings, some of which were later determined to be human-caused (illegally shot). Dedicated surveys were also conducted in 2016 and 2017. Because similar data are not available for 2013 and 2014, the data were averaged over the 3 years of survey effort for a more informed estimate of mean annual mortality.

Harbor seal stock identifications for observed mortality and mean annual mortality:
^{BB}Bristol Bay stock
^{NK}North Kodiak stock
^{SK}South Kodiak stock
^{CI}Cook Inlet/Shelikof Strait stock
^{PW}Prince William Sound stock

Alaska Native Subsistence/Harvest Information

The Alaska Native subsistence harvest of harbor seals has been estimated by the Alaska Native Harbor Seal Commission (ANHSC) and the Alaska Department of Fish and Game (ADF&G). Information from the ADF&G indicates the average harvest levels for the 12 stocks of harbor seals identified in Alaska from 2004 to 2008, including struck and lost animals (Table 6: average annual harvest column). Data on community subsistence harvests were collected for Kodiak Island, Prince William Sound, and Southeast Alaska in 2011 and 2012, Prince William Sound and Cook Inlet/Shelikof Strait in 2014, and Bristol Bay in 2017 (Table 6: annual harvest columns). The remaining stocks do not have updated community subsistence data, therefore, the most recent 5-years of harvest data (2004-2008) will be used for these stocks.

Table 6. Summary of the subsistence harvest data for all 12 harbor seal stocks in Alaska, 2004-2008, 2011-2012, 2014, and 2017. Data are from Wolfe et al. (2005, 2006, 2008, 2009a, 2009b, 2012, 2013); NMFS, unpubl. data.

Stock	Minimum annual harvest 2004-2008	Maximum annual harvest 2004-2008	Average annual harvest 2004-2008	Annual harvest 2011 or 2012	Annual harvest 2014	Annual harvest 2017
Aleutian Islands	50	146	90	N/A	N/A	N/A
Pribilof Islands	0	0	0	N/A	N/A	N/A
Bristol Bay[a]	82	188	141	N/A	N/A	15[b]
North Kodiak	66	260	131	37	N/A	N/A
South Kodiak	46	126	78	126	N/A	N/A
Prince William Sound	325	600	439	255[c]	387	N/A
Cook Inlet/Shelikof Strait	177	288	233	N/A	104	N/A
Glacier Bay/Icy Strait	22	108	52	104	N/A	N/A
Lynn Canal/Stephens Passage	17	60	30	50	N/A	N/A
Sitka/Chatham Strait	97	314	222	77	N/A	N/A
Dixon/Cape Decision	100	203	157	69	N/A	N/A
Clarence Strait	71	208	164	40	N/A	N/A

[a]Seals taken in summer on shore in Bristol Bay could be either harbor seals or spotted seals. Absent specific identification, we have listed the species as reported to the ADF&G. NMFS will work with the organizations that work with harbor seals to determine how to apportion the harvest in this area between the two species.
[b]This is a minimum estimate because it includes subsistence harvest data from only one community (Clark's Point) and does not include the number of struck and lost animals.
[c]This is a minimum estimate because it includes subsistence harvest data from only one community (Yakutat).

Other Mortality

Reports to the NMFS Alaska Region stranding network of harbor seals entangled in marine debris or with injuries caused by other types of human interaction are another source of mortality and serious injury data (Delean et al. 2020). These mortality and serious injury estimates result from an actual count of verified human-caused deaths and serious injuries and are minimums because not all entangled animals strand nor are all stranded animals found, reported, or have the cause of death determined. From 2013 to 2017, reports to the NMFS Alaska Region stranding network resulted in mean annual mortality and serious injury rates of 2.3 Prince William Sound harbor seals illegally shot in the Copper River Delta (3-year average), 1.2 Bristol Bay harbor seals illegally shot, and 0.2 North Kodiak harbor seals entangled in ADF&G research trawl gear. Gunshot mortality of an additional five harbor seals was reported to the NMFS Alaska Region between 2013 and 2017, including two Cook Inlet/Shelikof Strait harbor seals (one each in 2013 and 2014) and three Prince William Sound harbor seals (two in 2014 and one in 2015). However, these events are not included in the estimate of the mean annual mortality and serious injury rate for 2013 to 2017 because it could not be confirmed that the deaths were due to illegal shooting and were not already accounted for in the estimate of animals struck and lost in the Alaska Native subsistence harvest.

STATUS OF STOCK

No harbor seal stocks in Alaska are designated as depleted under the MMPA or listed as threatened or endangered under the Endangered Species Act, and the minimum estimate of the mean annual level of human-caused mortality and serious injury does not exceed PBR for any of the stocks; therefore, none of the stocks are strategic. At present, mean annual mortality and serious injury rates incidental to U.S. commercial fisheries that are less than 10% of PBR can be considered insignificant and approaching a zero mortality and serious injury rate. Reliable estimates of the mean annual rates of mortality and serious injury incidental to U.S. commercial fisheries are unavailable. Therefore, it is unknown whether the mean annual mortality and serious injury rates due to U.S. commercial fishing are insignificant. The status of all 12 stocks of harbor seals identified in Alaska relative to their Optimum Sustainable Population is unknown.

There are key uncertainties in the assessment of the abundance and trend of harbor seals in Alaska. The population abundance is based on counts of visible animals and adjusted to account for seals in the water based on haul-out behavior data obtained from bio-logging studies. These deployments are confined to a small portion of the geographic range and only a portion of the recognized stocks. Additionally, many of these deployments rely on bio-loggers attached to seal hair with adhesive. These tags fall off during the annual molt. Since the surveys are typically conducted during the molt period, there is some additional uncertainty due to reduced sample size. Reduced funding and limited availability of suitable aircraft has prevented regular surveys that properly sample the full expanse of harbor seal distribution in Alaska. Instead, resources are prioritized to areas of special conservation or management concern. This means some stocks or portions of stocks are not surveyed annually and, consequently, uncertainty is increased for those areas.

In addition to uncertainties related to assessment, evaluation and documentation of human-caused mortality could be improved. There are multiple nearshore commercial fisheries which are not observed; thus, there is likely to be unreported fishery-related mortality and serious injury of harbor seals. Estimates of human-caused mortality and serious injury from stranding data are underestimates because not all animals strand nor are all stranded animals found, reported, or have the cause of death determined.

Aleutian Islands: At present, U.S. commercial fishery-related mean annual mortality and serious injury rates less than 9.7 animals (i.e., 10% of PBR) can be considered insignificant and approaching a zero mortality and serious injury rate. A reliable estimate of the mean annual rate of mortality and serious injury incidental to U.S. commercial fisheries is unavailable. Therefore, it is unknown whether the mean annual mortality and serious injury rate due to U.S. commercial fishing is insignificant. Based on the best scientific information available, the minimum estimated mean annual level of human-caused mortality and serious injury (0.4 (commercial fisheries) + 90 (harvest) + 0 (other fisheries + other mortality and serious injury) = 90) is not known to exceed the PBR (97). The Aleutian Islands stock of harbor seals is not classified as a strategic stock.

Pribilof Islands: At present, U.S. commercial fishery-related mean annual mortality and serious injury rates less than 0.7 animals (i.e., 10% of PBR) can be considered insignificant and approaching a zero mortality and serious injury rate. A reliable estimate of the mean annual rate of mortality and serious injury incidental to U.S. commercial fisheries is unavailable. Therefore, it is unknown whether the mean annual mortality and serious injury rate due to U.S. commercial fishing is insignificant. Based on the best scientific information available, the minimum estimated mean annual level of human-caused mortality and serious injury (0 + 0 + 0 = 0) is not known to exceed the PBR (7). The Pribilof Islands stock of harbor seals is not classified as a strategic stock.

Bristol Bay: At present, U.S. commercial fishery-related mean annual mortality and serious injury rates less than 161 animals (i.e., 10% of PBR) can be considered insignificant and approaching a zero mortality and serious injury rate. A reliable estimate of the mean annual rate of mortality and serious injury incidental to U.S. commercial fisheries is unavailable. Therefore, it is unknown whether the mean annual mortality and serious injury rate due to U.S. commercial fishing is insignificant. Based on the best scientific information available, the minimum estimated mean annual level of human-caused mortality and serious injury (3.8 + 15 + 1.2 = 20) is not known to exceed the PBR (1,607). The Bristol Bay stock of harbor seals is not classified as a strategic stock.

North Kodiak: At present, U.S. commercial fishery-related mean annual mortality and serious injury rates less than 23 animals (i.e., 10% of PBR) can be considered insignificant and approaching a zero mortality and serious injury rate. A reliable estimate of the mean annual rate of mortality and serious injury incidental to U.S. commercial fisheries is unavailable. Therefore, it is unknown whether the mean annual mortality and serious injury rate due to U.S. commercial fishing is insignificant. Based on the best scientific information available, the minimum estimated mean annual level of human-caused mortality and serious injury (0.3 + 37 + 0.2 = 38) is not known to exceed the PBR (228). The North Kodiak stock of harbor seals is not classified as a strategic stock.

South Kodiak: At present, U.S. commercial fishery-related mean annual mortality and serious injury rates less than 94 animals (i.e., 10% of PBR) can be considered insignificant and approaching a zero mortality and serious injury rate. A reliable estimate of the mean annual rate of mortality and serious injury incidental to U.S. commercial fisheries is unavailable. Therefore, it is unknown whether the mean annual mortality and serious injury rate due to U.S. commercial fishing is insignificant. Based on the best scientific information available, the minimum estimated mean annual level of human-caused mortality and serious injury (1.2 + 126 + 0 = 127) is not known to exceed the PBR (939). The South Kodiak stock of harbor seals is not classified as a strategic stock.

Prince William Sound: At present, U.S. commercial fishery-related mean annual mortality and serious injury rates less than 125 animals (i.e., 10% of PBR) can be considered insignificant and approaching a zero mortality and serious injury rate. A reliable estimate of the mean annual rate of mortality and serious injury incidental to U.S. commercial fisheries is unavailable. Therefore, it is unknown whether the mean annual mortality and serious injury rate due to U.S. commercial fishing is insignificant. Based on the best scientific information available, the minimum estimated mean annual level of human-caused mortality and serious injury (24 + 387 + 2.3 = 413) is not known to exceed the PBR (1,253). The Prince William Sound stock of harbor seals is not classified as a strategic stock.

Cook Inlet/Shelikof Strait: At present, U.S. commercial fishery-related mean annual mortality and serious injury rates less than 81 animals (i.e., 10% of PBR) can be considered insignificant and approaching a zero mortality and serious injury rate. A reliable estimate of the mean annual rate of mortality and serious injury incidental to U.S. commercial fisheries is unavailable. Therefore, it is unknown whether the mean annual mortality and serious injury rate due to U.S. commercial fishing is insignificant. Based on the best scientific information available, the minimum estimated mean annual level of human-caused mortality and serious injury (2.5 + 104 + 0.4 = 107) is not known to exceed the PBR (807). The Cook Inlet/Shelikof Strait stock of harbor seals is not classified as a strategic stock.

Glacier Bay/Icy Strait: At present, U.S. commercial fishery-related mean annual mortality and serious injury rates less than 12 animals (i.e., 10% of PBR) can be considered insignificant and approaching a zero mortality and serious injury rate. A reliable estimate of the mean annual rate of mortality and serious injury incidental to U.S. commercial fisheries is unavailable. Therefore, it is unknown whether the mean annual mortality and serious injury rate due to U.S. commercial fishing is insignificant. Based on the best scientific information available, the minimum estimated mean annual level of human-caused mortality and serious injury (0 + 104 + 0 = 104) is not known to exceed the PBR (120). The Glacier Bay/Icy Strait stock of harbor seals is not classified as a strategic stock.

Lynn Canal/Stephens Passage: At present, U.S. commercial fishery-related mean annual mortality and serious injury rates less than 21 animals (i.e., 10% of PBR) can be considered insignificant and approaching a zero mortality and serious injury rate. A reliable estimate of the mean annual rate of mortality and serious injury incidental to U.S. commercial fisheries is unavailable. Therefore, it is unknown whether the mean annual mortality and serious injury rate due to U.S. commercial fishing is insignificant. Based on the best scientific information available, the minimum estimated mean annual level of human-caused mortality and serious injury (0 + 50 + 0 = 50) is not known to exceed the PBR (214). The Lynn Canal/Stephens Passage stock of harbor seals is not classified as a strategic stock.

Sitka/Chatham Strait: At present, U.S. commercial fishery-related mean annual mortality and serious injury rates less than 36 animals (i.e., 10% of PBR) can be considered insignificant and approaching a zero mortality and serious injury rate. A reliable estimate of the mean annual rate of mortality and serious injury incidental to U.S. commercial fisheries is unavailable. Therefore, it is unknown whether the mean annual mortality and serious injury rate due to U.S. commercial fishing is insignificant. Based on the best scientific information available, the minimum estimated mean annual level of human-caused mortality and serious injury (0 + 77 + 0 = 77) is not known to exceed the PBR (356). The Sitka/Chatham Strait stock of harbor seals is not classified as a strategic stock.

Dixon/Cape Decision: At present, U.S. commercial fishery-related mean annual mortality and serious injury rates less than 64 animals (i.e., 10% of PBR) can be considered insignificant and approaching a zero mortality and serious injury rate. A reliable estimate of the mean annual rate of mortality and serious injury incidental to U.S. commercial fisheries is unavailable. Therefore, it is unknown whether the mean annual mortality and serious injury rate due to U.S. commercial fishing is insignificant. Based on the best scientific information available, the minimum estimated mean annual level of human-caused mortality and serious injury (0 + 69 + 0 = 69) is not known to exceed the PBR (644). The Dixon/Cape Decision stock of harbor seals is not classified as a strategic stock.

Clarence Strait: At present, U.S. commercial fishery-related mean annual mortality and serious injury rates less than 75 animals (i.e., 10% of PBR) can be considered insignificant and approaching a zero mortality and serious injury rate. A reliable estimate of the mean annual rate of mortality and serious injury incidental to U.S. commercial fisheries is unavailable. Therefore, it is unknown whether the mean annual mortality and serious injury rate due to U.S. commercial fishing is insignificant. Based on the best scientific information available, the minimum estimated mean annual level of human-caused mortality and serious injury (0 + 40 + 0 = 40) is not known to exceed the PBR (746). The Clarence Strait stock of harbor seals is not classified as a strategic stock.

HABITAT CONCERNS

Glacial fjords in Alaska are critical for harbor seal whelping, nursing, and molting. Several of these areas have experienced a ten-fold increase in tour ship visitation since the 1980s. This increase in the presence of tour vessels has resulted in additional levels of disturbance to pups and adults (Jansen et al. 2015, Matthews et al. 2016). The level of serious injury or mortality resulting from increased disturbance is not known.

CITATIONS

Bigg, M. A. 1969. The harbor seal in British Columbia. Bull. Fish. Res. Board Can. 172:1-33.

Bigg, M. A. 1981. Harbour seal: *Phoca vitulina* Linnaeus, 1758, and *Phoca largha* Pallas, 1811, p. 1-27. *In* S. H. Ridgway and R. J. Harrison (eds.), Handbook of Marine Mammals. Volume 2: Seals. Academic Press, London, UK.

Boveng, P. L., J. M. London, R. A. Montgomery, and J. M. Ver Hoef. 2011. Distribution and abundance of harbor seals in Cook Inlet, Alaska. Task I: Aerial surveys of seals ashore, 2003-2007. Final Report. BOEM Report 2011-063. Bureau of Ocean Energy Management, Alaska Outer Continental Shelf Region, Anchorage, AK. 46 p.

Boveng, P. L., J. M. London, and J. M. Ver Hoef. 2012. Distribution and abundance of harbor seals in Cook Inlet, Alaska. Task III: Movements, marine habitat use, diving behavior, and population structure, 2004-2006. Final Report. BOEM Report 2012-065. Bureau of Ocean Energy Management, Alaska Outer Continental Shelf Region, Anchorage, AK. 58 p.

Boveng, P. L., J. M. Ver Hoef, D. E. Withrow, and J. M. London. 2018. A Bayesian analysis of abundance, trend, and population viability for harbor seals in Iliamna Lake, Alaska. Risk Analysis 38:1988-2009. DOI: dx.doi.org/10.1111/risa.12988 .

Boveng, P. L., J. M. London, J. M. Ver Hoef, J. K. Jansen, and S. Hardy. 2019. Abundance and trend of harbor seals in Alaska, 2004-2018. Memorandum to the Record. Available from Marine Mammal Laboratory, AFSC, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Breiwick, J. M. 2013. North Pacific marine mammal bycatch estimation methodology and results, 2007-2011. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-260, 40 p.

Calambokidis, J., B. L. Taylor, S. D. Carter, G. H. Steiger, P. K. Dawson, and L. D. Antrim. 1987. Distribution and haul-out behavior of harbor seals in Glacier Bay, Alaska. Can. J. Zool. 65:1391-1396.

Delean, B. J., V. T. Helker, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, J. Jannot, and N. C. Young. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2013-2017. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-401, 86 p.

Fisher, H. D. 1952. The status of the harbour seal in British Columbia, with particular reference to the Skeena River. Bull. Fish. Res. Board Can. 93:1-58.

Fisheries and Oceans Canada. 2010. Population assessment Pacific harbour seal (*Phoca vitulina richardsi*). DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2009/011.

Frost, K. J., L. F. Lowry, and J. M. Ver Hoef. 1999. Monitoring the trend of harbor seals in Prince William Sound, Alaska, after the *Exxon Valdez* oil spill. Mar. Mammal Sci. 15:494-506.

Hall, D. K., C. S. Benson, and W. O. Field. 1995. Changes of glaciers in Glacier Bay, Alaska, using ground and satellite measurements. Phys. Geogr. 16:27-41.

Härkönen, T., and K. C. Harding. 2001. Spatial structure of harbour seal populations and the implications thereof. Can. J. Zool. 79:2115-2127.

Hastings, K. K., K. J. Frost, M. A. Simpkins, G. W. Pendleton, U. G. Swain, and R. J. Small. 2004. Regional differences in diving behavior of harbor seals in the Gulf of Alaska. Can. J. Zool. 82:1755-1773.

Hoover-Miller, A., S. Atkinson, S. Conlon, J. Prewitt, and P. Armato. 2011. Persistent decline in abundance of harbor seals *Phoca vitulina richardsi* over three decades in Aialik Bay, an Alaskan tidewater glacial fjord. Mar. Ecol. Progr. Ser. 424:259-271.

Jansen, J. K., P. L. Boveng, S. P. Dahle, and J. L. Bengtson. 2010. Reaction of harbor seals to cruise ships. J. Wildl. Manage. 74:1186-1194. DOI: dx.doi.org/10.2193/2008-192 .

Jansen, J. K., P. L. Boveng, J. M. Ver Hoef, S. P. Dahle, and J. L. Bengtson. 2015. Natural and human effects on harbor seal abundance and spatial distribution in an Alaskan glacial fjord. Mar. Mammal Sci. 31(1):66-89.

Jeffries, S., H. Huber, J. Calambokidis, and J. Laake. 2003. Trends and status of harbor seals in Washington State: 1978-1999. J. Wildl. Manage. 67:207-218.

Jemison, L. A., G. W. Pendleton, C. A. Wilson, and R. J. Small. 2006. Long-term trends in harbor seal numbers at Tugidak Island and Nanvak Bay, Alaska. Mar. Mammal Sci. 22:339-360.

Lowry, L. F., K. J. Frost, J. M. Ver Hoef, and R. A. DeLong. 2001. Movements of satellite-tagged subadult and adult harbor seals in Prince William Sound, Alaska. Mar. Mammal Sci. 17:835-861.

Mathews, E. A. 1995. Long-term trends in abundance of harbor seals (*Phoca vitulina richardsi*) and development of monitoring methods in Glacier Bay National Park, Southeast Alaska, p. 254-263. *In* D. R. Engstrom (ed.), Proceedings of the Third Glacier Bay Science Symposium, Gustavus, Alaska. U.S. National Park Service, Glacier Bay National Park and Preserve, Gustavus, AK.

Mathews, E. A., and G. W. Pendleton. 2006. Declines in harbor seal (*Phoca vitulina*) numbers in Glacier Bay National Park, Alaska, 1992-2002. Mar. Mammal Sci. 22:167-189.

Mathews, E. A., L. A. Jemison, G. W. Pendleton, K. M. Blejwas, K. E. Hood, and K. L. Raum-Suryan. 2016. Haul-out patterns and effects of vessel disturbance on harbor seals (*Phoca vitulina*) on glacial ice in Tracy Arm, Alaska. Fish. Bull., U.S. 114(2):186-202.

Molnia, B. F. 2007. Late nineteenth to early twenty-first century behavior of Alaskan glaciers as indicators of changing regional climate. Global Planet. Change 56:23-56.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2019.

O'Corry-Crowe, G. 2012. Population structure of Cook Inlet harbor seals revealed by mitochondrial DNA and microsatellite analysis. Final report to the National Marine Mammal Laboratory, Population Biology and Behavioral Ecology Program, Harbor Branch Oceanographic Institute, Florida Atlantic University: 11.

O'Corry-Crowe, G. M., K. K. Martien, and B. L. Taylor. 2003. The analysis of population genetic structure in Alaskan harbor seals, *Phoca vitulina*, as a framework for the identification of management stocks. Southwest Fisheries Science Center Admin. Rep. LJ-03-08. 54 p.

Olesiuk, P. F., M. A. Bigg, and G. M. Ellis. 1990. Recent trends in the abundance of harbor seals, *Phoca vitulina*, in British Columbia. Can. J. Fish. Aquat. Sci. 47:992-1003.

Pitcher, K. W. 1990. Major decline in number of harbor seals, *Phoca vitulina richardsi*, on Tugidak Island, Gulf of Alaska. Mar. Mammal Sci. 6:121-134.

Pitcher, K. W., and D. C. McAllister. 1981. Movements and haulout behavior of radio-tagged harbor seals, *Phoca vitulina*. Can. Field-Nat. 95:292-297.

Scheffer, V. B., and J. W. Slipp. 1944. The harbor seal in Washington State. Am. Midland Nat. 32:373-416.

Small, R. J., G. W. Pendleton, and K. W. Pitcher. 2003. Trends in abundance of Alaska harbor seals, 1983-2001. Mar. Mammal Sci. 19:344-362.

Small, R. J., L. F. Lowry, J. M. Ver Hoef, K. J. Frost, R. A. DeLong, and M. J. Rehberg. 2005. Differential movements by harbor seal pups in contrasting Alaska environments. Mar. Mammal Sci. 21:671-694.

Small, R. J., P. L. Boveng, V. G. Byrd, and D. E. Withrow. 2008. Harbor seal population decline in the Aleutian archipelago. Mar. Mammal Sci. 24:845-863.

Streveler, G. P. 1979. Distribution, population ecology and impact susceptibility of the harbor seal in Glacier Bay, Alaska. U.S. National Park Service Final Report. 49 p.

Swain, U., J. Lewis, G. Pendleton, and K. Pitcher. 1996. Movements, haul-out, and diving behaviour of harbor seals in Southeast Alaska and Kodiak Island, p. 59-144. *In* Annual Report: harbor seal investigations in Alaska, NOAA Grant NA57FX0367. Division of Wildlife Conservation, Alaska Department of Fish and Game, Douglas, AK.

Ver Hoef, J. M., and K. J. Frost. 2003. A Bayesian hierarchical model for monitoring harbor seal changes in Prince William Sound, Alaska. Environ. Ecol. Stat. 10(2):201-219.

Westlake, R. L., and G. M. O'Corry-Crowe. 2002. Macrogeographic structure and patterns of genetic diversity in harbor seals (*Phoca vitulina*) from Alaska to Japan. J. Mammal. 83:1111-1126.

Wolfe, R. J., J. A. Fall, and R. T. Stanek. 2005. The subsistence harvest of harbor seals and sea lions by Alaska Natives in 2004. Alaska Department of Fish and Game Division of Subsistence, Technical Paper No. 303, Juneau, AK.

Wolfe, R. J., J. A. Fall, and R. T. Stanek. 2006. The subsistence harvest of harbor seals and sea lions by Alaska Natives in 2005. Alaska Department of Fish and Game Division of Subsistence, Technical Paper No. 319, Juneau, AK.

Wolfe, R. J., J. A. Fall, and M. Riedel. 2008. The subsistence harvest of harbor seals and sea lions by Alaska Natives in 2006. Alaska Native Harbor Seal Commission and Alaska Department of Fish and Game Division of Subsistence, Technical Paper No. 339, Juneau, AK. 91 p.

Wolfe, R. J., J. A. Fall, and M. Riedel. 2009a. The subsistence harvest of harbor seals and sea lions by Alaska Natives in 2007. Alaska Native Harbor Seal Commission and Alaska Department of Fish and Game Division of Subsistence, Technical Paper No. 345, Juneau, AK. 95 p.

Wolfe, R. J., J. A. Fall, and M. Riedel. 2009b. The subsistence harvest of harbor seals and sea lions by Alaska Natives in 2008. Alaska Native Harbor Seal Commission and Alaska Department of Fish and Game Division of Subsistence, Technical Paper No. 347, Juneau, AK. 93 p.

Wolfe, R. J., L. Hutchinson-Scarbrough, and M. Riedel. 2012. The subsistence harvest of harbor seals and sea lions on Kodiak Island in 2011. Alaska Native Harbor Seal Commission and Alaska Department of Fish and Game Division of Subsistence, Technical Paper No. 374. 54 p.

Wolfe, R. J., J. Bryant, L. Hutchinson-Scarbrough, M. Kookesh, and L.A. Still. 2013. The subsistence harvest of harbor seals and sea lions in Southeast Alaska in 2012. Alaska Native Harbor Seal Commission and Alaska Department of Fish and Game Division of Subsistence, Technical Paper No. 383. 79 p.

Womble, J. N. 2012. Foraging ecology, diving behavior, and migration patterns of harbor seals (*Phoca vitulina richardii*) from a glacial fjord in Alaska in relation to prey availability and oceanographic features. Ph.D. Dissertation, Oregon State University.

Womble, J. N., and S. M. Gende. 2013. Post-breeding season migrations of a top predator, the harbor seal (*Phoca vitulina richardii*), from a marine protected area in Alaska. PLoS ONE 8(2):e55386.

Womble, J. N., G. W. Pendleton, E. A. Mathews, G. M. Blundell, N. M. Bool, and S. M. Gende. 2010. Harbor seal (*Phoca vitulina richardii*) decline continues in the rapidly changing landscape of Glacier Bay National Park, Alaska 1992–2008. Mar. Mammal Sci. 26:686-697.

Wynne, K. M., D. Hicks, and N. Munro. 1991. 1990 salmon gillnet fisheries observer programs in Prince William Sound and South Unimak Alaska. Annual Report NMFS/NOAA Contract 50ABNF000036. 65 p. Available from NMFS Alaska Region, Office of Marine Mammals, P.O. Box 21668, Juneau, AK 99802.

Wynne, K. M., D. Hicks, and N. Munro. 1992. 1991 marine mammal observer program for the salmon driftnet fishery of Prince William Sound Alaska. Annual Report NMFS/NOAA Contract 50ABNF000036. 53 p. Available from NMFS Alaska Region, Office of Marine Mammals, P.O. Box 21668, Juneau, AK 99802.

SPOTTED SEAL (*Phoca largha*): Bering Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Spotted seals are distributed along the continental shelf of the Bering, Chukchi, and Beaufort seas, and the Sea of Okhotsk south to the western Sea of Japan and northern Yellow Sea (Fig. 1). Eight main areas of spotted seal breeding have been reported (Shaughnessy and Fay 1977). On the basis of small samples and preliminary analyses of genetic composition, potential geographic barriers, and significance of breeding groups, Boveng et al. (2009) grouped those breeding areas into three Distinct Population Segments (DPSs): the Bering DPS, which includes breeding areas in the Bering Sea and portions of the East Siberian, Chukchi, and Beaufort seas that may be occupied outside the breeding period; the Okhotsk DPS; and the Southern DPS, which includes spotted seals breeding in the Yellow Sea and Peter the Great Bay in the Sea of Japan. The Bering stock of spotted seals is defined as the Bering DPS. This stock assessment considers only the portion of the stock found within U.S. waters bounded by the U.S. Exclusive Economic Zone (EEZ; Fig. 1), because the relevant stock assessment data on abundance and human-caused mortality and serious injury are generally not available for the broader range of the stock or even for waters adjacent to the U.S. EEZ.

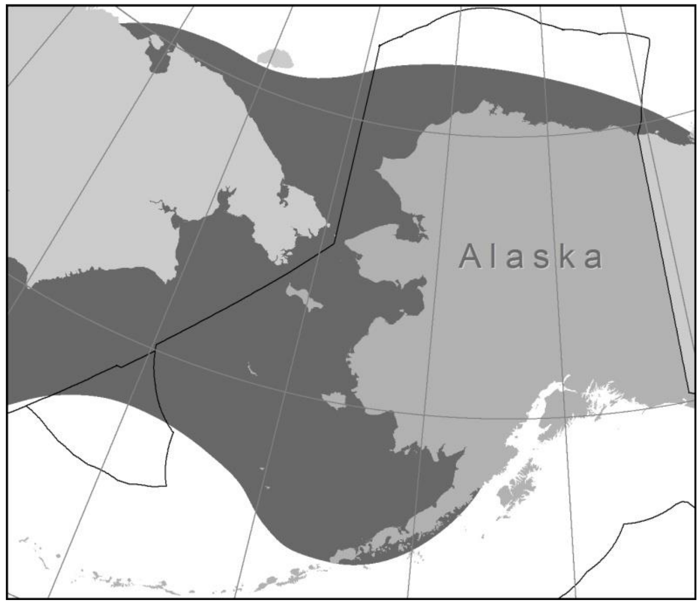

Figure 1. Approximate distribution of spotted seals in the Bering stock (dark shaded area), which is defined as the Bering DPS. This stock assessment considers only the portion of the stock occurring within U.S. waters (i.e., the U.S. Exclusive Economic Zone delineated by a black line).

The distribution of spotted seals is seasonally related to specific life-history events that can be broadly divided into two periods: late-fall through spring, when whelping, nursing, breeding, and molting occur in association with the presence of sea ice on which the seals haul out, and summer through fall when seasonal sea ice has melted and most spotted seals use land for hauling out (Boveng et al. 2009, Citta et al. 2018). Satellite-tagging studies showed that seals tagged in the northeastern Chukchi Sea moved south in October and passed through the Bering Strait in November. Seals overwintered in the Bering Sea along the ice edge and made east-west movements along the edge (Lowry et al. 1998). During spring they tend to prefer small floes (i.e., <20 m in diameter), and inhabit mainly the southern margin of the ice in areas where water depth does not exceed 200 m, and move to coastal habitats after molting and the retreat of the sea ice (Fay 1974, Shaughnessy and Fay 1977, Lowry et al. 2000, Simpkins et al. 2003). In summer and fall, spotted seals use coastal haul-out sites regularly (Frost et al. 1993, Lowry et al. 1998) and may be found as far north as 69-72°N in the Chukchi and Beaufort seas (Porsild 1945, Shaughnessy and Fay 1977). To the south, along the west coast of Alaska, spotted seals are known to occur around the Pribilof Islands, Bristol Bay, and the eastern Aleutian Islands. Spotted seals are closely related to, and often mistaken for, Pacific harbor seals (*Phoca vitulina richardii*). The two species are often seen together and are partially sympatric, as their ranges overlap in the southern part of the Bering Sea (Quakenbush 1988). Yet, spotted seals breed earlier and are less social during the breeding season, and only spotted seals are strongly associated with pack ice (Shaughnessy and Fay 1977). These and other ecological, behavioral, genetic, and morphological differences support their recognition as two separate species (Quakenbush 1988, O'Corry-Crowe and Westlake 1997, Berta and Churchill 2012).

POPULATION SIZE

In the spring of 2012 and 2013, U.S. and Russian researchers conducted aerial abundance and distribution surveys over the entire ice-covered portions of the Bering Sea (Moreland et al. 2013). Conn et al. (2014), using a sub-sample of the data collected from the U.S. portion of the Bering Sea in 2012, calculated an abundance estimate of 461,625 spotted seals (95% CI: 388,732-560,348) in those waters. Although this is a preliminary abundance estimate it is also the best available and it is a reasonable estimate for the entire portion of the Bering spotted seal stock in U.S. waters because relatively few spotted seals are expected north of the Bering Strait during the surveys.

Minimum Population Estimate

The minimum population estimate (N_{MIN}) for a stock is usually calculated using Equation 1 from the potential biological removal (PBR) guidelines (NMFS 2016): $N_{MIN} = N/\exp(0.842\times[\ln(1+[CV(N)]^2)]^{½})$, which approximates the 20th percentile of a distribution that is assumed to be log-normal. However, the abundance estimate based on Conn et al. (2014) was calculated using a Bayesian hierarchical framework, so we used the 20th percentile of the posterior distribution of abundance estimates as a more direct estimator of N_{MIN} than Equation 1 to provide an N_{MIN} of 423,237 spotted seals in the U.S. Bering Sea in the spring.

Current Population Trend

Reliable data on trends in population abundance for the Bering stock of spotted seals or the portion of the stock within U.S. waters are not available.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate (R_{MAX}) is not available for the Bering stock of spotted seals or for any portion of the stock within U.S. waters. Until additional data become available, the default pinniped maximum theoretical net productivity rate of 12% will be used for this stock (NMFS 2016).

POTENTIAL BIOLOGICAL REMOVAL

PBR is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: PBR = $N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 1.0, a value that may be used for stocks that are not known to be decreasing and are taken primarily by aboriginal subsistence hunters, provided there have not been recent increases in the levels of takes (NMFS 2016). Using the N_{MIN} based on Conn et al. (2014) for spotted seals in the U.S. portion of the stock, the PBR is 25,394 seals (423,237 × 0.06 × 1.0).

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2014 and 2018 is listed, by marine mammal stock, in Young et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports. The minimum estimated mean annual level of human-caused mortality and serious injury for the portion of the Bering spotted seal stock in U.S. waters between 2014 and 2018 is 5,254 seals: 1 in U.S. commercial fisheries, 0.4 incidental to Marine Mammal Protection Act (MMPA)-authorized research, and 5,253 in the Alaska Native subsistence harvest (average statewide harvest, including struck and lost animals, in 2015, based on a recently published analysis (Nelson et al. 2019) that is higher and likely more accurate than previous estimates but also revealed stable or decreasing trends in harvest numbers; see below). However, the total mortality and serious injury due to commercial fisheries is unknown because some of the reported harbor seal takes in U.S. commercial fisheries may actually have been spotted seals (since it is virtually impossible to distinguish between these two species without genetic analysis), and there have been no observer programs in nearshore Bristol Bay fisheries that are known to interact with spotted seals. Additional potential threats most likely to result in direct human-caused mortality or serious injury of this stock include the increased potential for oil spills due to an increase in vessel traffic in Alaska waters (with changes in sea-ice coverage).

Fisheries Information

Information for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is available in Appendix 3 of the Alaska Stock Assessment Reports (observer coverage) and in the NMFS List of Fisheries (LOF) and the fact sheets linked to fishery names in the LOF (observer coverage and reported incidental takes of marine mammals: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-list-fisheries, accessed December 2020).

Between 2014 and 2018, incidental mortality and serious injury of spotted seals in U.S. waters occurred in one of the federally-managed U.S. commercial fisheries in Alaska monitored for incidental mortality and serious injury by fisheries observers: the Bering Sea/Aleutian Islands flatfish trawl fishery (Table 1; Breiwick 2013; MML, unpubl. data). This resulted in a minimum estimated mean annual mortality and serious injury rate of one spotted seal incidental to U.S. commercial fisheries between 2014 and 2018, based exclusively on observer data.

Mortality and serious injury of harbor seals incidental to U.S. commercial fisheries occurred between 2014 and 2018 and, because it is virtually impossible to distinguish between harbor seals and spotted seals without genetic analysis, some of the reported harbor seal takes may actually have been spotted seals. Further, there have been no observer programs on nearshore Bristol Bay fisheries that are known to interact with spotted seals, making the total mortality and serious injury due to fisheries unknown.

Table 1. Summary of incidental mortality and serious injury of Bering spotted seals in U.S. waters due to U.S. commercial fisheries between 2014 and 2018 and calculation of the mean annual mortality and serious injury rate (Breiwick 2013; MML, unpubl. data). Methods for calculating percent observer coverage are described in Appendix 3 of the Alaska Stock Assessment Reports.

Fishery name	**Years**	**Data type**	**Percent observer coverage**	**Observed mortality**	**Estimated mortality (CV)**	**Mean estimated annual mortality**
Bering Sea/Aleutian Is. flatfish trawl	2014 2015 2016 2017 2018	obs data	100 100 99 100 100	0 2 1 2 0	0 2 (0.03) 1 (0.05) 2 (0.03) 0	1 (CV = 0.02)
Minimum total estimated annual mortality						1 (CV = 0.02)

Alaska Native Subsistence/Harvest Information

NMFS signed an agreement with the Ice Seal Committee (ISC; 2006) to co-manage Alaska ice seal populations. This co-management agreement promotes full and equal participation by Alaska Natives in decisions affecting the subsistence management of ice seals (to the maximum extent allowed by law) as a tool for conserving ice seal populations in Alaska (https://www.fisheries.noaa.gov/alaska/marine-mammal-protection/co-management-marine-mammals-alaska, accessed December 2020).

Spotted seals are an important resource for Alaska Native subsistence hunters. Approximately 64 coastal communities in Alaska, from Bristol Bay to the Beaufort Sea, harvest ice seals (ISC 2019). The ISC, as co-managers with NMFS, recognizes the importance of harvest information and has collected it since 2008. Annual household survey results compiled in a statewide harvest report include historical ice seal harvest information from 1960 to 2017 (Quakenbush et al. 2009, ISC 2019). To estimate the recent subsistence harvest of ice seals, Nelson et al. (2019) used ice seal harvest survey data collected from 1992 to 2014 for 41 of 55 communities that regularly hunt ice seals, as well as the per capita removal estimates (based on the 2015 human population) from the surveyed communities, to estimate the average regional and statewide subsistence harvest (Table 2). The best statewide estimate of the average number of spotted seals harvested in 2015, including struck and lost animals, is 5,253 seals (Nelson et al. 2019). The authors also found stable or decreasing trends in the annual numbers of ice seals harvested (Nelson et al. 2019).

Table 2. Average regional and statewide subsistence harvest (including struck and lost animals) of Bering spotted seals in 2015 (Nelson et al. 2019). See Figure 1 in Nelson et al. (2019) for a list of the communities in each region.

Region	Average harvest (including struck and lost animals)
North Slope Borough	89
Maniilaq	507
Kawerak	3,175
Association of Village Council Presidents	1,205
Bristol Bay Native Association	277
Statewide total	5,253

Other Mortality

Mortality and serious injury may occasionally occur incidental to marine mammal research activities authorized under MMPA permits issued to a variety of government, academic, and other research organizations. Between 2014 and 2018, there were two reports of mortality incidental to research on the Bering stock of spotted seals (one each in 2014 and 2016), resulting in a mean annual mortality and serious injury rate of 0.4 spotted seals from this stock (Table 3; Young et al. 2020).

In 2011, NMFS and the U.S. Fish and Wildlife Service declared an Unusual Mortality Event (UME) for pinnipeds in the Bering and Chukchi seas, due to the unusual number of sick or dead seals and walruses discovered with skin lesions, bald patches, and other symptoms. The UME occurred from 1 May 2011 to 31 December 2016 and primarily affected ice seals, including ringed seals, bearded seals, ribbon seals, and spotted seals. The investigation concluded that the skin and hair symptoms were signs of a molt abnormality; however, no infectious disease agent or environmental cause for the UME symptoms and mortality was identified (https://www.fisheries.noaa.gov/national/marine-life-distress/active-and-closed-unusual-mortality-events, accessed December 2020). Patchy baldness and delayed molt, however, continue to be observed in limited numbers (<20 per year) of harvested and beachcast ringed seals, bearded seals, ribbon seals, and spotted seals in Alaska.

Since 1 June 2018, elevated numbers of ice seal strandings have occurred in the Bering and Chukchi seas in Alaska and NMFS declared a UME for bearded seals, ringed seals, and spotted seals from 1 June 2018 to present in the Bering and Chukchi seas (https://www.fisheries.noaa.gov/national/marine-life-distress/active-and-closed-unusual-mortality-events, accessed December 2020). As of 31 July 2020, 298 ice seal strandings of all age classes have been reported, including 88 bearded seals, 72 ringed seals, 49 spotted seals, and 89 unidentified seals. A subset of seals has been sampled for genetics and harmful algal bloom exposure and a few have had histopathology samples collected.

Table 3. Summary of mortality and serious injury of Bering spotted seals in U.S. waters, by year and type, reported to the NMFS Office of Protected Resources between 2014 and 2018 (Young et al. 2020).

Cause of injury	2014	2015	2016	2017	2018	Mean annual mortality
Incidental to MMPA-authorized research	1	0	1	0	0	0.4
Total incidental to MMPA-authorized research						0.4

STATUS OF STOCK

The Bering spotted seal stock is not designated as depleted under the MMPA or listed as threatened or endangered under the Endangered Species Act (ESA). NMFS completed a comprehensive status review of the spotted seal under the ESA in 2009 (Boveng et al. 2009) and concluded that listing the Bering DPS of spotted seals, which corresponds to the Bering stock of spotted seals, was not warranted at that time (73 FR 51615, 20 October 2009). The Bering stock of spotted seals is not considered a strategic stock. The best estimate of the mean annual level of human-caused mortality and serious injury in the portion of the stock in U.S. waters is 5,254 spotted seals, which is less than the PBR (25,394 seals). The minimum estimated mean annual rate of U.S. commercial fishery-related mortality and serious injury (one seal) is less than 10% of the PBR (10% of PBR = 2,539) and, therefore, can be considered insignificant and approaching a zero mortality and serious injury rate. Population trends and status of this stock relative to its Optimum Sustainable Population are unknown.

There are key uncertainties in the assessment of the Bering stock of spotted seals. The 2012 Bering Sea abundance estimate by Conn et al. (2014) was calculated using only a sub-sample of the survey data and may be biased. Further, the sample size available for genetics analysis was small so there could be additional stock structure within the Bering stock. Nearshore commercial fisheries are not observed, and fishery-related mortality and serious injury in these fisheries could occur undetected. Based on the best available information, spotted seals are likely to be moderately sensitive to climate change.

HABITAT CONCERNS

The main concern about the conservation status of spotted seals is long-term habitat loss and modification resulting from climate change (Boveng et al. 2009). Laidre et al. (2008) concluded that on a worldwide basis spotted seals were likely to be moderately sensitive to climate change, based on an analysis of various life-history features that could be affected by climate. Climate models consistently project substantial reductions in both the extent and timing of sea ice within the range of spotted seals in Alaska waters; however, the sea ice in the Bering Sea is expected to continue forming annually in winter for the foreseeable future. Spotted seals are associated with sea ice during the periods of reproduction and molting. The presence of sea ice is considered a requirement for whelping and nursing young, providing a platform out of the water to facilitate these life-history events. Similarly, the molt is believed to be promoted by elevated skin temperatures that, in polar regions, can only be achieved when seals haul out of the water. There will likely be more frequent years in which ice coverage is reduced, resulting in a decline in the long-term average ice extent, but Bering Sea spotted seals will likely continue to encounter sufficient ice to support adequate vital rates. Even if sea ice were to vanish completely from the Bering Sea, there may be prospects for spotted seals to adjust their breeding grounds to follow the northward shift of the annual ice front into the Chukchi Sea.

A second major concern, driven primarily by the production of carbon dioxide (CO_2) emissions, is the modification of habitat by ocean acidification, which may alter prey populations and other important aspects of the marine ecosystem. Ocean acidification, a result of increased CO_2 in the atmosphere, may affect spotted seal survival and recruitment through disruption of trophic regimes that are dependent on calcifying organisms. The nature and timing of such impacts are extremely uncertain. As described in Boveng et al. (2009), changes in spotted seal prey, anticipated in response to ocean warming and loss of sea ice, have the potential for negative impacts, but the possibilities are complex. Ecosystem responses may have very long lags as they propagate through trophic webs. Because of spotted seals' apparent dietary flexibility, this threat should be of less immediate concern than the direct effects of sea-ice degradation.

Additional habitat concerns include the potential effects from increased shipping (particularly in the Bering Strait), such as disturbance from vessel traffic and the potential for oil spills.

CITATIONS

Berta, A., and M. Churchill. 2012. Pinniped taxonomy: review of currently recognized species and subspecies, and evidence used for their description. Mammal Rev. 42(3):207-234.

Boveng, P. L., J. L. Bengtson, T. W. Buckley, M. F. Cameron, S. P. Dahle, B. P. Kelly, B. A. Megrey, J. E. Overland, and N. J. Williamson. 2009. Status review of the spotted seal (*Phoca largha*). U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-200, 153 p.

Breiwick, J. M. 2013. North Pacific marine mammal bycatch estimation methodology and results, 2007-2011. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-260, 40 p.

Citta, J. J., L. F. Lowry, L. T. Quakenbush, B. P. Kelly, A. S. Fischbach, J. M. London, C. V. Jay, K. J. Frost, G. O'Corry-Crowe, J. A. Crawford, P. L. Boveng, M. Cameron, A. L. Von Duyke, M. Nelson, L. A. Harwood, P. Richard, R. Suydam, M. P. Heide-Jørgensen, R. C. Hobbs, D. I. Litovka, M. Marcoux, A. Whiting, A. S. Kennedy, J. C. George, J. Orr, and T. Gray. 2018. A multi-species synthesis of satellite telemetry data in the Pacific Arctic (1987–2015): overlap of marine mammal distributions and core use areas. Deep-Sea Res. II 152:132-153 . DOI: dx.doi.org/10.1016/j.dsr2.2018.02.006 .

Conn, P. B., J. M. Ver Hoef, B. T. McClintock, E. E. Moreland, J. M. London, M. F. Cameron, S. P. Dahle, and P. L. Boveng. 2014. Estimating multispecies abundance using automated detection systems: ice-associated seals in the Bering Sea. Methods Ecol. Evol. 5:1280-1293. DOI: dx.doi.org/10.1111/2041-210X.12127 .

Fay, F. H. 1974. The role of ice in the ecology of marine mammals of the Bering Sea, p. 383-399. *In* D. W. Hood and E. J. Kelley (eds.), Oceanography of the Bering Sea. University of Alaska Fairbanks, Institute of Marine Science, Occasional Publication 2.

Frost, K. J., L. F. Lowry, and G. Carroll. 1993. Beluga whale and spotted seal use of a coastal lagoon system in the northeastern Chukchi Sea. Arctic 46:8-16.

Ice Seal Committee (ISC). 2019. The subsistence harvest of ice seals in Alaska – a compilation of existing information, 1960-2017. 86 p. Available online: http://www.north-slope.org/departments/wildlife-management/co-management-organizations/ice-seal-committee . Accessed December 2020.

Laidre, K. L., I. Stirling, L. Lowry, Ø. Wiig, M. P. Heide-Jørgensen, and S. Ferguson. 2008. Quantifying the sensitivity of arctic marine mammals to climate-induced habitat change. Ecol. Appl. 18(2):S97-S125.

Lowry, L. F., K. J. Frost, R. Davis, D. P. DeMaster, and R. S. Suydam. 1998. Movements and behavior of satellite-tagged spotted seals (*Phoca largha*) in the Bering and Chukchi Seas. Polar Biol. 19:221-230.

Lowry, L. F., V. N. Burkanov, K. J. Frost, M. A. Simpkins, A. Springer, D. P. DeMaster, and R. Suydam. 2000. Habitat use and habitat selection by spotted seals (*Phoca largha*) in the Bering Sea. Can. J. Zool. 78:1959-1971.

Moreland, E., M. Cameron, and P. Boveng. 2013. Bering Okhotsk Seal Surveys (BOSS), joint U.S.-Russian aerial surveys for ice-associated seals, 2012-13. Alaska Fisheries Science Center Quarterly Report (July-August-September 2013):1-6.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2020.

Nelson, M. A., L. T. Quakenbush, B. D. Taras, and Ice Seal Committee. 2019. Subsistence harvest of ringed, bearded, spotted, and ribbon seals in Alaska is sustainable. Endang. Species Res. 40:1-16. DOI: dx.doi.org/10.3354/esr00973 .

O'Corry-Crowe, G. M., and R. L. Westlake. 1997. Molecular investigations of spotted seals (*Phoca larhga*) and harbor seals (*P. vitulina*), and their relationships in areas of sympatry, p. 291-304. *In* A. E. Dizon, S. J. Chivers, and W. F. Perrin (eds.), Molecular Genetics of Marine Mammals. The Society for Marine Mammalogy, Spec. Publ. 3.

Porsild, A. E. 1945. Mammals of the Mackenzie Delta. Can. Field-Nat. 59:4-22.

Quakenbush, L. T. 1988. Spotted seal, *Phoca largha*, p. 107-124. *In* J. W. Lentfer (ed.), Selected Marine Mammals of Alaska: Species Accounts with Research and Management Recommendations. Marine Mammal Commission, Washington, DC.

Quakenbush, L., J. Citta, and J. Crawford. 2009. Biology of the spotted seal (*Phoca largha*) in Alaska from 1962 to 2008. Report to NMFS. Arctic Marine Mammal Program, Alaska Department of Fish and Game, Fairbanks, AK. 66 p.

Shaughnessy, P. D., and F. H. Fay. 1977. A review of the taxonomy and nomenclature of North Pacific harbour seals. J. Zool. (Lond.) 182:385-419.

Simpkins, M. A., L. M. Hiruki-Raring, G. Sheffield, J. M. Grebmeier, and J. L. Bengtson. 2003. Habitat selection by ice-associated pinnipeds near St. Lawrence Island, Alaska in March 2001. Polar Biol. 26:577-586.

Young, N. C., B. J. Delean, V. T. Helker, J. C. Freed, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2014-2018. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-413, 142 p.

BEARDED SEAL (*Erignathus barbatus nauticus*): Beringia Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Bearded seals are a boreoarctic species with a circumpolar distribution (Fedoseev 1965; Johnson et al. 1966; Burns 1967, 1981; Burns and Frost 1979; Smith 1981; Kelly 1988). Their normal range extends from the Arctic Ocean (85°N) south to Sakhalin Island (45°N) in the Pacific Ocean and south to Hudson Bay (55°N) in the Atlantic Ocean (Allen 1880, Ognev 1935, King 1983). Bearded seals inhabit the seasonally ice-covered seas of the Northern Hemisphere, where they whelp and rear their pups and molt their coats on the ice in the spring and early summer. Bearded seals feed primarily on benthic organisms, including epifaunal and infaunal invertebrates, and demersal fishes and are closely linked to areas where the seafloor is shallow (less than 200 m).

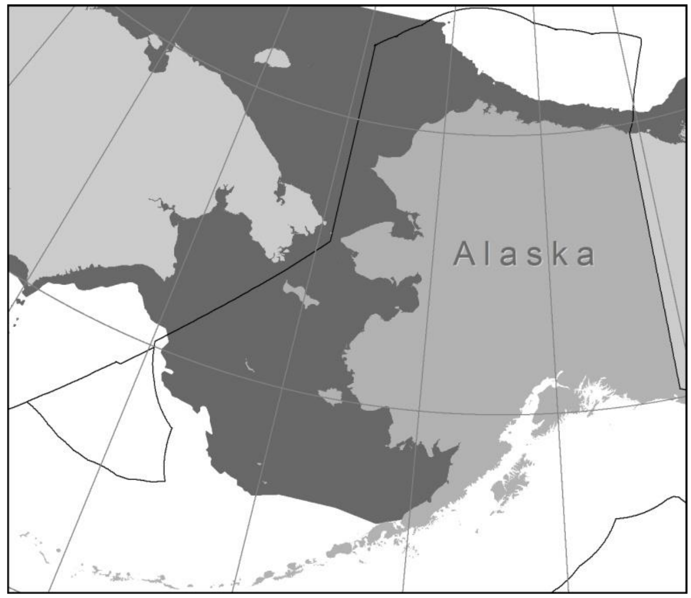

Figure 1. The Beringia bearded seal stock is defined as the Beringia DPS of the *E. B. nauticus* subspecies (dark shaded area). This stock assessment considers only the portion of the stock occurring in U.S. waters (i.e., the U.S. Exclusive Economic Zone delineated by a black line).

Two subspecies have been described: *Erignathus barbatus barbatus* from the Laptev Sea, Barents Sea, North Atlantic Ocean, and Hudson Bay (Rice 1998); and *E. b. nauticus* from the remaining portions of the Arctic Ocean, the Bering Sea, and the Sea of Okhotsk (Ognev 1935, Scheffer 1958, Manning 1974, Heptner et al. 1976). The geographic distributions of these subspecies are not separated by conspicuous gaps, and there are regions of intergrading generally described as somewhere along the northern Russian and central Canadian coasts. NMFS defined longitude 145°E as the Eurasian delineation between the two subspecies and 130°W in western Canada as the North American delineation between the two subspecies (Cameron et al. 2010; 77 FR 76740, 28 December 2012). Based on evidence for discreteness and ecological uniqueness of bearded seals in the Sea of Okhotsk, under the Endangered Species Act (ESA) the *E. b. nauticus* subspecies was further divided into an Okhotsk Distinct Population Segment (DPS) and a Beringia DPS (77 FR 76740), so named because the continental shelf waters of the Bering, Chukchi, Beaufort, and East Siberian seas that are the bearded seals' range in this region overlie much of the land bridge that was exposed during the last glaciation, which has been referred to as Beringia. This stock is defined as the Beringia DPS; however, this stock assessment considers only the portion of the Beringia stock found within U.S. waters bounded by the U.S. Exclusive Economic Zone (EEZ; Fig. 1), because the relevant stock assessment data on abundance and human-caused mortality and serious injury are generally not available for the broader range of the stock or even for waters adjacent to the U.S. EEZ.

Spring surveys conducted in 1999 and 2000 along the Alaska coast indicate that bearded seals are typically more abundant 20-100 nautical miles (nmi) from shore than within 20 nmi from shore, except for high concentrations nearshore to the south of Kivalina (Bengtson et al. 2000, 2005; Simpkins et al. 2003). Many seals that winter in the Bering Sea move north through the Bering Strait from late April through June and spend the summer in the Chukchi Sea (Burns 1967, 1981). Bearded seal sounds (produced by adult males) have been recorded nearly year-round (peak occurrence in December-June, when sea-ice concentrations were >50%) at multiple locations in the Bering, Chukchi, and Beaufort seas, and calling behavior is closely related to the presence of sea ice (MacIntyre et al. 2013, 2015; Jimbo et al. 2019). The overall summer distribution is quite broad, with seals rarely hauled out on land, and some seals, mostly juveniles, may not follow the ice northward but remain near the coasts of the Bering and Chukchi seas (Burns 1967, 1981; Heptner et al. 1976; Nelson 1981; Cameron et al. 2018). As the ice forms again in the fall and winter, most seals move south with the advancing ice edge through the Bering Strait into the Bering Sea where they spend the winter (Burns and Frost 1979; Frost et al. 2005, 2008; Cameron and Boveng 2007, 2009; Breed et al. 2018; Cameron et al. 2018). This southward migration is less noticeable and predictable than the northward movements in late spring and early summer (Burns and Frost 1979, Burns 1981, Kelly 1988). During winter, the central and northern parts of the Bering Sea shelf have the highest densities of bearded seals (Fay 1974, Heptner et al. 1976, Burns and Frost 1979, Braham et al. 1981, Burns 1981, Nelson et al. 1984, Citta et al. 2018). In late winter and early spring, bearded seals are widely, but not uniformly, distributed in the broken, drifting pack ice ranging from the Chukchi Sea to the ice front in the Bering Sea. In these areas, they tend to avoid the coasts and areas of fast ice (Burns 1967, Burns and Frost 1979).

POPULATION SIZE

Although a reliable population estimate for the entire stock is not available, survey methods have been developed and applied to substantial portions of the stock's range in U.S. waters. In the spring of 2012 and 2013, U.S. and Russian researchers conducted aerial abundance and distribution surveys over the entire ice-covered portions of the Bering Sea (Moreland et al. 2013). Conn et al. (2014), using a sub-sample of the data collected from the U.S. portion of the Bering Sea in 2012, calculated an abundance estimate of 301,836 bearded seals (95% CI: 238,195-371,147) in those waters. Researchers expect to provide a population estimate for the entire U.S. portion of the bearded seal stock once the final Bering Sea results are combined with the results from spring surveys of the Chukchi Sea (conducted in 2016) and Beaufort Sea (planned for 2021).

Minimum Population Estimate

A minimum population estimate (N_{MIN}) for the entire U.S. portion of the stock cannot be determined because reliable abundance estimates are not yet available for the Chukchi and Beaufort seas. Using the 2012 Bering Sea density estimate by Conn et al. (2014), however, we are able to calculate an N_{MIN} of 273,676 bearded seals in the U.S. Bering Sea. The N_{MIN} for a stock is usually calculated using Equation 1 from the potential biological removal (PBR) guidelines (NMFS 2016): $N_{MIN} = N/\exp(0.842 \times [\ln(1+[CV(N)]^2)]^{½})$, which approximates the 20th percentile of a distribution that is assumed to be log-normal. However, the abundance estimate based on Conn et al. (2014) was calculated using a Bayesian hierarchical framework, so we used the 20th percentile of the posterior distribution of abundance estimates as a more direct estimator of N_{MIN} than Equation 1. This N_{MIN} is negatively biased as an estimator of the Beringia bearded seal stock, and even the U.S. portion of the stock, because the estimate is based solely on the Bering Sea and, therefore, doesn't include the many bearded seals that inhabit the Chukchi and Beaufort seas (e.g., Bengtson et al. 2005, Laidre et al. 2015).

Current Population Trend

Reliable data on trends in population abundance for the Beringia stock of bearded seals or the portion of the stock within U.S. waters are not available.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate (R_{MAX}) is not available for the Beringia stock of bearded seals or any portion of the stock within U.S. waters. Until additional data become available, the default pinniped maximum theoretical net productivity rate of 12% will be used for this stock (NMFS 2016).

POTENTIAL BIOLOGICAL REMOVAL

PBR is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: PBR = $N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 0.5, the value for pinniped stocks listed as threatened under the ESA (NMFS 2016). Using the negatively biased N_{MIN} for bearded seals in the U.S. portion of the Beringia stock, PBR is 8,210 seals (273,676 × 0.06 × 0.5). This PBR is negatively biased because of its dependence on the negatively biased N_{MIN} estimate.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2014 and 2018 is listed, by marine mammal stock, in Young et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports. The minimum estimated mean annual level of human-caused mortality and serious injury for the portion of the Beringia bearded seal stock in U.S. waters between 2014 and 2018 is 6,709 seals: 1.8 in U.S. commercial fisheries, 6,707 in the Alaska Native subsistence harvest (average statewide harvest, including struck and lost animals, in 2015, based on a recently published analysis (Nelson et al. 2019) that is higher and likely more accurate than previous estimates but also revealed stable or decreasing trends in harvest numbers; see below), and 0.4 due to Marine Mammal Protection Act (MMPA)-authorized research-related permanent removals from the population. Additional potential threats most likely to result in direct human-caused mortality or serious injury of this stock include the increased potential for oil spills due to an increase in vessel traffic in Alaska waters (with changes in sea-ice coverage).

Fisheries Information

Information for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is available in Appendix 3 of the Alaska Stock Assessment Reports (observer coverage) and in the NMFS List of Fisheries (LOF) and the fact sheets linked to fishery names in the LOF (observer coverage and reported incidental takes of marine mammals: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-list-fisheries, accessed December 2020).

Between 2014 and 2018, incidental mortality and serious injury of bearded seals in U.S. waters occurred in two of the federally-managed U.S. commercial fisheries in Alaska monitored for incidental mortality and serious injury by fisheries observers: the Bering Sea/Aleutian Islands pollock trawl and Bering Sea/Aleutian Islands flatfish trawl fisheries (Table 1; Breiwick 2013; MML, unpubl. data). The minimum estimated mean annual mortality and serious injury rate incidental to U.S. commercial fisheries between 2014 and 2018 is 1.8 bearded seals, based exclusively on observer data.

Table 1. Summary of incidental mortality and serious injury of Beringia bearded seals in U.S. waters due to U.S. commercial fisheries between 2014 and 2018 and calculation of the mean annual mortality and serious injury rate (Breiwick 2013; MML, unpubl. data). Methods for calculating percent observer coverage are described in Appendix 3 of the Alaska Stock Assessment Reports.

Fishery name	Years	Data type	Percent observer coverage	Observed mortality	Estimated mortality (CV)	Mean estimated annual mortality
Bering Sea/Aleutian Is. pollock trawl	2014 2015 2016 2017 2018	obs data	98 99 99 99 99	1 0 0 1 0	1.0 (0.14) 0 0 1.0 (0.1) 0	0.4 (CV = 0.09)
Bering Sea/Aleutian Is. pollock trawl	2016	obs data	99	1*	N/A	0.2 (CV = N/A)
Bering Sea/Aleutian Is. flatfish trawl	2014 2015 2016 2017 2018	obs data	100 100 99 100 100	1 2 1 1 1	1 (0.05) 2 (0.03) 1 (0.05) 1 (0.04) 1 (0.05)	1.2 (CV = 0.02)
Minimum total estimated annual mortality						1.8 (CV = 0.03)

*This seal was discovered during a vessel offload. Because it could not be associated with a haul number, it was not included in the bycatch estimate for the fishery.

Alaska Native Subsistence/Harvest Information

NMFS signed an agreement with the Ice Seal Committee (ISC; 2006) to co-manage Alaska ice seal populations. This co-management agreement promotes full and equal participation by Alaska Natives in decisions affecting the subsistence management of ice seals (to the maximum extent allowed by law) as a tool for conserving ice seal populations in Alaska (https://www.fisheries.noaa.gov/alaska/marine-mammal-protection/co-management-marine-mammals-alaska, accessed December 2020).

Bearded seals are an important resource for Alaska Native subsistence hunters. Approximately 64 coastal communities in Alaska, from Bristol Bay to the Beaufort Sea, harvest ice seals (ISC 2019). The ISC, as co-managers with NMFS, recognizes the importance of harvest information and has collected it since 2008. Annual household survey results compiled in a statewide harvest report include historical ice seal harvest information from 1960 to 2017 (Quakenbush et al. 2011, ISC 2019).To estimate the recent subsistence harvest of ice seals, Nelson et al. (2019) used ice seal harvest survey data collected from 1992 to 2014 for 41 of 55 communities that regularly hunt ice seals, as well as the per capita removal estimates (based on the 2015 human population) from the surveyed communities, to estimate the average regional and statewide subsistence harvest (Table 2). The best statewide estimate of the average number of bearded seals harvested in 2015, including struck and lost animals, is 6,707 seals (Nelson et al. 2019). The authors also found stable or decreasing trends in the annual numbers of ice seals harvested (Nelson et al. 2019).

Table 2. Average regional and statewide subsistence harvest (including struck and lost animals) of Beringia bearded seals in 2015 (Nelson et al. 2019). See Figure 1 in Nelson et al. (2019) for a list of the communities in each region.

Region	Average harvest (including struck and lost animals)
North Slope Borough	1,031
Maniilaq	1,038
Kawerak	3,248
Association of Village Council Presidents	1,360
Bristol Bay Native Association	30
Statewide total	6,707

Other Mortality

Permanent removals from the population may occasionally occur during marine mammal research activities authorized under MMPA permits issued to a variety of government, academic, and other research organizations. Between 2014 and 2018, two research-related permanent removals (one seal each in 2014 and 2015) were reported for the Beringia stock of bearded seals (Young et al. 2020; Table 3), resulting in a mean annual rate of 0.4 bearded seals.

In 2011, NMFS and the U.S. Fish and Wildlife Service declared an Unusual Mortality Event (UME) for pinnipeds in the Bering and Chukchi seas, due to the unusual number of sick or dead seals and walruses discovered with skin lesions, bald patches, and other symptoms. The UME occurred from 1 May 2011 to 31 December 2016 and primarily affected ice seals, including ringed seals, bearded seals, ribbon seals, and spotted seals. The investigation concluded that the skin and hair symptoms were signs of a molt abnormality; however, no infectious disease agent or environmental cause for the UME symptoms and mortality was identified (https://www.fisheries.noaa.gov/national/marine-life-distress/active-and-closed-unusual-mortality-events, accessed December 2020). Patchy baldness and delayed molt, however, continue to be observed in limited numbers (<20 per year) of harvested and beachcast ringed seals, bearded seals, ribbon seals, and spotted seals in Alaska.

Since 1 June 2018, elevated numbers of ice seal strandings have occurred in the Bering and Chukchi seas in Alaska and NMFS declared a UME for bearded seals, ringed seals, and spotted seals from 1 June 2018 to present in the Bering and Chukchi seas (https://www.fisheries.noaa.gov/national/marine-life-distress/active-and-closed-unusual-mortality-events, accessed December 2020). As of 31 July 2020, 298 ice seal strandings of all age classes have been reported, including 88 bearded seals, 72 ringed seals, 49 spotted seals, and 89 unidentified seals. A subset of seals has been sampled for genetics and harmful algal bloom exposure and a few have had histopathology samples collected.

Table 3. Summary of mortality and serious injury of Beringia bearded seals, by year and type, reported to the NMFS Alaska Region marine mammal stranding network and NMFS Office of Protected Resources between 2014 and 2018 (Young et al. 2020).

Cause of Injury	2014	2015	2016	2017	2018	Mean annual mortality
MMPA-authorized research-related permanent removals	1	1	0	0	0	0.4
Total MMPA-authorized research-related permanent removals						0.4

STATUS OF STOCK

On 28 December 2012, NMFS listed the Beringia DPS bearded seal (*E. b. nauticus*), which corresponds to the Beringia stock of bearded seals, as threatened under the ESA (77 FR 76740). The primary concern for this population is the ongoing and projected loss of sea-ice cover resulting from climate change, which is expected to pose a significant threat to the persistence of these seals in the foreseeable future (based on projections through the end of the 21st century: Cameron et al. 2010). Because of its threatened status under the ESA, this stock is designated as depleted under the MMPA and is classified as a strategic stock. The best estimate of the mean annual level of human-caused mortality and serious injury in the portion of the stock in U.S. waters is 6,709 bearded seals, which is less than the negatively biased PBR of 8,210 seals. The minimum estimated mean annual rate of U.S. commercial fishery-related mortality and serious injury (1.8 seals) is less than 10% of the PBR (10% of PBR = 821) and, therefore, can be considered insignificant and approaching a zero mortality and serious injury rate. Population trends and status of this stock relative to its Optimum Sustainable Population are unknown.

There are key uncertainties in the assessment of the Beringia stock of bearded seals. Abundance and mortality and serious injury estimates are not available for the vast majority of the stock's range. Within U.S. waters, where abundance estimates are being developed and data are currently available on mortality and serious injury in commercial fisheries and the Alaska Native subsistence harvest, key abundance estimates for the Beaufort and Chukchi seas are not yet available. The negatively biased N_{MIN} used here, based on a 2012 Bering Sea density estimate from Conn et al. (2014), was calculated using only a sub-sample of the data and may be biased as an estimate for the U.S. waters of the Bering Sea. Also, it represents just a portion of the population of bearded seals in U.S. waters and is, therefore, not very reliable for comparison with mortality and serious injury numbers for the entire U.S. portion of the stock. Based on the best available information, bearded seals are likely to be highly sensitive to climate change.

HABITAT CONCERNS

The main concern about the conservation status of bearded seals is long-term habitat loss and modification resulting from climate change (77 FR 76740, 28 December 2012). Laidre et al. (2008) concluded that on a worldwide basis bearded seals were likely to be highly sensitive to climate change, based on an analysis of various life-history features that could be affected by climate. Climate models consistently project substantial reductions in both the extent and timing of sea ice within the range of bearded seals in Alaska waters (Cameron et al. 2010). Bearded seals are closely associated with sea ice, particularly during the periods of reproduction and molting. The presence of sea ice is considered a requirement for whelping and nursing young. Similarly, the molt is believed to be promoted by elevated skin temperatures that, in polar regions, can only be achieved when seals haul out of the water. If suitable ice cover is absent from shallow feeding areas during times of peak whelping and nursing (April/May) or molting (May/June and sometimes through August), bearded seals would be forced to seek either sea-ice habitat over deeper waters (perhaps with poor access to food) or onshore haul-out sites (perhaps with increased risks of disturbance, predation, and competition). Both scenarios would require bearded seals to adapt to novel (i.e., potentially suboptimal) conditions and to exploit habitats to which they may not be well adapted, likely compromising their reproduction and survival rates.

A second major concern, driven primarily by the production of carbon dioxide (CO_2) emissions, is the modification of habitat by ocean acidification, which may alter prey populations and other important aspects of the marine ecosystem. Ocean acidification, a result of increased CO_2 in the atmosphere, may affect bearded seal survival and recruitment through disruption of trophic regimes that are dependent on calcifying organisms. The nature and timing of such impacts are extremely uncertain. As discussed in Cameron et al. (2010), changes in bearded seal prey, anticipated in response to ocean warming and loss of sea ice, have the potential for negative impacts, but the possibilities are complex. Ecosystem responses may have very long lags as they propagate through trophic webs. Because of bearded seals' apparent dietary flexibility, this threat may be of less immediate concern than the threats from sea-ice degradation.

Additional habitat concerns include the potential effects from increased shipping (particularly in the Bering Strait), such as disturbance from vessel traffic and the potential for oil spills.

CITATIONS

Allen, J. A. 1880. History of North American Pinnipeds: A Monograph of the Walruses, Sea-lions, Sea-bears and Seals of North America. U.S. Department of the Interior, U.S. Government Printing Office, Washington, D.C. 785 p.

Bengtson, J. L., P. L. Boveng, L. M. Hiruki-Raring, K. L. Laidre, C. Pungowiyi, and M. A. Simpkins. 2000. Abundance and distribution of ringed seals (*Phoca hispida*) in the coastal Chukchi Sea, p. 149-160. *In* A. L. Lopez and D. P. DeMaster (eds.), Marine Mammal Protection Act and Endangered Species Act Implementation Program 1999. AFSC Processed Rep. 2000-11, Alaska Fisheries Science Center, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Bengtson, J. L., L. M. Hiruki-Raring, M. A. Simpkins, and P. L. Boveng. 2005. Ringed and bearded seal densities in the eastern Chukchi Sea, 1999-2000. Polar Biol. 28:833-845.

Braham, H. W., J. J. Burns, G. A. Fedoseev, and B. D. Krogman. 1981. Distribution and density of ice-associated pinnipeds in the Bering Sea. Available from Marine Mammal Laboratory, AFSC, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115. 27 p.

Breed, G. A., M. F. Cameron, J. M. Ver Hoef, P. L. Boveng, A. Whiting, and K. J. Frost. 2018. Seasonal sea ice dynamics drive movement and migration of juvenile bearded seals *Erignathus barbatus*. Mar. Ecol. Progr. Ser. 600:223-237. DOI: dx.doi.org/10.3354/meps12659 .

Breiwick, J. M. 2013. North Pacific marine mammal bycatch estimation methodology and results, 2007-2011. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-260, 40 p.

Burns, J. J. 1967. The Pacific bearded seal. Alaska Department of Fish and Game, Pittman-Robertson Project Report W-6-R and W-14-R. 66 p.

Burns, J. J. 1981. Bearded seal-*Erignathus barbatus* Erxleben, 1777, p. 145-170. *In* S. H. Ridgway and R. J. Harrison (eds.), Handbook of Marine Mammals. Vol. 2. Seals. Academic Press, New York.

Burns, J. J., and K. J. Frost. 1979. The natural history and ecology of the bearded seal, *Erignathus barbatus*. Alaska Department of Fish and Game. 77 p.

Cameron, M., and P. Boveng. 2007. Abundance and distribution surveys for ice seals aboard the USCG *Healy* and the *Oscar Dyson*, 10 April - 18 June 2007. Alaska Fisheries Science Center Quarterly Report (April-May-June 2007):12-14.

Cameron, M., and P. Boveng. 2009. Habitat use and seasonal movements of adult and sub-adult bearded seals. Alaska Fisheries Science Center Quarterly Report (October-November-December 2009):1-4.

Cameron, M. F., J. L. Bengtson, P. L. Boveng, J. K. Jansen, B. P. Kelly, S. P. Dahle, E. A. Logerwell, J. E. Overland, C. L. Sabine, G. T. Waring, and J. M. Wilder. 2010. Status review of the bearded seal (*Erignathus barbatus*). U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-211, 246 p.

Cameron, M. F., K. J. Frost, J. M. Ver Hoef, G. A. Breed, A. V. Whiting, J. Goodwin, and P. L. Boveng. 2018. Habitat selection and seasonal movements of young bearded seals (*Erignathus barbatus*) in the Bering Sea. PLoS ONE 13(2):e0192743. DOI: dx.doi.org/10.1371/journal.pone.0192743 .

Citta, J. J., L. F. Lowry, L. T. Quakenbush, B. P. Kelly, A. S. Fischbach, J. M. London, C. V. Jay, K. J. Frost, G. O'Corry-Crowe, J. A. Crawford, P. L. Boveng, M. Cameron, A. L. Von Duyke, M. Nelson, L. A. Harwood, P. Richard, R. Suydam, M. P. Heide-Jørgensen, R. C. Hobbs, D. I. Litovka, M. Marcoux, A. Whiting, A. S. Kennedy, J. C. George, J. Orr, and T. Gray. 2018. A multi-species synthesis of satellite telemetry data in the Pacific Arctic (1987–2015): overlap of marine mammal distributions and core use areas. Deep-Sea Res. II 152:132-153 . DOI: dx.doi.org/10.1016/j.dsr2.2018.02.006 .

Conn, P. B., J. M. Ver Hoef, B. T. McClintock, E. E. Moreland, J. M. London, M. F. Cameron, S. P. Dahle, and P. L. Boveng. 2014. Estimating multispecies abundance using automated detection systems: ice-associated seals in the Bering Sea. Methods Ecol. Evol. 5:1280-1293. DOI: dx.doi.org/10.1111/2041-210X.12127 .

Fay, F. H. 1974. The role of ice in the ecology of marine mammals of the Bering Sea, p. 383-399. *In* D. W. Hood and E. J. Kelley (eds.), Oceanography of the Bering Sea. University of Alaska, Fairbanks, Institute of Marine Science, Occasional Publication 2.

Fedoseev, G. A. 1965. The ecology of the reproduction of seals on the northern part of the Sea of Okhotsk. Izvestiya TINRO 65:212-216. (Translated from Russian by the Fisheries and Marine Service, Quebec, Canada, Translation Series No. 3369. 8 p.)

Frost, K. J., M. F. Cameron, M. Simpkins, C. Schaeffer, and A. Whiting. 2005. Diving behavior, habitat use, and movements of bearded seal (*Erignathus barbatus*) pups in Kotzebue Sound and Chukchi Sea, p. 98-99. *In* Proceedings of the Sixteenth Biennial Conference on the Biology of Marine Mammals, San Diego, CA.

Frost, K. J., A. Whiting, M. F. Cameron, and M. A. Simpkins. 2008. Habitat use, seasonal movements and stock structure of bearded seals in Kotzebue Sound, Alaska. Tribal Wildlife Grants Program, Fish and Wildlife Service, Tribal Wildlife Grants Study U-4-IT. Final Report from the Native Village of Kotzebue, Kotzebue, AK, for U.S. Fish and Wildlife Service, Anchorage, AK. 16 p.

Heptner, L. V. G., K. K. Chapskii, V. A. Arsen'ev, and V. T. Sokolov. 1976. Bearded seal. *Erignathus barbatus* (Erxleben, 1777), p. 166-217. *In* L. V. G. Heptner, N. P. Naumov, and J. Mead (eds.), Mammals of the Soviet Union. Volume II, Part 3--Pinnipeds and Toothed Whales, Pinnipedia and Odontoceti. Vysshaya Shkola Publishers, Moscow, Russia. (Translated from Russian by P. M. Rao, 1996, Science Publishers, Inc., Lebanon, NH.)

Ice Seal Committee (ISC). 2019. The subsistence harvest of ice seals in Alaska – a compilation of existing information, 1960-2017. 86 p. Available online: http://www.north-slope.org/departments/wildlife-management/co-management-organizations/ice-seal-committee . Accessed December 2020.

Jimbo, M., D. Mizuguchi, H. Shirakawa, K. Tsujii, A. Fujiwara, K. Miyashita, and Y. Mitani. 2019. Seasonal variations in the call presence of bearded seals in relation to sea ice in the southern Chukchi Sea. Polar Biol. 42:1953. DOI: dx.doi.org/10.1007/s00300-019-02569-2 .

Johnson, M. L., C. H. Fiscus, B. T. Stenson, and M. L. Barbour. 1966. Marine mammals, p. 877-924. *In* N. J. Wilimovsky and J. N. Wolfe (eds.), Environment of the Cape Thompson Region, Alaska. U.S. Atomic Energy Commission, Oak Ridge, TN.

Kelly, B. P. 1988. Bearded seal, *Erignathus barbatus*, p. 77-94. *In* J. W. Lentfer (ed.), Selected Marine Mammals of Alaska: Species Accounts with Research and Management Recommendations. Marine Mammal Commission, Washington, DC.

King, J. E. 1983. Seals of the World. 2nd edition. British Museum (Natural History) and Oxford University Press, London, UK. 240 p.

Laidre, K. L., I. Stirling, L. Lowry, Ø. Wiig, M. P. Heide-Jørgensen, and S. Ferguson. 2008. Quantifying the sensitivity of arctic marine mammals to climate-induced habitat change. Ecol. Appl. 18(2):S97-S125.

Laidre, K., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724-737.

MacIntyre, K. Q., K. M. Stafford, C. L. Berchok, and P. L. Boveng. 2013. Year-round acoustic detection of bearded seals (*Erignathus barbatus*) in the Beaufort Sea relative to changing environmental conditions, 2008-2010. Polar Biol. 36(8):1161-1173.

MacIntyre, K. Q., K. M. Stafford, P. B. Conn, K. L. Laidre, and P. L. Boveng. 2015. The relationship between sea ice concentration and the spatio-temporal distribution of vocalizing bearded seals (*Erignathus barbatus*) in the Bering, Chukchi, and Beaufort seas from 2008 to 2011. Prog. Oceanogr. 136:241-249. DOI: dx.doi.org/10.1016/j.pocean.2015.05.008 .

Manning, T. H. 1974. Variation in the skull of the bearded seal, *Erignathus barbatus* (Erxleben). Biological Papers of the University of Alaska 16:1-21.

Moreland, E., M. Cameron, and P. Boveng. 2013. Bering Okhotsk Seal Surveys (BOSS), joint U.S.-Russian aerial surveys for ice-associated seals, 2012-13. Alaska Fisheries Science Center Quarterly Report (July-August-September 2013):1-6.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2020.

Nelson, R. K. 1981. Harvest of the sea: coastal subsistence in modern Wainwright. North Slope Borough, Barrow, AK. 125 p.

Nelson, R. R., J. J. Burns, and K. J. Frost. 1984. The bearded seal (*Erignathus barbatus*), p. 1-6. *In* J. J. Burns (ed.), Marine Mammal Species Accounts. Wildlife Technical Bulletin No. 7. Alaska Department of Fish and Game, Juneau, AK.

Nelson, M. A., L. T. Quakenbush, B. D. Taras, and Ice Seal Committee. 2019. Subsistence harvest of ringed, bearded, spotted, and ribbon seals in Alaska is sustainable. Endang. Species Res. 40:1-16. DOI: dx.doi.org/10.3354/esr00973 .

Ognev, S. I. 1935. Mammals of the U.S.S.R. and Adjacent Countries. Vol. 3. Carnivora (Fissipedia and Pinnipedia). Gosudarst. Izdat. Biol. Med. Lit., Moscow. (Translated from Russian by Israel Program for Scientific Translations, 1962. 741 p.)

Quakenbush, L., J. Citta, and J. Crawford. 2011. Biology of the bearded seal (*Erignathus barbatus*) in Alaska, 1961–2009. Final Report to NMFS. Arctic Marine Mammal Program, Alaska Department of Fish and Game, Fairbanks, AK. 71 p.

Rice, D. W. 1998. Marine Mammals of the World: Systematics and Distribution. Soc. Mar. Mammal. Spec. Publ. No. 4.

Scheffer, V. B. 1958. Seals, Sea Lions and Walruses: A Review of the Pinnipedia. Stanford University Press, Palo Alto, CA. 179 p.

Simpkins, M. A., L. M. Hiruki-Raring, G. Sheffield, J. M. Grebmeier, and J. L. Bengtson. 2003. Habitat selection by ice-associated pinnipeds near St. Lawrence Island, Alaska in March 2001. Polar Biol. 26:577-586.

Smith, T. G. 1981. Notes on the bearded seal, *Erignathus barbatus*, in the Canadian Arctic. Department of Fisheries and Oceans, Arctic Biological Station, Can. Tech. Rep. Fish. Aquat. Sci. No. 1042. 49 p.

Young, N. C., B. J. Delean, V. T. Helker, J. C. Freed, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2014-2018. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-413, 142 p.

RINGED SEAL (*Pusa hispida hispida*): Arctic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Ringed seals (*Pusa hispida*) have a circumpolar distribution and are found in all seasonally ice-covered seas of the Northern Hemisphere as well as in certain freshwater lakes (King 1983). Most taxonomists currently recognize five subspecies of ringed seals: *P. h. hispida* in the Arctic Ocean and Bering Sea; *P. h. ochotensis* in the Sea of Okhotsk and northern Sea of Japan; *P. h. botnica* in the northern Baltic Sea; *P. h. lagodensis* in Lake Ladoga, Russia; and *P. h. saimensis* in Lake Saimaa, Finland. Morphologically, the Baltic and Okhotsk subspecies are fairly well differentiated from the Arctic subspecies (Ognev 1935, Müller-Wille 1969, Rice 1998) and the Ladoga and Saimaa subspecies differ significantly from each other and from the Baltic subspecies (Müller-Wille 1969, Hyvärinen and Nieminen 1990, Amano et al. 2002). Genetic analyses support isolation of the lake-inhabiting populations (Palo 2003, Palo et al. 2003, Valtonen et al. 2012). Lack of differentiation between the Baltic and the Arctic subspecies may reflect recurrent gene flow (Martinez-Bakker et al. 2013) but is more likely due to retention of high diversity within the relatively large effective population size of the Baltic subspecies since separation from the Arctic subspecies (Nyman et al. 2014). Widespread mixing within the Arctic subspecies is the likely explanation for its high diversity and apparent lack of population structure (Palo et al. 2001, Davis et al. 2008, Kelly et al. 2009, Martinez-Bakker et al. 2013). Differences in body size, morphology, growth rates, and/or diet between Arctic ringed seals in shorefast versus pack ice have been taken as evidence of separate breeding populations in some locations (McLaren 1958, Fedoseev 1975, Finley et al. 1983). This has not been thoroughly examined, however, and the taxonomic status and population structure of the Arctic subspecies remain unresolved (Berta and Churchill 2012). The stock, therefore, may be as large as the entire *P. h. hispida* subspecies range. This stock assessment considers only the portion of the stock found within U.S. waters bounded by the U.S. Exclusive Economic Zone (EEZ; Fig. 1), because the relevant stock assessment data on abundance and human-caused mortality and serious injury are generally not available for the broader range of the stock or even for waters adjacent to the U.S. EEZ.

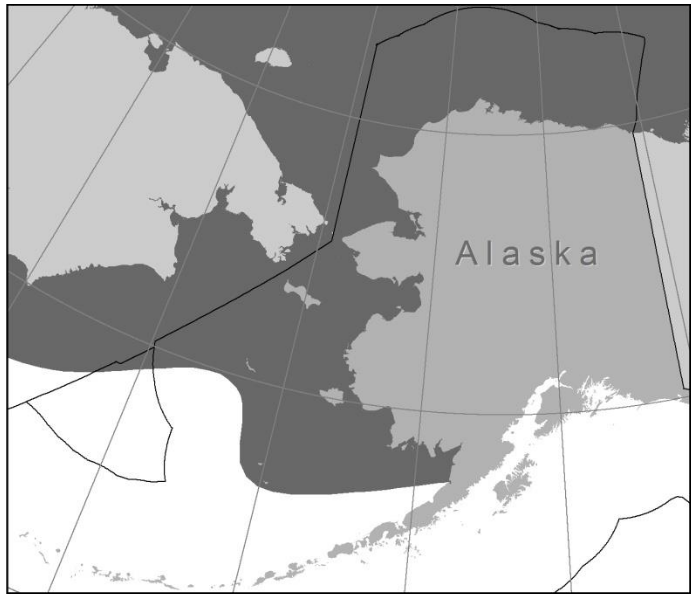

Figure 1. The Arctic ringed seal stock is defined as the population of the Arctic subspecies (*P. h. hispida*). This stock assessment considers only the portion of the stock occurring in U.S. waters (i.e., the U.S. Exclusive Economic Zone delineated by a black line). The dark shaded area shows the approximate winter distribution of the Arctic ringed seal stock around Alaska.

Throughout their range, ringed seals have an affinity for ice-covered waters and are well adapted to occupying both shorefast and pack ice (Kelly 1988). They remain with the ice most of the year and use it as a platform for pupping and nursing in late winter to early spring, for molting in late spring to early summer, and for resting at other times of the year. Arctic ringed seals rarely come ashore in the Arctic, although they have been observed during summer months resting on land in the White Sea (Lukin et al. 2006) and, recently, in a fjord system in Svalbard (Lydersen et al. 2017). In Alaska waters, during winter and early spring when sea ice is at its maximal extent, ringed seals are abundant in the northern Bering Sea, Norton and Kotzebue Sounds, and throughout the Chukchi and Beaufort seas. They occur as far south as Bristol Bay in years of extensive ice coverage but generally are not abundant south of Norton Sound except in nearshore areas (Frost 1985). However, surveys conducted in the Bering Sea in the spring of 2012 and 2013 documented numerous ringed seals in both nearshore and offshore habitat extending south of Norton Sound (79 FR 73010, 9 December 2014). Although details of their seasonal movements have not been adequately documented, most ringed seals that winter in the Bering, Chukchi, and Beaufort seas are thought to migrate north in the spring as the seasonal ice melts and retreats (Burns 1970, Kelly et al. 2010b) and spend summers in the pack ice of the northern Chukchi and Beaufort seas, as well as on nearshore ice remnants in the Beaufort Sea (Frost 1985, Kelly et al. 2010b). During summer, ringed seals range hundreds to thousands of kilometers to forage along ice edges or in highly productive open-water areas (Harwood and Stirling 1992, Freitas et al. 2008, Kelly et al. 2010b, Harwood et al. 2015). With the onset of freeze-up in the fall, ringed seal movements become increasingly restricted. Seals that have summered in the Beaufort Sea are thought to move west and south with the advancing ice pack, with many seals dispersing throughout the Chukchi and Bering seas while some remain in the Beaufort Sea (Frost and Lowry 1984, Crawford et al. 2012, Harwood et al. 2012). Some adult ringed seals return to the same small home ranges they occupied during the previous winter (Kelly et al. 2010b).

POPULATION SIZE

Although a reliable population estimate for the entire stock is not available, survey methods have been developed and applied to substantial portions of the stock's range in U.S. waters. In the spring of 2012 and 2013, U.S. and Russian researchers conducted aerial abundance and distribution surveys over the entire ice-covered portions of the Bering Sea (Moreland et al. 2013). Conn et al. (2014), using a sub-sample of the data collected from the U.S. portion of the Bering Sea in 2012, calculated an abundance estimate of 171,418 ringed seals (95% CI: 141,588-201,090). This estimate did not account for availability bias due to seals in the water at the time of the surveys and did not include ringed seals in the shorefast ice zone, which were surveyed using a different trackline design that will require a separate analysis. Thus, the actual number of ringed seals in the U.S. portion of the Bering Sea is likely much higher, perhaps by a factor of two or more. Researchers expect to provide a population estimate, corrected for availability bias, for the entire U.S. portion of the ringed seal stock once the final Bering Sea results are combined with the results from spring surveys of the Chukchi Sea (conducted in 2016) and Beaufort Sea (planned for 2020).

Minimum Population Estimate

A minimum population estimate (N_{MIN}) for the entire U.S. portion of the stock cannot be determined because reliable abundance estimates are not yet available for the Chukchi and Beaufort seas. Using the 2012 Bering Sea density estimate by Conn et al. (2014), however, we are able to calculate an N_{MIN} of 158,507 ringed seals in the U.S. Bering Sea. The N_{MIN} for a stock is usually calculated using Equation 1 from the potential biological removal (PBR) guidelines (NMFS 2016): $N_{MIN} = N/\exp(0.842\times[\ln(1+[CV(N)]^2)]^{½})$, which approximates the 20th percentile of a distribution that is assumed to be log-normal. However, the abundance estimate based on Conn et al. (2014) was calculated using a Bayesian hierarchical framework, so we used the 20th percentile of the posterior distribution of abundance estimates as a more direct estimator of N_{MIN} than Equation 1. This N_{MIN} is negatively biased as an estimator of the Arctic ringed seal stock, and even the U.S. portion of the stock, because the estimate is based solely on the Bering Sea and, therefore, doesn't include the many ringed seals that inhabit the Chukchi and Beaufort seas (e.g., Kelly et al. 2010a, Laidre et al. 2015) and because the Conn et al. (2014) study did not adjust densities for seals in the water (not detectable by the surveys).

Current Population Trend

Reliable data on trends in population abundance for the Arctic stock of ringed seals or the portion of the stock within U.S. waters are not available.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate (R_{MAX}) is not available for the Arctic stock of ringed seals or any portion of the stock within U.S. waters. Until additional data become available, the default pinniped maximum theoretical net productivity rate of 12% will be used for this stock (NMFS 2016).

POTENTIAL BIOLOGICAL REMOVAL

PBR is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: PBR = $N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 0.5, the value for pinniped stocks listed as threatened under the Endangered Species Act (ESA) (NMFS 2016). Using the negatively biased N_{MIN} for ringed seals in the U.S. portion of the Arctic stock, PBR is 4,755 seals (158,507 × 0.06 × 0.5). This PBR is negatively biased because of its dependence on the negatively biased N_{MIN} estimate.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2014 and 2018 is listed, by marine mammal stock, in Young et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports. The minimum estimated mean annual level of human-caused mortality and serious injury for the portion of the Arctic ringed seal stock in U.S. waters between 2014 and 2018 is 6,459 seals: 5 in U.S. commercial fisheries, 6,454 in the Alaska Native subsistence harvest (average statewide harvest, including struck and lost animals, in 2015, based on a recently published analysis (Nelson et al. 2019) that is higher and likely more accurate than previous estimates but also revealed stable or decreasing trends in harvest numbers; see below), 0.2 in marine debris, and 0.2 incidental to Marine Mammal Protection Act (MMPA)-authorized research. Additional potential threats most likely to result in direct human-caused mortality or serious injury of this stock include the increased potential for oil spills due to an increase in vessel traffic in Alaska waters (with changes in sea-ice coverage).

Fisheries Information

Information for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is available in Appendix 3 of the Alaska Stock Assessment Reports (observer coverage) and in the NMFS List of Fisheries (LOF) and the fact sheets linked to fishery names in the LOF (observer coverage and reported incidental takes of marine mammals: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-list-fisheries, accessed December 2020).

Between 2014 and 2018, incidental mortality and serious injury of ringed seals in U.S. waters was reported in two of the federally-managed U.S. commercial fisheries in Alaska monitored for incidental mortality and serious injury by fisheries observers: the Bering Sea/Aleutian Islands flatfish trawl and Bering Sea/Aleutian Islands pollock trawl fisheries (Table 1; Breiwick 2013; MML, unpubl. data). Based on observer data from 2014 to 2018, the minimum average annual rate of mortality and serious injury incidental to U.S. commercial fishing operations is 4.8 ringed seals.

One ringed seal mortality resulting from entanglement in unidentified commercial gear in U.S. waters was reported to the NMFS Alaska Region marine mammal stranding network in 2017 (Young et al. 2020), resulting in a mean annual mortality and serious injury rate of 0.2 ringed seals between 2014 and 2018 (Table 3). This mortality and serious injury estimate results from an actual count of verified human-caused deaths and serious injuries and is a minimum because not all entangled animals strand nor are all stranded animals found, reported, or have the cause of death determined.

Table 1. Summary of incidental mortality and serious injury of Arctic ringed seals in U.S. waters due to U.S. commercial fisheries between 2014 and 2018 and calculation of the mean annual mortality and serious injury rate (Breiwick 2013; MML, unpubl. data). Methods for calculating percent observer coverage are described in Appendix 3 of the Alaska Stock Assessment Reports.

Fishery name	Years	Data type	Percent observer coverage	Observed mortality	Estimated mortality (CV)	Mean estimated annual mortality
Bering Sea/Aleutian Is. flatfish trawl	2014 2015 2016 2017 2018	obs data	100 100 99 100 100	0 1 0 8 14	0 1 (0.05) 0 8.0 (0.01) 14 (0.02)	4.6 (CV = 0.01)
Bering Sea/Aleutian Is. pollock trawl	2017	obs data	100	1[a]	N/A	0.2 (CV = N/A)
Minimum total estimated annual mortality						4.8 (CV = 0.01)

[a]This seal was discovered during a vessel offload. Because it could not be associated with a haul number, it was not included in the bycatch estimate for the fishery.

Alaska Native Subsistence/Harvest Information

NMFS signed an agreement with the Ice Seal Committee (ISC; 2006) to co-manage Alaska ice seal populations. This co-management agreement promotes full and equal participation by Alaska Natives in decisions affecting the subsistence management of ice seals (to the maximum extent allowed by law) as a tool for conserving ice seal populations in Alaska (https://www.fisheries.noaa.gov/alaska/marine-mammal-protection/co-management-marine-mammals-alaska, accessed December 2020).

Ringed seals are an important resource for Alaska Native subsistence hunters. Approximately 64 coastal communities in Alaska, from Bristol Bay to the Beaufort Sea, harvest ice seals (ISC 2019). The ISC, as co-managers with NMFS, recognizes the importance of harvest information and has collected it since 2008. Annual household survey results compiled in a statewide harvest report include historical ice seal harvest information from 1960 to 2017 (Quakenbush et al. 2011, ISC 2019). To estimate the recent subsistence harvest of ice seals, Nelson et al. (2019) used ice seal harvest survey data collected from 1992 to 2014 for 41 of 55 communities that regularly hunt ice seals, as well as the per capita removal estimates (based on the 2015 human population) from the surveyed communities, to estimate the average regional and statewide subsistence harvest (Table 2). The best statewide estimate of the average number of ringed seals harvested in 2015, including struck and lost animals, is 6,454 seals (Nelson et al. 2019). The authors also found stable or decreasing trends in the annual numbers of ice seals harvested (Nelson et al. 2019).

Table 2. Average regional and statewide subsistence harvest (including struck and lost animals) of Arctic ringed seals in 2015 (Nelson et al. 2019). See Figure 1 in Nelson et al. (2019) for a list of the communities in each region.

Region	**Average harvest (including struck and lost animals)**
North Slope Borough	1,146
Maniilaq	493
Kawerak	2,287
Association of Village Council Presidents	2,484
Bristol Bay Native Association	44
Statewide total	6,454

Other Mortality

Reports to the NMFS Alaska Region marine mammal stranding network of ringed seals entangled in marine debris or with injuries caused by other types of human interaction are another source of mortality and serious injury data. These mortality and serious injury estimates result from an actual count of verified human-caused deaths and serious injuries and are minimums because not all entangled animals strand nor are all stranded animals found, reported, or have the cause of death determined. One ringed seal mortality due to entanglement in marine debris in U.S. waters was reported in 2017, resulting in a mean annual mortality and serious injury rate of 0.2 ringed seals between 2014 and 2018 (Table 3; Young et al. 2020).

Ringed seal mortality due to gunshot wounds reported to the NMFS Alaska Region stranding network (Young et al. 2020) is presumed to be animals struck and lost in the Alaska Native subsistence hunt and, therefore, is not included in the mean annual mortality and serious injury rate for 2014 to 2018.

Mortality and serious injury may occasionally occur incidental to marine mammal research activities authorized under MMPA permits issued to a variety of government, academic, and other research organizations. Between 2014 and 2018, there was one report, in 2016, of a mortality incidental to research on the Arctic stock of ringed seals (Table 3; Young et al. 2020), resulting in a mean annual mortality and serious injury rate of 0.2 ringed seals.

In 2011, NMFS and the U.S. Fish and Wildlife Service declared an Unusual Mortality Event (UME) for pinnipeds in the Bering and Chukchi seas, due to the unusual number of sick or dead seals and walruses discovered with skin lesions, bald patches, and other symptoms. The UME occurred from 1 May 2011 to 31 December 2016 and primarily affected ice seals, including ringed seals, bearded seals, ribbon seals, and spotted seals. The investigation concluded that the skin and hair symptoms were signs of a molt abnormality; however, no infectious disease agent or environmental cause for the UME symptoms and mortality was identified (https://www.fisheries.noaa.gov/national/marine-life-distress/active-and-closed-unusual-mortality-events, accessed December 2020). Patchy baldness and delayed molt, however, continue to be observed in limited numbers (<20 per year) of harvested and beachcast ringed seals, bearded seals, ribbon seals, and spotted seals in Alaska.

Since 1 June 2018, elevated numbers of ice seal strandings have occurred in the Bering and Chukchi seas in Alaska and NMFS declared a UME for bearded seals, ringed seals, and spotted seals from 1 June 2018 to present in the Bering and Chukchi seas (https://www.fisheries.noaa.gov/national/marine-life-distress/active-and-closed-unusual-mortality-events, accessed December 2020). As of 31 July 2020, 298 ice seal strandings of all age classes have been reported, including 88 bearded seals, 72 ringed seals, 49 spotted seals, and 89 unidentified seals. A subset of seals has been sampled for genetics and harmful algal bloom exposure and a few have had histopathology samples collected.

Table 3. Summary of Arctic ringed seal mortality and serious injury in U.S. waters, by year and type, reported to the NMFS Alaska Region marine mammal stranding network and NMFS Office of Protected Resources between 2014 and 2018 (Young et al. 2020). Animals that were disentangled and released with non-serious injuries have been excluded from this table.

Cause of injury	**2014**	**2015**	**2016**	**2017**	**2018**	**Mean annual mortality**
Entangled in unidentified commercial gear	0	0	0	1	0	0.2
Entangled in marine debris	0	0	0	1	0	0.2
Incidental to MMPA-authorized research	0	0	1	0	0	0.2
Total in commercial fisheries						0.2
Total in marine debris						0.2
Total incidental to MMPA-authorized research						0.2

STATUS OF STOCK

On 28 December 2012, NMFS listed the Arctic ringed seal subspecies (*P. h. hispida*), which corresponds to the Arctic stock of ringed seals, as threatened under the ESA (77 FR 76706). The primary concern for this population is the ongoing and anticipated loss of sea ice and snow cover resulting from climate change, which is expected to pose a significant threat to the persistence of these seals in the foreseeable future (based on projections through the end of the 21st century; Kelly et al. 2010a). Because of its threatened status under the ESA, this stock is designated as depleted under the MMPA and is classified as a strategic stock. The best estimate of the mean annual level of human-caused mortality and serious injury in the U.S. waters portion of the stock is 6,459 ringed seals, which is greater than the negatively biased PBR of 4,755 seals. However, because this exceedance of PBR stems from an unrealistically low N_{MIN}, it should not be taken as indicative of a risk to this stock. The PBR was obtained from an N_{MIN} that is known to be an extreme underestimate of the abundance in the U.S. waters of the Bering Sea, which in turn is just a portion of the Arctic ringed seal stock in U.S. waters, and the best estimate of human-caused mortality and serious injury is for the entire U.S. portion of the stock, including, for example, Alaska Native subsistence takes in the Chukchi and Beaufort seas. Previous estimates from the U.S. waters of the Chukchi Sea (Bengtson et al. 2005) and results from a recent (2016) NOAA survey of those waters indicate that there are several hundreds of thousands of ringed seals in that region that are not included in N_{MIN} because the former results are outdated and the latter have not yet been published. Furthermore, ringed seals are known to remain abundant in the U.S. waters of the Beaufort Sea (which are also not included in N_{MIN}) based, for example, on hunter reports to the ISC and NOAA test surveys conducted in 2019. NMFS believes with high confidence that the number of ringed seals in Alaska waters greatly exceeds the number of individuals that would be required for the current take to balance the PBR (i.e., N_{MIN} × Mortality and Serious Injury / PBR = 215,310 individuals). Therefore, the apparent exceedance of PBR in this case reflects inadequacy in the abundance estimates, rather than an indication of excessive take. The minimum estimated mean annual rate of U.S. commercial fishery-related mortality and serious injury (5 seals) is less than 10% of the negatively biased PBR (10% of PBR = 476) and, therefore, can be considered insignificant and approaching a zero mortality and serious injury rate. Population trends and status of this stock relative to its Optimum Sustainable Population are unknown.

There are key uncertainties in the assessment of the Arctic stock of ringed seals. Abundance and mortality and serious injury estimates are not available for the vast majority of the stock's range. Within U.S. waters, where abundance estimates are being developed and data are currently available on mortality and serious injury in commercial fisheries and the Alaska Native subsistence harvest, key abundance estimates for the Beaufort and Chukchi seas are not yet available. The negatively biased N_{MIN} used here, based on a 2012 Bering Sea density estimate from Conn et al. (2014), was calculated using only a sub-sample of the data and is likely to be an underestimate for the U.S. waters of the Bering Sea because of availability bias. Also, it represents just a portion of the population of ringed seals in U.S. waters and is, therefore, not very reliable for comparison with mortality and serious injury numbers for the entire U.S. portion of the stock. Based on the best available information, ringed seals are likely to be highly sensitive to climate change.

HABITAT CONCERNS

The main concern about the conservation status of ringed seals is long-term habitat loss and modification resulting from climate change (77 FR 76706, 28 December 2012). Laidre et al. (2008) concluded that on a worldwide basis ringed seals were likely to be highly sensitive to climate change based on an analysis of various life-history features that could be affected by climate.

Climate models consistently project substantial reductions in sea ice and on-ice snow depths (Kelly et al. 2010a, Hezel et al. 2012). Ringed seals excavate subnivean lairs (snow caves) in drifts over their breathing holes in the ice, in which they rest, give birth, and nurse their pups for 5-9 weeks during late winter and spring (Chapskii 1940, McLaren 1958, Smith and Stirling 1975). Substantial data indicate high pup mortality due to hypothermia and predation as a consequence of inadequate snow cover (e.g., Kumlien 1879, Lukin and Potelov 1978, Lydersen and Smith 1989, Smith and Lydersen 1991, Hammill and Smith 1991, Stirling and Smith 2004). Decreases in ice, and especially on-ice snow depths, are expected to lead to increased juvenile mortality from premature weaning, hypothermia, and predation (Kelly et al. 2010a). Changes in the ringed seal's habitat will be rapid relative to their generation time and, thereby, will limit adaptive responses (Kelly et al. 2010a).

A second major concern, driven primarily by the production of carbon dioxide (CO_2) emissions, is the modification of habitat by ocean acidification, which may alter prey populations and other important aspects of the marine ecosystem. Ocean acidification, a result of increased CO_2 in the atmosphere, may affect ringed seal survival and recruitment through disruption of trophic regimes that are dependent on calcifying organisms. The nature and timing of such impacts are extremely uncertain. As discussed by Kelly et al. (2010a), changes in ringed seal prey, anticipated in response to ocean warming and loss of sea ice, have the potential for negative impacts, but the possibilities are complex. Ecosystem responses may have very long lags as they propagate through trophic webs. Because of ringed seals' apparent dietary flexibility, this threat may be of less immediate concern than the threats from sea-ice degradation.

Additional habitat concerns include the potential effects from increased shipping (particularly in the Bering Strait), such as disturbance from vessel traffic and the potential for oil spills.

CITATIONS

Amano, M., A. Hayano, and N. Miyazaki. 2002. Geographic variation in the skull of the ringed seal, *Pusa hispida*. J. Mammal. 83:370-380.

Bengtson, J. L., L. M. Hiruki-Raring, M. A. Simpkins, and P. L. Boveng. 2005. Ringed and bearded seal densities in the eastern Chukchi Sea, 1999-2000. Polar Biol. 28:833-845.

Berta, A., and M. Churchill. 2012. Pinniped taxonomy: review of currently recognized species and subspecies, and evidence used for their description. Mammal Rev. 42:207-234.

Breiwick, J. M. 2013. North Pacific marine mammal bycatch estimation methodology and results, 2007-2011. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-260, 40 p.

Burns, J. J. 1970. Remarks on the distribution and natural history of pagophilic pinnipeds in the Bering and Chukchi seas. J. Mammal. 51:445-454.

Chapskii, K. K. 1940. The ringed seal of western seas of the Soviet Arctic (The morphological characteristic, biology and hunting production), p. 147. *In* N. A. Smirnov (ed.), Proceedings of the Arctic Scientific Research Institute, Chief Administration of the Northern Sea Route. Izd. Glavsevmorputi, Leningrad, Moscow. (Translated from Russian by the Fisheries Research Board of Canada, Ottawa, Canada, Translation Series No. 1665, 147 p.)

Conn, P. B., J. M. Ver Hoef, B. T. McClintock, E. E. Moreland, J. M. London, M. F. Cameron, S. P. Dahle, and P. L. Boveng. 2014. Estimating multispecies abundance using automated detection systems: ice-associated seals in the Bering Sea. Methods Ecol. Evol. 5:1280-1293. DOI: dx.doi.org/10.1111/2041-210X.12127 .

Crawford, J. A., K. J. Frost, L. T. Quakenbush, and A. Whiting. 2012. Different habitat use strategies by subadult and adult ringed seals (*Phoca hispida*) in the Bering and Chukchi seas. Polar Biol. 35:241-255.

Davis, C. S., I. Stirling, C. Strobeck, and D. W. Coltman. 2008. Population structure of ice-breeding seals. Mol. Biol. 17:3078-3094.

Fedoseev, G. A. 1975. Ecotypes of the ringed seal (*Pusa hispida* Schreber, 1777) and their reproductive capabilities. Biology of the Seal. Proceedings of a Symposium held in Guelph, 14-17 August 1972. Rapports et Proces-verbaux des Réunions. Conseil International pour l'Éxploration de la Mer. 169:156-160.

Finley, K. J., G. W. Miller, R. A. Davis, and W. R. Koski. 1983. A distinctive large breeding population of ringed seals (*Phoca hispida*) inhabiting the Baffin Bay pack ice. Arctic 36:162-173.

Freitas, C., K. M. Kovacs, R. A. Ims, M. A. Fedak, and C. Lydersen. 2008. Ringed seal post-moulting movement tactics and habitat selection. Oecologia 155:193-204.

Frost, K. J. 1985. The ringed seal (*Phoca hispida*), p. 79-87. *In* J. J. Burns, K. J. Frost, and L. F. Lowry (eds.), Marine Mammal Species Accounts. Alaska Department of Fish and Game, Juneau, AK.

Frost, K. J., and L. F. Lowry. 1984. Trophic relationships of vertebrate consumers in the Alaskan Beaufort Sea, p. 381-401. *In* P. W. Barnes, D. M. Schell, and E. Reimnitz (eds.), The Alaskan Beaufort Sea: Ecosystems and Environments. Academic Press, Inc., New York, NY.

Hammill, M. O., and T. G. Smith. 1991. The role of predation in the ecology of the ringed seal in Barrow Strait, Northwest Territories, Canada. Mar. Mammal Sci. 7:123-135.

Harwood, L. A., and I. Stirling. 1992. Distribution of ringed seals in the southeastern Beaufort Sea during late summer. Can. J. Zool. 70(5):891-900.

Harwood, L. A., T. G. Smith, and J. C. Auld. 2012. Fall migration of ringed seals (*Phoca hispida*) through the Beaufort and Chukchi seas, 2001-02. Arctic 65:35-44.

Harwood, L. A., T. G. Smith, J. C. Auld, H. Melling, and D. J. Yurkowski. 2015. Seasonal movements and diving of ringed seals, *Pusa hispida*, in the western Canadian Arctic, 1999-2001 and 2010-11. Arctic 68(2):193-209.

Hezel, P. J., X. Zhang, C. M. Bitz, B. P. Kelly, and F. Massonnet. 2012. Projected decline in spring snow depth on Arctic sea ice caused by progressively later autumn open ocean freeze-up this century. Geophys. Res. Lett. 39:L17505.

Hyvärinen, H., and M. Nieminen. 1990. Differentiation of the ringed seal in the Baltic Sea, Lake Ladoga and Lake Saimaa. Finnish Game Res. 47:21-27.

Ice Seal Committee (ISC). 2019. The subsistence harvest of ice seals in Alaska – a compilation of existing information, 1960-2017. 86 p. Available online: http://www.north-slope.org/departments/wildlife-management/co-management-organizations/ice-seal-committee . Accessed December 2020.

Kelly, B. P. 1988. Ringed seal, *Phoca hispida*, p. 57-75. *In* J. W. Lentfer (ed.), Selected Marine Mammals of Alaska: Species Accounts with Research and Management Recommendations. Marine Mammal Commission, Washington, DC.

Kelly, B. P., M. Ponce, D. A. Tallmon, B. J. Swanson, and S. K. Sell. 2009. Genetic diversity of ringed seals sampled at breeding sites; implications for population structure and sensitivity to sea ice loss. University of Alaska Southeast, North Pacific Research Board 631 Final Report. 28 p.

Kelly, B. P., J. L. Bengtson, P. L. Boveng, M. F. Cameron, S. P. Dahle, J. K. Jansen, E. A. Logerwell, J. E. Overland, C. L. Sabine, G. T. Waring, and J. M. Wilder. 2010a. Status review of the ringed seal (*Phoca hispida*). U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-212, 250 p.

Kelly, B. P., O. H. Badajos, M. Kunnasranta, J. R. Moran, M. Martinez-Bakker, D. Wartzok, and P. Boveng. 2010b. Seasonal home ranges and fidelity to breeding sites among ringed seals. Polar Biol. 33:1095-1109.

King, J. E. 1983. Seals of the World. 2nd edition. British Museum (Natural History), London. 240 p.

Kumlien, L. 1879. Mammals, p. 55-61. *In* Contributions to the Natural History of Arctic America Made in Connection with the Howgate Polar Expedition 1877-78. Government Printing Office, Washington, DC.

Laidre, K. L., I. Stirling, L. F. Lowry, Ø. Wiig, M. P. Heide-Jørgensen, and S. H. Ferguson. 2008. Quantifying the sensitivity of arctic marine mammals to climate-induced habitat change. Ecol. Appl. 18(2):S97-S125.

Laidre, K., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724-737.

Lukin, L. R., and V. A. Potelov. 1978. Living conditions and distribution of ringed seal in the White Sea in the winter. Soviet J. Mar. Biol. 4:684-690.

Lukin, L. P., G. N. Ognetov, and N. S. Boiko. 2006. Ecology of the ringed seal in the White Sea. UrO RAN, Ekaterinburg, Russia. 165 p. (Translated from Russian by the Baltic Fund for Nature (BFN), State University of St. Petersburg, Russia.)

Lydersen, C., and T. G. Smith. 1989. Avian predation on ringed seal *Phoca hispida* pups. Polar Biol. 9:489-490.

Lydersen, C., J. Vaquie-Garcia, E. Lydersen, G. N. Christensen, and K. M. Kovacs. 2017. Novel terrestrial haul-out behaviour by ringed seals (*Pusa hispida*) in Svalbard, in association with harbour seals (*Phoca vitulina*). Polar Res. 36, 1374124. DOI: dx.doi.org/10.1080/17518369.2017.1374124 .

Martinez-Bakker, M. E., S. K. Sell, B. J. Swanson, B. P. Kelly, and D. A. Tallmon. 2013. Combined genetic and telemetry data reveal high rates of gene flow, migration, and long-distance dispersal potential in Arctic ringed seals (*Pusa hispida*). PLoS ONE 8:e77125.

McLaren, I. A. 1958. The biology of the ringed seal (*Phoca hispida* Schreber) in the eastern Canadian Arctic. Bull. Fish. Res. Board Can. 118:97.

Moreland, E., M. Cameron, and P. Boveng. 2013. Bering Okhotsk Seal Surveys (BOSS), joint U.S.-Russian aerial surveys for ice-associated seals, 2012-13. Alaska Fisheries Science Center Quarterly Report (July-August-September 2013):1-6.

Müller-Wille, L. L. 1969. Biometrical comparison of four populations of *Phoca hispida* Schreb. in the Baltic and White seas and lakes Ladoga and Saimaa. Commentationes Biologicae Societas Scientiarum Fennica 31:1-12.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2020.

Nelson, M. A., L. T. Quakenbush, B. D. Taras, and Ice Seal Committee. 2019. Subsistence harvest of ringed, bearded, spotted, and ribbon seals in Alaska is sustainable. Endang. Species Res. 40:1-16. DOI: dx.doi.org/10.3354/esr00973 .

Nyman, T., M. Valtonen, J. Aspi, M. Ruokonen, M. Kunnasranta, and J. U. Palo. 2014. Demographic histories and genetic diversities of Fennoscandian marine and landlocked ringed seal subspecies. Ecol. Evol. 4:3420-3434.

Ognev, S. I. 1935. Mammals of the U.S.S.R. and Adjacent Countries. Vol. 3. Carnivora. Glavpushnina NKVT, Moscow, Russia. 641 p. (Translated from Russian by the Israel Program for Scientific Translations, Jerusalem, Israel. 741 p.)

Palo, J. 2003. Genetic diversity and phylogeography of landlocked seals. Dissertation. University of Helsinki, Helsinki, Finland. 29 p.

Palo, J. U., H. S. Mäkinen, E. Helle, O. Stenman, and R. Väinölä. 2001. Microsatellite variation in ringed seals (*Phoca hispida*): genetic structure and history of the Baltic Sea population. Heredity 86:609-617.

Palo, J. U., H. Hyvärinen, E. Helle, H. S. Mäkinen, and R. Väinölä. 2003. Postglacial loss of microsatellite variation in the landlocked Lake Saimaa ringed seal. Conserv. Genet. 4:117-128.

Quakenbush, L., J. Citta, and J. Crawford. 2011. Biology of the ringed seal (*Phoca hispida*) in Alaska, 1960–2010. Final Report to NMFS. Arctic Marine Mammal Program, Alaska Department of Fish and Game, Fairbanks, AK. 72 p.

Rice, D. W. 1998. Marine Mammals of the World: Systematics and Distribution. Society for Marine Mammalogy, Lawrence, KS. 231 p.

Smith, T. G., and C. Lydersen. 1991. Availability of suitable land-fast ice and predation as factors limiting ringed seal populations, *Phoca hispida*, in Svalbard. Polar Res. 10:585-594.

Smith, T. G., and I. Stirling. 1975. The breeding habitat of the ringed seal (*Phoca hispida*). The birth lair and associated structures. Can. J. Zool. 53:1297-1305.

Stirling, I., and T. G. Smith. 2004. Implications of warm temperatures, and an unusual rain event for the survival of ringed seals on the coast of southeastern Baffin Island. Arctic 57:59-67.

Valtonen, M., J. Palo, M. Ruokonen, M. Kunnasranta, and T. Nyman. 2012. Spatial and temporal variation in genetic diversity of an endangered freshwater seal. Conserv. Genet. 13:1231-1245.

Young, N. C., B. J. Delean, V. T. Helker, J. C. Freed, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2014-2018. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-413, 142 p.

RIBBON SEAL (*Histriophoca fasciata*)

STOCK DEFINITION AND GEOGRAPHIC RANGE

Ribbon seals inhabit the North Pacific Ocean and adjacent parts of the Arctic Ocean. In Alaska waters, ribbon seals range from the North Pacific Ocean and Bering Sea into the Chukchi and western Beaufort seas (Fig. 1). Ribbon seals are very rarely seen on shorefast ice or land. From late March to early May, ribbon seals inhabit the Bering Sea ice front (Burns 1970, 1981; Braham et al. 1984). They are most abundant in the northern part of the ice front in the central and western parts of the Bering Sea (Burns 1970, Burns et al. 1981). As the ice recedes in May to mid-July, the seals move farther north in the Bering Sea, where they haul out on the receding ice edge and remnant ice (Burns 1970, 1981; Burns et al. 1981). As the ice melts, seals become more concentrated, with at least part of the Bering Sea population moving to the Bering Strait and the southern part of the Chukchi Sea. Ten ribbon seals satellite tagged in the spring of 2005 near the eastern coast of Kamchatka spent the summer and fall throughout the Bering Sea (Boveng et al. 2013). However, of 72 ribbon seals satellite tagged in the central Bering Sea from 2007 to 2010, 21 seals (29%) moved to the Bering Strait, Chukchi Sea, or Arctic Basin as the ice retreated northward, while the other 51 tagged seals did not pass north of the Bering Strait (Boveng et al. 2013). Passive acoustic sampling detected ribbon seal calls in August to early/mid-November in the Chukchi Sea and on the Chukchi Plateau (Moore et al. 2012, Hannay et al. 2013, Jones et al. 2014, Frouin-Mouy et al. 2019), as well as in the western Beaufort Sea in September to early November (Frouin-Mouy et al. 2019), similarly indicating presence of some ribbon seals north of the Bering Strait during summer and fall. The 72 seals tagged in the central Bering Sea and the 10 seals tagged near Kamchatka dispersed widely, occupying coastal areas as well as the middle of the Bering Sea, both on and off the continental shelf (Boveng et al. 2013).

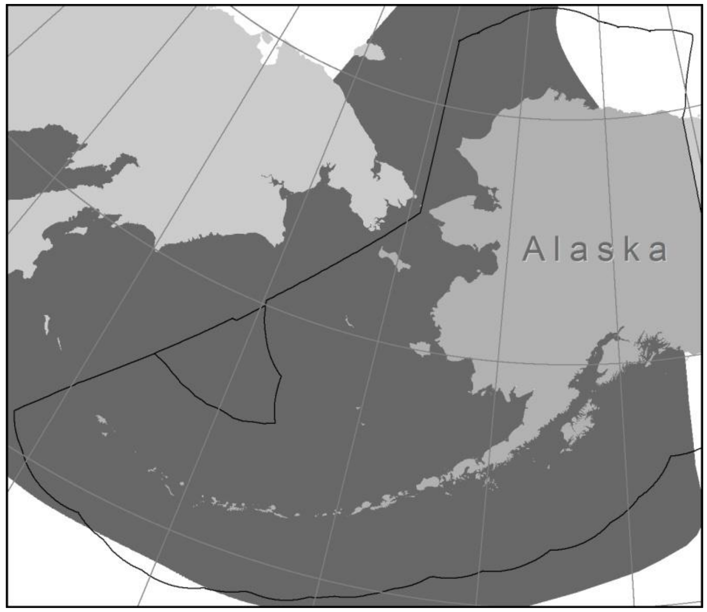

Figure 1. The ribbon seal stock is defined as the *Histriophoca fasciata* species (dark shaded areas depict the combined summer and winter distribution). This stock assessment considers only the portion of the stock occurring in U.S. waters (i.e., the U.S. Exclusive Economic Zone delineated by a black line).

This stock is defined as the *Histriophoca fasciata* species; however, this stock assessment considers only the portion of the stock found within U.S. waters bounded by the U.S. Exclusive Economic Zone (EEZ; Fig. 1), because the relevant stock assessment data on abundance and human-caused mortality and serious injury are generally not available for the broader range of the stock or even for waters adjacent to the U.S. EEZ.

POPULATION SIZE

In the spring of 2012 and 2013, U.S. and Russian researchers conducted aerial abundance and distribution surveys over the entire ice-covered portions of the Bering Sea and Sea of Okhotsk (Moreland et al. 2013). Conn et al. (2014), using a sub-sample of the data collected from the U.S. portion of the Bering Sea in 2012, calculated an abundance estimate of 184,697 ribbon seals (95% CI: 139,617-240,225) in those waters. Although this is a preliminary abundance estimate, it is also the best available and it is a reasonable estimate for the entire portion of the stock in U.S. waters because relatively few ribbon seals are expected north of the Bering Strait during the surveys. When the final analyses for the Bering Sea and Sea of Okhotsk are complete, they will provide the first range-wide estimates of ribbon seal abundance.

Minimum Population Estimate

The minimum population estimate (N_{MIN}) for a stock is usually calculated using Equation 1 from the potential biological removal (PBR) guidelines (NMFS 2016): $N_{MIN} = N/\exp(0.842\times[\ln(1+[CV(N)]^2)]^{½})$, which approximates the 20th percentile of a distribution that is assumed to be log-normal. However, the abundance estimate based on Conn et al. (2014) was calculated using a Bayesian hierarchical framework, so we used the 20th percentile of the posterior distribution of abundance estimates as a more direct estimator of N_{MIN} than Equation 1 to provide an N_{MIN} of 163,086 ribbon seals in the U.S. Bering Sea in the spring.

Current Population Trend

Reliable data on trends in population abundance for the ribbon seal stock or for the portion of the stock within U.S. waters are not available.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate (R_{MAX}) is not available for the ribbon seal stock or for any portion of the stock within U.S. waters. Until additional data become available, the default pinniped maximum theoretical net productivity rate of 12% will be used for this stock (NMFS 2016).

POTENTIAL BIOLOGICAL REMOVAL

PBR is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: PBR = $N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 1.0, a value that may be used for stocks that are not known to be decreasing and are taken primarily by aboriginal subsistence hunters, provided there have not been recent increases in the levels of takes (NMFS 2016). Using the N_{MIN} based on Conn et al. (2014) for ribbon seals in the U.S. portion of the stock, the PBR is 9,785 seals (163,086 × 0.06 × 1.0).

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2014 and 2018 is listed, by marine mammal stock, in Young et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports. The minimum estimated mean annual level of human-caused mortality and serious injury for the portion of the ribbon seal stock in U.S. waters between 2014 and 2018 is 163 seals: 0.9 in U.S. commercial fisheries and 162 in the Alaska Native subsistence harvest (average statewide harvest, including struck and lost animals, in 2015, based on a recently published analysis (Nelson et al. 2019) that is higher and likely more accurate than previous estimates but also revealed stable or decreasing trends in harvest numbers; see below). Additional potential threats most likely to result in direct human-caused mortality or serious injury of this stock include the increased potential for oil spills due to an increase in vessel traffic in Alaska waters (with changes in sea-ice coverage).

Fisheries Information

Information for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is available in Appendix 3 of the Alaska Stock Assessment Reports (observer coverage) and in the NMFS List of Fisheries (LOF) and the fact sheets linked to fishery names in the LOF (observer coverage and reported incidental takes of marine mammals: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-list-fisheries, accessed December 2020).

Between 2014 and 2018, incidental mortality and serious injury of ribbon seals in U.S. waters occurred in four of the federally-managed U.S. commercial fisheries in Alaska monitored for incidental mortality and serious injury by fisheries observers: the Bering Sea/Aleutian Islands flatfish trawl, Bering Sea/Aleutian Islands pollock trawl, Bering Sea/Aleutian Islands Pacific cod trawl, and Bering Sea/Aleutian Islands rockfish trawl fisheries (Table 1; Breiwick 2013; MML, unpubl. data). The minimum estimated mean annual mortality and serious injury rate incidental to U.S. commercial fisheries between 2014 and 2018 is 0.9 ribbon seals, based exclusively on observer data.

Table 1. Summary of incidental mortality and serious injury of ribbon seals in U.S. waters due to U.S. commercial fisheries between 2014 and 2018 and calculation of the mean annual mortality and serious injury rate (Breiwick 2013; MML, unpubl. data). Methods for calculating percent observer coverage are described in Appendix 3 of the Alaska Stock Assessment Reports.

Fishery name	Years	Data type	Percent observer coverage	Observed mortality	Estimated mortality (CV)	Mean estimated annual mortality
Bering Sea/Aleutian Is. flatfish trawl	2014 2015 2016 2017 2018	obs data	100 100 99 100 100	1 0 0 0 0	1 (0.04) 0 0 0 0	0.2 (CV = 0.04)
Bering Sea/Aleutian Is. pollock trawl	2014 2015 2016 2017 2018	obs data	98 99 99 99 99	0 0 1 0 0	0 0 1.0 (0.13) 0 0	0.2 (CV = 0.13)
Bering Sea/Aleutian Is. Pacific cod trawl	2014 2015 2016 2017 2018	obs data	80 72 68 68 73	1 0 0 0 0	1.3 (0.49) 0 0 0 0	0.3 (CV = 0.49)
Bering Sea/Aleutian Is. rockfish trawl	2014 2015 2016 2017 2018	obs data	100 100 100 100 100	1 0 0 0 0	1 (0) 0 0 0 0	0.2 (CV = 0)
Minimum total estimated annual mortality						0.9 (CV = 0.15)

Alaska Native Subsistence/Harvest Information

NMFS signed an agreement with the Ice Seal Committee (ISC; 2006) to co-manage Alaska ice seal populations. This co-management agreement promotes full and equal participation by Alaska Natives in decisions affecting the subsistence management of ice seals (to the maximum extent allowed by law) as a tool for conserving ice seal populations in Alaska (https://www.fisheries.noaa.gov/alaska/marine-mammal-protection/co-management-marine-mammals-alaska, accessed December 2020).

Ribbon seals are an important resource for Alaska Native subsistence hunters. Approximately 64 coastal communities in Alaska, from Bristol Bay to the Beaufort Sea, harvest ice seals (ISC 2019). The ISC, as co-managers with NMFS, recognizes the importance of harvest information and has collected it since 2008. Annual household survey results compiled in a statewide harvest report include historical ice seal harvest information from 1960 to 2017 (Quakenbush and Citta 2008, ISC 2019). To estimate the recent subsistence harvest of ice seals, Nelson et al. (2019) used ice seal harvest survey data collected from 1992 to 2014 for 41 of 55 communities that regularly hunt ice seals, as well as the per capita removal estimates (based on the 2015 human population) from the surveyed communities, to estimate the average regional and statewide subsistence harvest (Table 2). The best statewide estimate of the average number of ribbon seals harvested in 2015, including struck and lost animals, is 162 seals (Nelson et al. 2019). The authors also found stable or decreasing trends in the annual numbers of ice seals harvested (Nelson et al. 2019).

Table 2. Average regional and statewide subsistence harvest (including struck and lost animals) of ribbon seals in 2015 (Nelson et al. 2019). See Figure 1 in Nelson et al. (2019) for a list of the communities in each region.

Region	Average harvest (including struck and lost animals)
North Slope Borough	0
Maniilaq	9
Kawerak	130
Association of Village Council Presidents	23
Bristol Bay Native Association	0
Statewide total	162

Other Mortality

In 2011, NMFS and the U.S. Fish and Wildlife Service declared an Unusual Mortality Event (UME) for pinnipeds in the Bering and Chukchi seas, due to the unusual number of sick or dead seals and walruses discovered with skin lesions, bald patches, and other symptoms. The UME occurred from 1 May 2011 to 31 December 2016 and primarily affected ice seals, including ringed seals, bearded seals, ribbon seals, and spotted seals. The investigation concluded that the skin and hair symptoms were signs of a molt abnormality; however, no infectious disease agent or environmental cause for the UME symptoms and mortality was identified (https://www.fisheries.noaa.gov/national/marine-life-distress/active-and-closed-unusual-mortality-events, accessed December 2020). Patchy baldness and delayed molt, however, continue to be observed in limited numbers (<20 per year) of harvested and beachcast ringed seals, bearded seals, ribbon seals, and spotted seals in Alaska.

Since 1 June 2018, elevated numbers of ice seal strandings have occurred in the Bering and Chukchi seas in Alaska and NMFS declared a UME for bearded seals, ringed seals, and spotted seals from 1 June 2018 to present in the Bering and Chukchi seas (https://www.fisheries.noaa.gov/national/marine-life-distress/active-and-closed-unusual-mortality-events, accessed December 2020). As of 31 July 2020, 298 ice seal strandings of all age classes have been reported, including 88 bearded seals, 72 ringed seals, 49 spotted seals, and 89 unidentified seals. Although the UME was not declared for ribbon seals, some of the unidentified carcasses could have been ribbon seals that were too decomposed to be identified. A subset of seals has been sampled for genetics and harmful algal bloom exposure and a few have had histopathology samples collected.

STATUS OF STOCK

Ribbon seals are not designated as depleted under the Marine Mammal Protection Act (MMPA) or listed as threatened or endangered under the Endangered Species Act (ESA). NMFS completed a comprehensive status review of ribbon seals under the ESA in 2013 (Boveng et al. 2013) and concluded that listing ribbon seals was not warranted at that time (78 FR 41371, 10 July 2013). The ribbon seal stock is not considered a strategic stock. The best estimate of the mean annual level of human-caused mortality and serious injury in the portion of the stock in U.S. waters is 163 ribbon seals, which is less than the PBR (9,785 seals). The minimum estimated mean annual rate of U.S. commercial fishery-related mortality and serious injury (0.9 seals) is less than 10% of the PBR (10% of PBR = 979) and, therefore, can be considered insignificant and approaching a zero mortality and serious injury rate. Population trends and status of this stock relative to its Optimum Sustainable Population are unknown.

There are key uncertainties in the assessment of the ribbon seal stock. The N_{MIN} used here, based on a 2012 Bering Sea density estimate from Conn et al. (2014) was calculated using only a sub-sample of the survey data and may be biased. Based on the best available information, ribbon seals are likely to be moderately sensitive to climate change.

HABITAT CONCERNS

The main concern about the conservation status of ribbon seals is long-term habitat loss and modification resulting from climate change (Boveng et al. 2013). Laidre et al. (2008) concluded that on a worldwide basis ribbon seals were likely to be moderately sensitive to climate change, based on an analysis of various life-history features that could be affected by climate. Climate models consistently project substantial reductions in both the extent and timing of sea ice within the range of ribbon seals in Alaska waters; however, the sea ice in the Bering Sea is expected to continue forming annually in winter for the foreseeable future. Ribbon seals are closely associated with sea ice, particularly during the periods of reproduction and molting. The presence of sea ice is considered a requirement for whelping and nursing young, providing a platform out of the water to facilitate these life-history events. Similarly, the molt is believed to be promoted by elevated skin temperatures that, in polar regions, can only be achieved when seals haul out of the water. There will likely be more frequent years in which ice coverage is reduced, resulting in a decline in the long-term average ice extent; however, ribbon seals will likely continue to encounter sufficient ice to support adequate vital rates.

A second major concern, driven primarily by the production of carbon dioxide (CO_2) emissions, is the modification of habitat by ocean acidification, which may alter prey populations and other important aspects of the marine ecosystem. Ocean acidification, a result of increased CO_2 in the atmosphere, may affect ribbon seal survival and recruitment through disruption of trophic regimes that are dependent on calcifying organisms. The nature and timing of such impacts are extremely uncertain. As described in Boveng et al. (2013), changes in ribbon seal prey, anticipated in response to ocean warming and loss of sea ice, have the potential for negative impacts, but the possibilities are complex. Ecosystem responses may have very long lags as they propagate through trophic webs. Because of ribbon seals' apparent dietary flexibility, this threat may be of less immediate concern than the threats from sea-ice degradation.

Additional habitat concerns include the potential effects from increased shipping (particularly in the Bering Strait), such as disturbance from vessel traffic and the potential for oil spills.

CITATIONS

Boveng, P. L., J. L. Bengtson, M. F. Cameron, S. P. Dahle, E. A. Logerwell, J. M. London, J. E. Overland, J. T. Sterling, D. E. Stevenson, B. L. Taylor, and H. L. Ziel. 2013. Status review of the ribbon seal (*Histriophoca fasciata*). U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-255, 174 p.

Braham, H. W., J. J. Burns, G. A. Fedoseev, and B. D. Krogman. 1984. Habitat partitioning by ice-associated pinnipeds: distribution and density of seals and walruses in the Bering Sea, April 1976, p. 25-47. *In* F. H. Fay and G. A. Fedoseev (eds.), Soviet-American cooperative research on marine mammals. Vol. 1. Pinnipeds. U.S. Dep. Commer., NOAA Tech. Rep. NMFS-12.

Breiwick, J. M. 2013. North Pacific marine mammal bycatch estimation methodology and results, 2007-2011. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-260, 40 p.

Burns, J. J. 1970. Remarks on the distribution and natural history of pagophilic pinnipeds in the Bering and Chukchi Seas. J. Mammal. 51:445-454.

Burns, J. J. 1981. Ribbon seal-*Phoca fasciata*, p. 89-109. *In* S. H. Ridgway and R. J. Harrison (eds.), Handbook of Marine Mammals. Vol. 2. Seals. Academic Press, New York.

Burns, J. J., L. H. Shapiro, and F. H. Fay. 1981. Ice as marine mammal habitat in the Bering Sea, p. 781-797. *In* D. W. Hood and J. A. Calder (eds.), The Eastern Bering Sea Shelf: Oceanography and Resources. Vol. 2. U.S. Dep. Commer., NOAA, Office of Marine Pollution Assessment, Juneau, AK.

Conn, P. B., J. M. Ver Hoef, B. T. McClintock, E. E. Moreland, J. M. London, M. F. Cameron, S. P. Dahle, and P. L. Boveng. 2014. Estimating multispecies abundance using automated detection systems: ice-associated seals in the Bering Sea. Methods Ecol. Evol. 5:1280-1293. DOI: dx.doi.org/10.1111/2041-210X.12127 .

Frouin-Mouy, H., X. Mouy, C. L. Berchok, S. B. Blackwell, and K. M. Stafford. 2019. Acoustic occurrence and behavior of ribbon seals (*Histriophoca fasciata*) in the Bering, Chukchi, and Beaufort seas. Polar Biol. 42(4):657-674.

Hannay D. E., J. Delarue, X. Mouy, B. S. Martin, D. Leary, J. N. Oswald, and J. Vallarta. 2013. Marine mammal acoustic detections in the northeastern Chukchi Sea, September 2007–July 2011. Continental Shelf Research 67:127-146.

Ice Seal Committee (ISC). 2019. The subsistence harvest of ice seals in Alaska – a compilation of existing information, 1960-2017. 86 p. Available online: http://www.north-slope.org/departments/wildlife-management/co-management-organizations/ice-seal-committee . Accessed December 2020.

Jones, J. M., B. J. Thayre, E. H. Roth, M. Mahoney, I. Sia, K. Merculief, C. Jackson, C. Zeller, M. Clare, A. Bacon, S. Weaver, Z. Gentes, R. J. Small, I. Stirling, S. M. Wiggins, and J. A. Hildebrand. 2014. Ringed, bearded, and ribbon seal vocalizations north of Barrow, Alaska: seasonal presence and relationship with sea ice. Arctic 67(2):203-222.

Laidre, K. L., I. Stirling, L. F. Lowry, Ø. Wiig, M. P. Heide-Jørgensen, and S. H. Ferguson. 2008. Quantifying the sensitivity of Arctic marine mammals to climate-induced habitat change. Ecol. Appl. 18(2):S97-S125.

Moore, S. E., K. M. Stafford, H. Melling, C. Berchok, Ø. Wiig, K. M. Kovacs, C. Lydersen, and J. Richter-Menge. 2012. Comparing marine mammal acoustic habitats in Atlantic and Pacific sectors of the High Arctic: year-long records from Fram Strait and the Chukchi Plateau. Polar Biol. 35:475-480. DOI: dx.doi.org/10.1007/s00300-011-1086-y .

Moreland, E., M. Cameron, and P. Boveng. 2013. Bering Okhotsk Seal Surveys (BOSS): joint U.S.-Russian aerial surveys for ice associated-seals, 2012-13. Alaska Fisheries Science Center Quarterly Report (July-August-September 2013).

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2020.

Nelson, M. A., L. T. Quakenbush, B. D. Taras, and Ice Seal Committee. 2019. Subsistence harvest of ringed, bearded, spotted, and ribbon seals in Alaska is sustainable. Endang. Species Res. 40:1-16. DOI: dx.doi.org/10.3354/esr00973 .

Quakenbush, L., and J. Citta. 2008. Biology of the ribbon seal in Alaska. Report to NMFS. Arctic Marine Mammal Program, Alaska Department of Fish and Game, Fairbanks, AK. 45 p.

Young, N. C., B. J. Delean, V. T. Helker, J. C. Freed, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2014-2018. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-413, 142 p.

Revised 12/30/2020

BELUGA WHALE (*Delphinapterus leucas*): Beaufort Sea Stock

NOTE – April 2022: NMFS is evaluating whether scientific issues raised by co-management partners in November 2021 concerning the Eastern Bering Sea beluga whale Stock Assessment Report may also be applicable to the Beaufort Sea beluga whale Stock Assessment Report. Any resulting changes will be reflected in a future Stock Assessment Report.

STOCK DEFINITION AND GEOGRAPHIC RANGE

Beluga whales are distributed throughout seasonally ice-covered arctic and subarctic waters of the Northern Hemisphere (Gurevich 1980). In ice-covered regions, they are closely associated with open leads and polynyas (Hazard 1988). In Alaska, depending on season and region, beluga whales may occur in both offshore and coastal waters, with summer concentrations in upper Cook Inlet, Bristol Bay, eastern Bering Sea (i.e., Yukon River Delta, Norton Sound), eastern Chukchi Sea, and Beaufort Sea (Mackenzie River Delta) (Hazard 1988, O'Corry-Crowe et al. 2018) (Fig. 1). Seasonal distribution is affected by ice cover, tidal conditions, access to prey, temperature, and human interaction (Lowry 1985). Data from satellite transmitters attached to beluga whales from the Beaufort Sea, Eastern Chukchi Sea, Eastern Bering Sea, and Bristol Bay stocks identify ranges that are relatively distinct month to month for these stocks' summering areas and autumn migratory routes (e.g., Hauser et al. 2014, Citta et al. 2017, Lowry et al. 2019). Transmitters that lasted through the winter showed that beluga whales from these summering areas overwinter in the Bering Sea; these stocks are not known to overlap in space and time in the Bering Sea (Suydam 2009, Citta et al. 2017, Lowry et al. 2019).

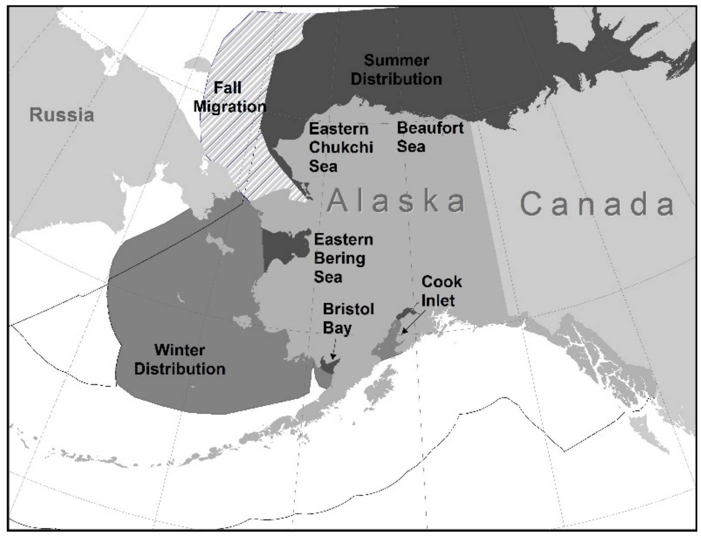

Figure 1. Approximate distribution for all five beluga whale stocks. The Beaufort Sea, Eastern Chukchi Sea, Eastern Bering Sea, and Bristol Bay beluga whale stocks summer in the Beaufort Sea (Beaufort Sea and Eastern Chukchi Sea stocks) and Bering Sea (Eastern Bering Sea and Bristol Bay stocks); they overwinter in the Bering Sea. The Bristol Bay and Cook Inlet beluga whale stocks show only small seasonal shifts in distribution, remaining in Bristol Bay and Cook Inlet, respectively, throughout the year. Summering areas are dark gray, wintering areas are lighter gray, and the hashed area is a region used by the Eastern Chukchi Sea and Beaufort Sea stocks for autumn migration. The U.S. Exclusive Economic Zone is delineated by a black line.

New genetic analyses have further defined five of the summering aggregations in the Bering, Chukchi, and Beaufort seas as follows: Bristol Bay, eastern Bering Sea (Norton Sound), eastern Chukchi Sea (Kasegaluk Lagoon), eastern Beaufort Sea (Mackenzie-Amundsen), and Gulf of Anadyr (Anadyr Bay) (O'Corry-Crowe et al. 2018). These genetic analyses, combined with new telemetry data, demonstrate that the demographically distinct summering aggregations return to discrete wintering areas and disperse and interbreed over limited distances but do not appear to interbreed extensively (O'Corry-Crowe et al. 2018).

The Beaufort Sea and Eastern Chukchi Sea stocks of beluga whales migrate between the Bering and Beaufort seas. Beaufort Sea beluga whales depart the Bering Sea in early spring, migrate through the Chukchi Sea and into the Canadian waters of the Beaufort Sea where they remain in the summer and fall, returning to the Bering Sea in late fall. Eastern Chukchi Sea beluga whales depart the Bering Sea in late spring and early summer, migrate through the Chukchi Sea and into the western Beaufort Sea where they remain in the summer, returning to the Bering Sea in the fall. The Eastern Bering Sea beluga whale stock remains in the Bering Sea but migrates south

near Bristol Bay in winter and returns north to Norton Sound and the mouth of the Yukon River in summer (Suydam 2009, Hauser et al. 2014, Citta et al. 2017, Lowry et al. 2019). Beluga whales tagged in Bristol Bay (Quakenbush 2003; Citta et al. 2016, 2017) and Cook Inlet (Goetz et al. 2012; Shelden et al. 2015, 2018; Lowry et al. 2019) remain in those areas throughout the year, showing only small seasonal shifts in distribution.

The following information was considered in classifying beluga whale stock structure based on the Dizon et al. (1992) phylogeographic approach: 1) Distributional data: geographic distribution discontinuous in summer (Frost and Lowry 1990); 2) Population response data: distinct population trends among regions occupied in summering areas (O'Corry-Crowe et al. 2018); 3) Phenotypic data: unknown; and 4) Genotypic data: mitochondrial DNA analyses indicate distinct differences among the five summering areas (O'Corry-Crowe et al. 2018). Based on this information, five beluga whale stocks are recognized within U.S. waters: 1) Cook Inlet, 2) Bristol Bay, 3) Eastern Bering Sea, 4) Eastern Chukchi Sea, and 5) Beaufort Sea (Fig. 1).

POPULATION SIZE

The sources of information to estimate abundance for beluga whales in waters of northern Alaska and western Canada have included both opportunistic and systematic observations. Duval (1993) reported an estimate of 21,000 beluga whales for the Beaufort Sea stock, similar to that reported by Seaman et al. (1985). The most recent aerial survey conducted in July 1992 resulted in an estimate of 19,629 beluga whales (CV = 0.229) in the eastern Beaufort Sea (Harwood et al. 1996). To account for availability bias, a correction factor (CF), which was not data-based, has been recommended for the Beaufort Sea beluga whale stock (Duval 1993), resulting in a population estimate of 39,258 whales (19,629 × 2). A coefficient of variation (CV) for the CF is not available; however, this CF was considered negatively biased by the Alaska Scientific Review Group (SRG) considering that aerial survey CFs for this stock were estimated between 2.5 and 3.27 (Frost and Lowry 1995). Additionally, the 1992 surveys did not encompass the entire summer range of Beaufort Sea beluga whales (Richard et al. 2001), thus, are negatively biased.

During summer 2019, the governments of the United States and Canada supported independent aerial line-transect surveys in the eastern Beaufort Sea to conduct an abundance survey for bowhead whales. Those data are also being analyzed to derive abundance estimates for the Beaufort Sea stock of beluga whales.

Minimum Population Estimate

For the Beaufort Sea beluga whale stock, the minimum population estimate (N_{MIN}) is calculated according to Equation 1 from the potential biological removal (PBR) guidelines (NMFS 2016): $N_{MIN} = N/\exp(0.842\times[\ln(1+[CV(N)]^2)]^{½})$. Using the population estimate (N) of 39,258 whales and an associated CV(N) of 0.229, N_{MIN} for this stock would be 32,453 whales. However, because the survey data are more than 8 years old, it is not considered a reliable minimum population estimate for calculating a PBR and N_{MIN} is considered unknown.

Current Population Trend

The current population trend of the Beaufort Sea stock of beluga whales is unknown. Aerial surveys seaward of the Mackenzie River Delta between 1982-1985 and 2007-2009 indicate that the stock in that area is at least stable or increasing (Harwood and Kingsley 2013).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate (R_{MAX}) is not available for the Beaufort Sea beluga whale stock. Until additional data become available, the default cetacean maximum theoretical net productivity rate of 4% will be used for this stock (NMFS 2016).

POTENTIAL BIOLOGICAL REMOVAL

PBR is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: PBR = $N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 1.0, a value that may be used for stocks that are not known to be decreasing and are taken primarily by aboriginal subsistence hunters, provided there have not been recent increases in the levels of takes (NMFS 2016). However, the 2016 guidelines for preparing Stock Assessment Reports (NMFS 2016) state that abundance estimates older than 8 years should not be used to calculate PBR due to a decline in confidence in the reliability of an aged abundance estimate. Therefore, the PBR for this stock is considered undetermined.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2014 and 2018 is listed, by marine mammal stock, in Young et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports. The minimum estimated mean annual level of human-caused mortality and serious injury for Beaufort Sea beluga whales between 2014 and 2018 is 104 beluga whales: 29 in subsistence takes by Alaska Natives and 75 in subsistence takes by Canadian Inuvialuit.

Fisheries Information

Information for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is available in Appendix 3 of the Alaska Stock Assessment Reports (observer coverage) and in the NMFS List of Fisheries (LOF) and the fact sheets linked to fishery names in the LOF (observer coverage and reported incidental takes of marine mammals: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-list-fisheries, accessed December 2020).

There were no reports of mortality or serious injury of this stock incidental to U.S. commercial fisheries or subsistence fisheries in Alaska between 2014 and 2018.

Alaska Native Subsistence/Harvest Information

NMFS signed an agreement with the Alaska Beluga Whale Committee (ABWC; 2000) to co-manage western Alaska beluga whale populations in the Bering Sea (including Bristol Bay), Chukchi Sea, and Beaufort Sea. This co-management agreement promotes full and equal participation by Alaska Natives in decisions affecting the subsistence management of beluga whales (to the maximum extent allowed by law) as a tool for conserving beluga whale populations in Alaska (https://www.fisheries.noaa.gov/alaska/marine-mammal-protection/co-management-marine-mammals-alaska, accessed December 2020).

The subsistence take of Beaufort Sea beluga whales within U.S. waters is reported by the ABWC. The most recent Alaska Native subsistence harvest estimates for the Beaufort Sea beluga whale stock are provided in Table 1 (ABWC, unpubl. data, 2020). The annual subsistence take by Alaska Native hunters averaged 29 Beaufort Sea beluga whales landed between 2014 and 2018. It should be noted that beluga whales harvested at Utqiaġvik (formerly Barrow) in spring are assumed to be from the Beaufort Sea stock, while those harvested in summer are assumed to be from the Eastern Chukchi Sea stock.

Table 1. Summary of Beaufort Sea beluga whales landed by Alaska Native subsistence hunters between 2014 and 2018 (ABWC, unpubl. data, 2020). These are minimum estimates of the total number of beluga whales taken, because not all landed whales and struck and lost whales are consistently reported.

Year	Number landed	Number struck and lost	Total (landed + struck and lost)
2014	24	7	31
2015	43	1	44
2016	43	no data	43
2017	10	no data	10
2018	13	4	17
Mean annual number (landed + struck and lost)			29

Canadian Inuvialuit Subsistence/Harvest Information

The subsistence take of beluga whales within the Canadian waters of the Beaufort Sea is reported by the Fisheries Joint Management Committee (FJMC). The data are collected through on-site harvest monitoring conducted by the FJMC at Inuvialuit communities in the Mackenzie River Delta, Northwest Territories. The Canadian Inuvialuit subsistence harvest estimates for the Beaufort Sea beluga whale stock between 2014 and 2018 are provided in Table 2 (FJMC Beluga Monitor Program, FJMC, Inuvik, NT, Canada). Given these data, the annual subsistence take in Canada averaged 75 beluga whales between 2014 and 2018.

Thus, the estimated mean annual subsistence take of Beaufort Sea beluga whales in U.S. and Canadian waters between 2014 and 2018 is 104 whales (29 + 75).

Table 2. Summary of Beaufort Sea beluga whales harvested by Canadian Inuvialuit subsistence hunters between 2014 and 2018 (FJMC, unpubl. data). N/A indicates that data are not available.

Year	Number landed	Number struck and lost	Total (landed + struck and lost)
2014	104	2	106
2015*	75	1	76
2016	48	1	49
2017	66	N/A	66
2018	76	2	78
Mean annual number taken (landed + struck and lost)			75

*The number of beluga whales landed in 2015 was changed from 82 to 75 whales (resulting in a change in the total harvest from 83 to 76 whales) based on updated harvest information from the FJMC (FJMC, unpubl. data).

STATUS OF STOCK

No fishery-related mortality or serious injury has been reported for the Beaufort Sea stock of beluga whales between 2014 and 2018; therefore, the mean annual mortality and serious injury rate incidental to U.S. commercial fisheries can be considered insignificant and approaching a zero mortality and serious injury rate. The minimum estimated mean annual level of human-caused mortality and serious injury for this stock is 104 beluga whales. Beaufort Sea beluga whales are not designated as depleted under the Marine Mammal Protection Act or listed as threatened or endangered under the Endangered Species Act. Therefore, the Beaufort Sea beluga whale stock is classified as a non-strategic stock. At this time, it is not possible to assess the status of this stock relative to its Optimum Sustainable Population.

There are key uncertainties in the assessment of the Beaufort Sea stock of beluga whales. The most recently analyzed surveys were conducted more than 8 years ago and did not cover the entire population; given the lack of information on population trend, the abundance estimates are not used to calculate an N_{MIN} and the PBR level is undetermined.

HABITAT CONCERNS

Evidence indicates that the arctic climate is changing rapidly and significantly, and one result of this change is a reduction in the extent and duration of sea ice in some regions (ACIA 2004, Johannessen et al. 2004). These changes are likely to affect marine mammal species in the Arctic. Ice-associated animals, such as the beluga whale, are sensitive to changes in arctic weather, sea-surface temperatures, and sea-ice extent, and the concomitant effect on prey availability. There are indications that decreases in seasonal sea ice have influenced beluga whale phenology; however, Beaufort Sea beluga whales did not show a statistically significant change in the timing of their southward migration in response to changes in sea ice (Hauser et al. 2017). An offshore shift in distribution of Beaufort Sea beluga whales between an earlier sample in 1982-1985 and a later sample in 2007-2009 was attributed either to increased habitat due to more open water or potential response to industrial activity (Harwood and Kingsley 2013). Decreases in seasonal sea ice may also increase the risk of killer whale predation (O'Corry-Crowe et al. 2016). There are insufficient data to make reliable predictions of the effects of arctic climate change on beluga whales; however, Laidre et al. (2008) and Heide-Jørgensen et al. (2010) concluded that on a worldwide basis beluga whales were likely to be less sensitive to climate change than other arctic cetaceans because of their wide distribution and flexible behavior. Increased human activity in the Arctic, including increased oil and gas exploration and development and increased nearshore development, has the potential to impact beluga whale habitat (Moore et al. 2000, Lowry et al. 2006). However, predicting the type and magnitude of these impacts is difficult.

CITATIONS

Arctic Climate Impact Assessment (ACIA). 2004. Impacts of a Warming Arctic: Arctic Climate Impact Assessment. Cambridge University Press, Cambridge, UK.

Citta, J. J., L. T. Quakenbush, K. J. Frost, L. Lowry, R. C. Hobbs, and H. Aderman. 2016. Movements of beluga whales (*Delphinapterus leucas*) in Bristol Bay, Alaska. Mar. Mammal Sci. 32:1272-1298. DOI: dx.doi.org/10.1111/mms.12337 .

Citta, J. J., P. Richard, L. F. Lowry, G. O'Corry-Crowe, M. Marcoux, R. Suydam, L. T. Quakenbush, R. C. Hobbs, D. I. Litovka, K. J. Frost, T. Gray, J. Orr, B. Tinker, H. Aderman, and M. L. Druckenmiller. 2017. Satellite telemetry reveals population specific winter ranges of beluga whales in the Bering Sea. Mar. Mammal Sci. 33:236-250. DOI: dx.doi.org/10.1111/mms.12357 .

Dizon, A. E., C. Lockyer, W. F. Perrin, D. P. DeMaster, and J. Sisson. 1992. Rethinking the stock concept: a phylogeographic approach. Conserv. Biol. 6:24-36.

Duval, W. S. 1993. Proceedings of a workshop on Beaufort Sea beluga: February 3-6, 1992, Vancouver, BC. Env. Studies Res. Found. Report No. 123. Calgary. 33 p. + appendices.

Frost, K. J., and L. F. Lowry. 1990. Distribution, abundance, and movements of beluga whales, *Delphinapterus leucas*, in coastal waters of western Alaska, p. 39-57. *In* T. G. Smith, D. J. St. Aubin, and J. R. Geraci (eds.), Advances in research on the beluga whale, *Delphinapterus leucas*. Can. Bull. Fish. Aquat. Sci. 224.

Frost, K. J., and L. F. Lowry. 1995. Radio tag based correction factors for use in beluga whale population estimates. Working paper for Alaska Beluga Whale Committee Scientific Workshop, Anchorage, AK, 5-7 April 1995. 12 p. Available from Alaska Department of Fish and Game, 1300 College Rd., Fairbanks, AK 99701.

Goetz, K. T., P. W. Robinson, R. C. Hobbs, K. L. Laidre, L. A. Huckstadt, and K. E. W. Shelden. 2012. Movement and dive behavior of beluga whales in Cook Inlet, Alaska. AFSC Processed Rep. 2012-03, 40 p. Alaska Fisheries Science Center, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Gurevich, V. S. 1980. Worldwide distribution and migration patterns of the white whale (beluga), *Delphinapterus leucas*. Rep. Int. Whal. Comm. 30:465-480.

Harwood, L. A., and M. C. S. Kingsley. 2013. Trends in the offshore distribution and relative abundance of Beaufort Sea belugas, 1982-85 vs 2007-09. Arctic 66(3):247-256.

Harwood, L. A., S. Innes, P. Norton, and M. C. S. Kingsley. 1996. Distribution and abundance of beluga whales in the Mackenzie Estuary, southeast Beaufort Sea and west Amundsen Gulf during late July 1992. Can. J. Fish. Aquat. Sci. 53:2262-2273.

Hauser, D. D. W., K. L. Laidre, R. S. Suydam, and P. R. Richard. 2014. Population-specific home ranges and migration timing of Pacific Arctic beluga whales (*Delphinapterus leucas*). Polar Biol. 37:1171-1183. DOI: dx.doi.org/10.1007/s00300-014-1510-1 .

Hauser, D. D. W., K. L. Laidre, K. M. Stafford, H. L. Stern, R. S. Suydam, and P. R. Richard. 2017. Decadal shifts in autumn migration timing by Pacific Arctic beluga whales are related to delayed annual sea ice formation. Glob. Change Biol. 23:2206-2217. DOI: dx.doi.org/10.1111/gcb.13564 .

Hazard, K. 1988. Beluga whale, *Delphinapterus leucas*, p. 195-235. *In* J. W. Lentfer (ed.), Selected Marine Mammals of Alaska. Species Accounts with Research and Management Recommendations. Marine Mammal Commission, Washington, DC.

Heide-Jørgensen, M., K. Laidre, D. Borchers, T. Marques, H. Stern, and M. Simon. 2010. The effect of sea-ice loss on beluga whales (*Delphinapterus leucas*) in West Greenland. Polar Res. 29:198-208. DOI: dx.doi.org/10.1111/j.1751-8369.2009.00142.x .

Johannessen, O. M., L. Bengtsson, M. W. Miles, S. I. Kuzmina, V. A. Semenov, G. V. Alexseev, A. P. Nagurnyi, V. F. Zakharov, L. P. Bobylev, L. H. Pettersson, K. Hasselmann, and H. P. Cattle. 2004. Arctic climate change: observed and modelled temperature and sea-ice variability. Tellus A 56(4):328-341. DOI: dx.doi.org/10.3402/tellusa.v56i4.14418 .

Laidre, K. L., I. Stirling, L. Lowry, Ø. Wiig, M. P. Heide-Jørgensen, and S. Ferguson. 2008. Quantifying the sensitivity of arctic marine mammals to climate-induced habitat change. Ecol. Appl. 18(2):S97-S125.

Lowry, L. F. 1985. The belukha whale (*Delphinapterus leucas*), p. 3-13. *In* J. J. Burns, K. J. Frost, and L. F. Lowry (eds.), Marine mammal species accounts. Alaska Department of Fish and Game, Game Tech. Bull. 7.

Lowry, L., G. O'Corry-Crowe, and D. Goodman. 2006. *Delphinapterus leucas* (Cook Inlet population). In: IUCN 2006. 2006 IUCN Red List of Threatened Species.

Lowry, L. F., J. J. Citta, G. O'Corry-Crowe, L. T. Quakenbush, K. J. Frost, R. Suydam, R. C. Hobbs, and T. Gray. 2019. Distribution, abundance, harvest, and status of western Alaska beluga whale, *Delphinapterus leucas*, stocks. Mar. Fish. Rev. 81(3-4):54-71.

Moore, S. E., K. E. W. Shelden, L. K. Litzky, B. A. Mahoney, and D. J. Rugh. 2000. Beluga, *Delphinapterus leucas*, habitat associations in Cook Inlet, Alaska. Mar. Fish. Rev. 62(3):60-80.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2020.

O'Corry-Crowe, G., A. R. Mahoney, R. Suydam, L. Quakenbush, A. Whiting, L. Lowry, and L. Harwood. 2016. Genetic profiling links changing sea-ice to shifting beluga whale migration patterns. Biol. Lett. 12:20160404. DOI: dx.doi.org/10.1098/rsbl.2016.0404 .

O'Corry-Crowe, G., R. Suydam, L. Quakenbush, B. Potgieter, L. Harwood, D. Litovka, T. Ferrer, J. Citta, V. Burkanov, K. Frost, and B. Mahoney. 2018. Migratory culture, population structure and stock identity in North Pacific beluga whales (*Delphinapterus leucas*). PLoS ONE 13(3):e0194201.

Quakenbush, L. 2003. Summer movements of beluga whales captured in the Kvichak River in May 2002 and 2003. Alaska Beluga Whale Committee Report 03-03. 15 p.

Richard P. R., A. R. Martin, and J. R. Orr. 2001. Summer and autumn movements of belugas of the eastern Beaufort Sea stock. Arctic 54: 223-236.

Seaman, G. A., K. J. Frost, and L. F. Lowry. 1985. Investigations of belukha whales in coastal waters of western and northern Alaska. Part I. Distribution, abundance and movements. U.S. Dep. Commer., NOAA, OCSEAP Final Report 56:153-220. Available from NOAA-OMA-OAD, Alaska Office, 701 C. Street, P.O. Box 56, Anchorage, AK 99513.

Shelden, K. E. W., K. T. Goetz, D. J. Rugh, D. G. Calkins, B. A. Mahoney, and R. C. Hobbs. 2015. Spatio-temporal changes in beluga whale, *Delphinapterus leucas*, distribution: results from aerial surveys (1977-2014), opportunistic sightings (1975-2014), and satellite tagging (1999-2003) in Cook Inlet, Alaska. Mar. Fish. Rev. 77(2):1-31 + appendices. DOI: dx.doi.org/10.7755/MFR.77.2.1 .

Shelden, K. E. W., K. T. Goetz, R. C. Hobbs, L. K. Hoberecht, K. L. Laidre, B. A. Mahoney, T. L. McGuire, S. A. Norman, G. O'Corry-Crowe, D. J. Vos, G. M. Ylitalo, S. A. Mizroch, S. Atkinson, K. A. Burek-Huntington, and C. Garner. 2018. Beluga whale, *Delphinapterus leucas*, satellite-tagging and health assessments in Cook Inlet, Alaska, 1999 to 2002. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-369, 227 p.

Suydam, R. S. 2009. Age, growth, reproduction, and movements of beluga whales (*Delphinapterus leucas*) from the eastern Chukchi Sea. Ph.D. Dissertation, University of Washington, School of Aquatic and Fishery Sciences, Seattle, WA.

Young, N. C., B. J. Delean, V. T. Helker, J. C. Freed, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2014-2018. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-413, 142 p.

BELUGA WHALE (*Delphinapterus leucas*): Eastern Chukchi Sea Stock

NOTE – April 2022: NMFS is evaluating whether scientific issues raised by co-management partners in November 2021 concerning the Eastern Bering Sea beluga whale Stock Assessment Report may also be applicable to the Eastern Chukchi Sea beluga whale Stock Assessment Report. Any resulting changes will be reflected in a future Stock Assessment Report.

STOCK DEFINITION AND GEOGRAPHIC RANGE

Beluga whales are distributed throughout seasonally ice-covered arctic and subarctic waters of the Northern Hemisphere (Gurevich 1980). In ice-covered regions, they are closely associated with open leads and polynyas (Hazard 1988). In Alaska, depending on season and region, beluga whales may occur in both offshore and coastal waters, with summer concentrations in upper Cook Inlet, Bristol Bay, eastern Bering Sea (i.e., Yukon River Delta, Norton Sound), eastern Chukchi Sea, and Beaufort Sea (Mackenzie River Delta) (Hazard 1988, O'Corry-Crowe et al. 2018) (Fig. 1). Seasonal distribution is affected by ice cover, tidal conditions, access to prey, temperature, and human interaction (Lowry 1985). Data from satellite transmitters attached to beluga whales from the Beaufort Sea, Eastern Chukchi Sea, Eastern Bering Sea, and Bristol Bay stocks identify ranges that are relatively distinct month to month for these stocks' summering areas and autumn migratory routes (e.g., Hauser et al. 2014, Citta et al. 2017, Lowry et al. 2019). Transmitters that lasted through the winter showed that beluga whales from these summering areas overwinter in the Bering Sea; these stocks are not known to overlap in space and time in the Bering Sea (Suydam 2009, Citta et al. 2017, Lowry et al. 2019).

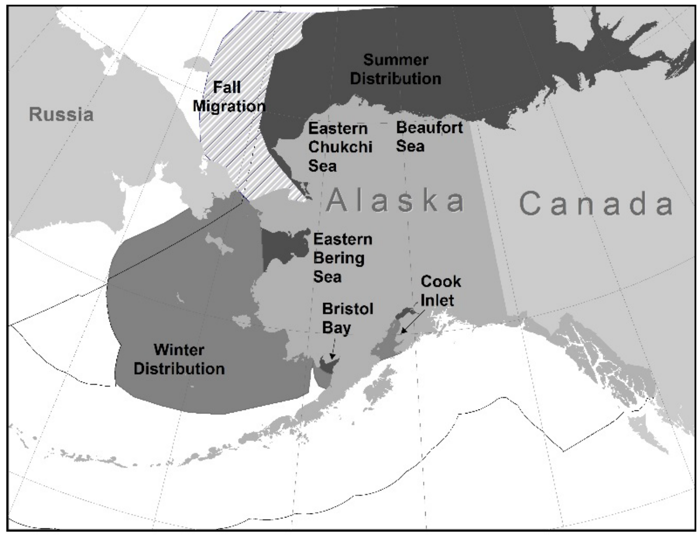

Figure 1. Approximate distribution for all five beluga whale stocks. The Beaufort Sea, Eastern Chukchi Sea, Eastern Bering Sea, and Bristol Bay beluga whale stocks summer in the Beaufort Sea (Beaufort Sea and Eastern Chukchi Sea stocks) and Bering Sea (Eastern Bering Sea and Bristol Bay stocks); they overwinter in the Bering Sea. The Bristol Bay and Cook Inlet beluga whale stocks show only small seasonal shifts in distribution, remaining in Bristol Bay and Cook Inlet, respectively, throughout the year. Summering areas are dark gray, wintering areas are lighter gray, and the hashed area is a region used by the Eastern Chukchi Sea and Beaufort Sea stocks for autumn migration. The U.S. Exclusive Economic Zone is delineated by a black line.

New genetic analyses have further defined five of the summering aggregations in the Bering, Chukchi, and Beaufort seas as follows: Bristol Bay, eastern Bering Sea (Norton Sound), eastern Chukchi Sea (Kasegaluk Lagoon), eastern Beaufort Sea (Mackenzie-Amundsen), and Gulf of Anadyr (Anadyr Bay) (O'Corry-Crowe et al. 2018). These genetic analyses, combined with new telemetry data, demonstrate that the demographically distinct summering aggregations return to discrete wintering areas and disperse and interbreed over limited distances but do not appear to interbreed extensively (O'Corry-Crowe et al. 2018).

The Beaufort Sea and Eastern Chukchi Sea stocks of beluga whales migrate between the Bering and Beaufort seas. Beaufort Sea beluga whales depart the Bering Sea in early spring, migrate through the Chukchi Sea and into the Canadian waters of the Beaufort Sea where they remain in the summer and fall, returning to the Bering Sea in late fall. Eastern Chukchi Sea beluga whales depart the Bering Sea in late spring and early summer, migrate through the Chukchi Sea and into the western Beaufort Sea where they remain in the summer, returning to the Bering Sea in the fall. The Eastern Bering Sea beluga whale stock remains in the Bering Sea but migrates south

near Bristol Bay in winter and returns north to Norton Sound and the mouth of the Yukon River in summer (Suydam 2009, Hauser et al. 2014, Citta et al. 2017, Lowry et al. 2019). Beluga whales tagged in Bristol Bay (Quakenbush 2003; Citta et al. 2016, 2017) and Cook Inlet (Goetz et al. 2012; Shelden et al. 2015, 2018; Lowry et al. 2019) remain in those areas throughout the year, showing only small seasonal shifts in distribution.

At least some of the Eastern Chukchi Sea beluga whales move along coastal areas in late June and animals are sighted in the area until about mid-July (Frost and Lowry 1990, Frost et al. 1993, Suydam et al. 2001). Data from satellite tags attached to Eastern Chukchi Sea beluga whales captured in Kasegaluk Lagoon during the summer showed these whales traveled 1,100 km north of the Alaska coastline, into the Canadian Beaufort Sea within 3 months (Suydam et al. 2001, Hauser et al. 2014). These movements indicated overlap in distribution with the Beaufort Sea beluga whale stock during late summer. Satellite-telemetry data from 24 whales tagged from 1998 to 2007 suggest variation in movement patterns for different age and/or sex classes during July to September (Suydam et al. 2005, Hauser et al. 2014). Compared to tagged adult females, tagged adult males used deeper waters and remained there for the summer. All beluga whales that moved into the Arctic Ocean (north of 75°N) were males, and males traveled through 90% pack ice to reach deeper waters in the Beaufort Sea and Arctic Ocean (79-80°N) by late July/early August. In September, males occupied the southern Canada Basin and Beaufort Sea shelf and slope, maintaining a small core area over Barrow Canyon and a larger core area over the eastern Canada Basin slope. In October, the male distribution shifted south and west, with one core area extending over the Beaufort Sea slope into Barrow Canyon and another over Herald Shoal in the Chukchi Sea. Adult females ranged from just offshore of the Kasegaluk Lagoon system to Barrow Canyon in July. In August, the distribution of females was limited to Barrow Canyon and the adjacent western Beaufort Sea shelf and slope. In September, the female distribution expanded to include the southern Canada Basin, before shifting south and west in October to the Chukchi Sea and western Beaufort Sea (Hauser et al. 2014). In late autumn, only six tags continued to transmit and those whales migrated south through the eastern Bering Strait into the northern Bering Sea, remaining north of Saint Lawrence Island during the winter (Hauser et al. 2014, Citta et al. 2017). A whale tagged in the eastern Chukchi Sea in 2007 overwintered in the waters north of Saint Lawrence Island during 2007/2008, then moved towards King Island in April and May before moving north through the Bering Strait in late May and early June (Suydam 2009).

The following information was considered in classifying beluga whale stock structure based on the Dizon et al. (1992) phylogeographic approach: 1) Distributional data: geographic distribution discontinuous in summer (Frost and Lowry 1990); 2) Population response data: distinct population trends among regions occupied in summering areas (O'Corry-Crowe et al. 2018); 3) Phenotypic data: unknown; and 4) Genotypic data: mitochondrial DNA analyses indicate distinct differences among the five summering areas (O'Corry-Crowe et al. 2018). Based on this information, five beluga whale stocks are recognized within U.S. waters: 1) Cook Inlet, 2) Bristol Bay, 3) Eastern Bering Sea, 4) Eastern Chukchi Sea (Fig. 1), and 5) Beaufort Sea.

POPULATION SIZE

Frost et al. (1993) estimated the minimum size of the Eastern Chukchi Sea beluga whale stock at 1,200 whales, based on whale counts from aerial surveys conducted from 1989 to 1991. Survey effort was concentrated sea side of the 170-km long Kasegaluk Lagoon, an area known to be regularly used by beluga whales during the open-water season. The offshore areas that these beluga whales are known to frequent were not surveyed. Therefore, the targeted surveys provided only a minimum count. If this count is corrected using radio-telemetry data for the proportion of whales that were diving and thus not visible at the surface (2.62: Frost and Lowry 1995), and for the proportion of newborns and yearlings not observed due to small size and dark coloration (1.18: Brodie 1971), the total corrected abundance estimate for the Eastern Chukchi Sea stock is 3,710 whales (1,200 × 2.62 × 1.18).

During 25 June to 6 July 1998, aerial surveys were conducted in the eastern Chukchi Sea (DeMaster et al. 1998). The maximum single day count (1,172 whales) was derived from a photographic count of a large aggregation near Icy Cape (1,018 whales), plus whales counted along an ice edge transect (154 whales). This count is an underestimate, because it was clear to the observers that many more whales were present along and in the ice than they were able to count and only a small portion of the ice edge habitat was surveyed. Furthermore, only one of five beluga whales equipped with satellite tags a few days earlier remained within the survey area when the peak count occurred (DeMaster et al. 1998). It is not possible to estimate abundance from the 1998 survey. Not only were a large number of whales unavailable for counting, but the large Icy Cape aggregation was in shallow, clear water (DeMaster et al. 1998) and a correction factor (to account for missed whales) does not exist for beluga whales encountered in such conditions.

In July 2002, aerial surveys were conducted again in the eastern Chukchi Sea (Lowry and Frost 2002). Those surveys resulted in a peak count of 582 whales. A correction factor for whales that were not visible for this count is not available. Offshore sightings during this survey combined with satellite-tag data collected in 2001 (Lowry and Frost 2001, 2002) indicate that nearshore surveys for beluga whales will only result in partial counts for this stock.

A new strategy for deriving a population abundance estimate for the Eastern Chukchi Sea stock of beluga whales was based on summer aerial survey data from the Beaufort Sea, after the stock had migrated through the eastern Chukchi Sea. Analyses of satellite telemetry data from beluga whales belonging to the Eastern Chukchi Sea and Beaufort Sea stocks (Hauser et al. 2014) identified an area in the Beaufort Sea (140°W to 157°W) and period (19 July-20 August) when the two stocks did not overlap (Lowry et al. 2017). These aerial surveys were conducted as part of the Aerial Surveys of Arctic Marine Mammals (ASAMM) project in the northeastern Chukchi and Alaska Beaufort seas from 19 July to 20 August 2012-2017 (Clarke et al. 2018). A geographically stratified line-transect analysis that was based on the assumption that the Beaufort Sea and Eastern Chukchi Sea stocks are geographically segregated from mid-July through August (Hauser et al. 2014) resulted in the following population estimates of the Eastern Chukchi Sea beluga whales in the study area for each year from 2012 to 2017, respectively: 7,355 (CV=0.47), 6,813 (CV=0.47), 16,598 (CV=0.49), 6,456 (CV=0.48), 6,965 (CV=0.49) and 13,305 (CV=0.51) (Givens et al. 2019). These estimates incorporate a correction factor of 1.85 (Lowry et al. 2017) for whales that were submerged and, therefore, not visible to the aerial observers. These estimates do not account for whales that might have been outside the project area during the survey period.

The assumption that Eastern Chukchi Sea beluga whales are isolated from Beaufort Sea beluga whales is possibly flawed based on three lines of evidence: the assumption of a lack of overlap within the Alaska Beaufort Sea from late July to late August is based on satellite-tag data that are dated (few beluga whales from either stock have been tagged in the last decade); the assumed distribution of all Eastern Chukchi Sea and Beaufort Sea beluga whales in July and August cannot be determined from tags that were deployed at the same time and in locations that were too far apart for the tagged whales to overlap in July and August (all Eastern Chukchi Sea beluga whales were tagged near Point Lay in July and all Beaufort Sea beluga whales were tagged in the Mackenzie Delta mainly in July and in August, although numbers in these areas indicate the stocks were more wide-spread at this time); and genetic evidence from harvested beluga whales indicates that Beaufort Sea beluga whales are sometimes found in the Chukchi Sea in late July (O'Corry-Crowe et al. 2018). However, the Givens et al. (2019) abundance estimate reflects the best available data for Eastern Chukchi Sea beluga whales at this time.

Minimum Population Estimate

For the Eastern Chukchi Sea beluga whale stock, the minimum population estimate (N_{MIN}) is calculated according to Equation 1 from the potential biological removal (PBR) guidelines (NMFS 2016): $N_{MIN} = N/\exp(0.842 \times [\ln(1+[CV(N)]^2)]^{½})$. Using the 2017 population estimate of 13,305 and the associated coefficient of variation (CV) of 0.51, N_{MIN} for this stock is 8,875 whales; however, this N_{MIN} may be positively biased due to possible overlap between the Eastern Chukchi Sea and Beaufort Sea stocks of beluga whales during the survey in late July to late August.

Current Population Trend

There is no statistically significant trend in the abundance of the Eastern Chukchi Sea beluga whale stock inside the ASAMM study area from 19 July to 20 August in 2012-2017 (Givens et al. 2019). However, the interannual variation among the abundance estimates and the estimated CVs are both large.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate (R_{MAX}) is not available for the Eastern Chukchi Sea beluga whale stock. Until additional data become available, the default cetacean maximum theoretical net productivity rate of 4% will be used for this stock (NMFS 2016).

POTENTIAL BIOLOGICAL REMOVAL

PBR is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: $PBR = N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 1.0, a value that may be used for stocks that are not known to be decreasing and are taken primarily by aboriginal subsistence hunters, provided there have not been recent increases in the levels of takes (NMFS 2016). Therefore, the PBR for this stock is 178 beluga whales (8,875 × 0.02 × 1.0).

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2014 and 2018 is listed, by marine mammal stock, in Young et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports. The minimum estimated mean annual level of human-caused mortality and serious injury for Eastern Chukchi Sea beluga whales between 2014 and 2018 is 56 beluga whales in subsistence takes by Alaska Natives. Potential threats most likely to result in direct human-caused mortality and serious injury of this stock include entanglement in fishing gear.

Fisheries Information

Information for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is available in Appendix 3 of the Alaska Stock Assessment Reports (observer coverage) and in the NMFS List of Fisheries (LOF) and the fact sheets linked to fishery names in the LOF (observer coverage and reported incidental takes of marine mammals: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-list-fisheries, accessed December 2020).

In the nearshore waters of the southeastern Chukchi Sea, substantial efforts occur in gillnet (mostly set nets) and personal-use fisheries. Although a potential source of mortality, there have been no reported beluga whale takes as a result of these fisheries and such incidental takes could be counted as subsistence harvest.

There were no reports of mortality or serious injury of this stock incidental to U.S. commercial fisheries or subsistence fisheries in Alaska between 2014 and 2018.

Alaska Native Subsistence/Harvest Information

NMFS signed an agreement with the Alaska Beluga Whale Committee (ABWC; 2000) to co-manage western Alaska beluga whale populations in the Bering Sea (including Bristol Bay), Chukchi Sea, and Beaufort Sea. This co-management agreement promotes full and equal participation by Alaska Natives in decisions affecting the subsistence management of beluga whales (to the maximum extent allowed by law) as a tool for conserving beluga whale populations in Alaska (https://www.fisheries.noaa.gov/alaska/marine-mammal-protection/co-management-marine-mammals-alaska, accessed December 2020).

The subsistence take of Eastern Chukchi Sea beluga whales is reported by the ABWC. The most recent subsistence harvest estimates for the Eastern Chukchi Sea stock are provided in Table 1 (ABWC, unpubl. data, 2020). The annual subsistence take by Alaska Native hunters averaged 56 Eastern Chukchi Sea beluga whales landed between 2014 and 2018. It should be noted that beluga whales harvested at Utqiaġvik (formerly Barrow) in spring are assumed to be from the Beaufort Sea stock, while those harvested in summer are assumed to be from the Eastern Chukchi Sea stock.

Table 1. Summary of Eastern Chukchi Sea beluga whales landed by Alaska Native subsistence hunters between 2014 and 2018 (ABWC, unpubl. data, 2020). It should be noted that these harvest levels include takes from Kotzebue Sound (10 in 2014, 1 in 2015, 9 in 2016, 2 in 2017, and 15 in 2018; no data are available for struck and lost animals in Kotzebue Sound) which are likely from a population that is genetically distinct from the Eastern Chukchi Sea beluga whale stock. These are minimum estimates of the total number of beluga whales taken, because not all landed whales and struck and lost whales are consistently reported.

Year	Number landed	Number struck and lost	Total (landed + struck and lost)
2014	60	no data	60
2015	72	4	76
2016	23	0	23
2017	40	2	42
2018	80	0	80
Mean annual number (landed + struck and lost)			56

STATUS OF STOCK

No fishery-related mortality or serious injury has been reported for the Eastern Chukchi Sea stock of beluga whales between 2014 and 2018; therefore, the mean annual mortality and serious injury rate incidental to U.S. commercial fisheries can be considered insignificant and approaching a zero mortality and serious injury rate. The minimum estimated mean annual level of human-caused mortality and serious injury (56 beluga whales) is less than the PBR (178 whales). Eastern Chukchi Sea beluga whales are not designated as depleted under the Marine Mammal Protection Act or listed as threatened or endangered under the Endangered Species Act. Therefore, the Eastern Chukchi Sea stock of beluga whales is not classified as a strategic stock. The historical level and overall population trend is unknown and, given the uncertainty of the data, we are unable at this time to assess the status of this stock relative to its Optimum Sustainable Population. Recent data indicate no statistically significant trend from 2012 to 2017 (Givens et al. 2019).

There are key uncertainties in the assessment of the Eastern Chukchi Sea stock of beluga whales. The proportion of the stock within the ASAMM study area during the survey period used in the Lowry et al. (2017) and Givens et al. (2019) abundance analyses is unknown. The assumption that the Eastern Chukchi Sea and Beaufort Sea stocks are geographically segregated during the July-August time period used in Lowry et al.'s (2017) and Givens et al.'s (2019) abundance estimates is based on a relatively limited number of whales tagged between 1993 and 2007. Beaufort Sea beluga whales are found in Kotzebue (Chukchi Sea) in July of some years, indicating that the two stocks may overlap in July. This may result in a positive bias in the estimate of abundance for the Eastern Chukchi Sea stock. Coastal subsistence fisheries can occasionally cause incidental mortality or serious injury of a beluga whale; these incidental takes used for subsistence purposes are not always reported to the ABWC as a fishery interaction and may be included in the subsistence harvest reports for the stock.

HABITAT CONCERNS

Evidence indicates that the arctic climate is changing rapidly and significantly, and one result of this change is a reduction in the extent and duration of sea ice in some regions (ACIA 2004, Johannessen et al. 2004). These changes are likely to affect marine mammal species in the Arctic. Ice-associated animals, such as the beluga whale, are sensitive to changes in arctic weather, sea-surface temperatures, and sea-ice extent, and the concomitant effect on prey availability. There are indications that decreases in seasonal sea ice have influenced beluga whale phenology. Eastern Chukchi Sea beluga whales tagged between 2004 and 2012 were distributed farther north and east in September-November than those tagged between 1993 and 2002 (Hauser et al. 2017). Further, the median date at which tagged whales departed the Beaufort and Chukchi seas during their southbound migrations was 14-33 days later overall in 2004-2012 versus 1993-2002 (Hauser et al. 2017). Decreases in seasonal sea ice may also increase the risk of killer whale predation (O'Corry-Crowe et al. 2016).

There are insufficient data to make reliable predictions of the effects of arctic climate change on beluga whales; however, Laidre et al. (2008) and Heide-Jørgensen et al. (2010) concluded that on a worldwide basis beluga whales were likely to be less sensitive to climate change than other arctic cetaceans because of their wide distribution and flexible behavior. Stafford et al. (2016) found that dive behavior of Eastern Chukchi Sea beluga whales was correlated to wind speed and direction. When winds were from the WSW, whales made shallow dives likely exploiting the front developed by the Alaska Coastal Current between the coast and the deep Arctic basin. Strong winds from the ENE resulted in deeper, longer dives (Stafford et al. 2016). East winds are increasing in the Arctic (Pickart et al. 2009), thus, beluga whales may be spending more time diving at greater depths. Increased human activity in the Arctic, including increased oil and gas exploration and development and increased nearshore development, has the potential to impact beluga whale habitat (Moore et al. 2000, Lowry et al. 2006). However, predicting the type and magnitude of these impacts is difficult.

CITATIONS

Arctic Climate Impact Assessment (ACIA). 2004. Impacts of a Warming Arctic: Arctic Climate Impact Assessment. Cambridge University Press, Cambridge, UK.

Brodie, P. F. 1971. A reconsideration of aspects of growth, reproduction, and behavior of the white whale with reference to the Cumberland Sound, Baffin Island, population. J. Fish. Res. Bd. Can. 28:1309-1318.

Citta, J. J., L. T. Quakenbush, K. J. Frost, L. Lowry, R. C. Hobbs, and H. Aderman. 2016. Movements of beluga whales (*Delphinapterus leucas*) in Bristol Bay, Alaska. Mar. Mammal Sci. 32:1272-1298. DOI: dx.doi.org/10.1111/mms.12337 .

Citta, J. J., P. Richard, L. F. Lowry, G. O'Corry-Crowe, M. Marcoux, R. Suydam, L. T. Quakenbush, R. C. Hobbs, D. I. Litovka, K. J. Frost, T. Gray, J. Orr, B. Tinker, H. Aderman, and M. L. Druckenmiller. 2017.

Satellite telemetry reveals population specific winter ranges of beluga whales in the Bering Sea. Mar. Mammal Sci. 33:236-250. DOI: dx.doi.org/10.1111/mms.12357 .

Clarke, J. T., A. A. Brower, M. C. Ferguson, and A. L. Willoughby. 2018. Distribution and relative abundance of marine mammals in the eastern Chukchi and western Beaufort Seas, 2017. Annual Report, OCS Study BOEM 2018-023. Marine Mammal Laboratory, Alaska Fisheries Science Center, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

DeMaster, D. P. 1995. Minutes from the 4-5 and 11 January 1995 meeting of the Alaska Scientific Review Group, Anchorage, Alaska. 27 p. + appendices. Available from Alaska Fisheries Science Center, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

DeMaster, D. P., W. Perryman, and L. F. Lowry. 1998. Beluga whale surveys in the eastern Chukchi Sea, July, 1998. Alaska Beluga Whale Committee Report 98-2. 16 p.

Dizon, A. E., C. Lockyer, W. F. Perrin, D. P. DeMaster, and J. Sisson. 1992. Rethinking the stock concept: a phylogeographic approach. Conserv. Biol. 6:24-36.

Frost, K. J., and L. F. Lowry. 1990. Distribution, abundance, and movements of beluga whales, *Delphinapterus leucas*, in coastal waters of western Alaska, p. 39-57. *In* T. G. Smith, D. J. St. Aubin, and J. R. Geraci (eds.), Advances in research on the beluga whale, *Delphinapterus leucas*. Can. Bull. Fish. Aquat. Sci. 224.

Frost, K. J., and L. F. Lowry. 1995. Radio tag based correction factors for use in beluga whale population estimates. Working paper for Alaska Beluga Whale Committee Scientific Workshop, Anchorage, AK, 5-7 April 1995. 12 p. Available from Alaska Department of Fish and Game, 1300 College Rd., Fairbanks, AK 99701.

Frost, K. J., L. F. Lowry, and G. Carroll. 1993. Beluga whale and spotted seal use of a coastal lagoon system in the northeastern Chukchi Sea. Arctic 46:8-16.

Givens, G. H., M. C. Ferguson, J. T. Clarke, A. Willoughby, A. Brower, and R. Suydam. 2019. Abundance of the Eastern Chukchi Sea stock of beluga whales, 2012-2017. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/68a/ASI/09). 15 p.

Goetz, K. T., P. W. Robinson, R. C. Hobbs, K. L. Laidre, L. A. Huckstadt, and K. E. W. Shelden. 2012. Movement and dive behavior of beluga whales in Cook Inlet, Alaska. AFSC Processed Rep. 2012-03, 40 p. Alaska Fisheries Science Center, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Gurevich, V. S. 1980. Worldwide distribution and migration patterns of the white whale (beluga), *Delphinapterus leucas*. Rep. Int. Whal. Comm. 30:465-480.

Hauser, D. D. W., K. L. Laidre, R. S. Suydam, and P. R. Richard. 2014. Population-specific home ranges and migration timing of Pacific Arctic beluga whales (*Delphinapterus leucas*). Polar Biol. 37(8):1171-1183. DOI: dx.doi.org/10.1007/s00300-014-1510-1 .

Hauser, D. D. W., K. L. Laidre, K. M. Stafford, H. L. Stern, R. S. Suydam, and P. R. Richard. 2017. Decadal shifts in autumn migration timing by Pacific Arctic beluga whales are related to delayed annual sea ice formation. Glob. Change Biol. 23:2206-2217. DOI: dx.doi.org/10.1111/gcb.13564 .

Hazard, K. 1988. Beluga whale, *Delphinapterus leucas*, p. 195-235. *In* J. W. Lentfer (ed.), Selected Marine Mammals of Alaska. Species Accounts with Research and Management Recommendations. Marine Mammal Commission, Washington, DC.

Heide-Jørgensen, M., K. Laidre, D. Borchers, T. Marques, H. Stern, and M. Simon. 2010. The effect of sea-ice loss on beluga whales (*Delphinapterus leucas*) in West Greenland. Polar Res. 29:198–208. DOI: dx.doi.org/10.1111/j.1751-8369.2009.00142.x .

Johannessen, O. M., L. Bengtson, M. W. Miles, S. I. Kuzmina, V. A. Semenov, G. V. Alexseev, A. P. Nagurnyi, V. F. Zakharov, L. P. Bobylev, L. H. Pettersson, K. Hasselmann, and H. P. Cattle. 2004. Arctic climate change: observed and modeled temperature and sea-ice variability. Tellus 56A:328-341.

Laidre, K. L., I. Stirling, L. Lowry, Ø. Wiig, M. P. Heide-Jørgensen, and S. Ferguson. 2008. Quantifying the sensitivity of Arctic marine mammals to climate-induced habitat change. Ecol. Appl. 18(2):S97-S125.

Lowry, L. F. 1985. The belukha whale (*Delphinapterus leucas*), p. 3-13. *In* J. J. Burns, K. J. Frost, and L. F. Lowry (eds.), Marine mammals species accounts. Alaska Department of Fish and Game, Game Tech. Bull. 7.

Lowry, L., and K. Frost. 2001. Beluga whale surveys in the Chukchi Sea, July 2001. Alaska Beluga Whale Committee Rep. 01-1 submitted to NMFS, Juneau, AK. 9 p.

Lowry, L., and K. Frost. 2002. Beluga whale surveys in the eastern Chukchi Sea, July 2002. Alaska Beluga Whale Committee Report 02-2 submitted to NMFS, Juneau, AK. 10 p.

Lowry, L., G. O'Corry-Crowe, and D. Goodman. 2006. *Delphinapterus leucas* (Cook Inlet population). *In* IUCN 2006. 2006 IUCN Red List of Threatened Species.

Lowry, L. F., M. C. S. Kingsley, D. D. W. Hauser, J. Clarke, and R. Suydam. 2017. Aerial survey estimates of abundance of the Eastern Chukchi Sea stock of beluga whales (*Delphinapterus leucas*) in 2012. Arctic 70(3):273-286. DOI: dx.doi.org/10.14430/arctic4667 .

Lowry, L. F., J. J. Citta, G. O'Corry-Crowe, L. T. Quakenbush, K. J. Frost, R. Suydam, R. C. Hobbs, and T. Gray. 2019. Distribution, abundance, harvest, and status of western Alaska beluga whale, *Delphinapterus leucas*, stocks. Mar. Fish. Rev. 81(3-4):54-71.

Moore, S. E., K. E. W. Shelden, L. K. Litzky, B. A. Mahoney, and D. J. Rugh. 2000. Beluga, *Delphinapterus leucas*, habitat associations in Cook Inlet, Alaska. Mar. Fish. Rev. 62(3):60-80.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2020.

O'Corry-Crowe G., A. R. Mahoney, R. Suydam, L. Quakenbush A. Whiting, L. Lowry, and L. Harwood. 2016. Genetic profiling links changing sea-ice to shifting beluga whale migration patterns. Biol. Lett. 12:20160404. DOI: dx.doi.org/10.1098/rsbl.2016.0404 .

O'Corry-Crowe, G., R. Suydam, L. Quakenbush, B. Potgieter, L. Harwood, D. Litovka, T. Ferrer, J. Citta, V. Burkanov, K. Frost, and B. Mahoney. 2018. Migratory culture, population structure and stock identity in North Pacific beluga whales (*Delphinapterus leucas*). PLoS ONE 13(3):e0194201.

Pickart, R. S., G. W. K. Moore, D. J. Torres, P. S. Fratantoni, R. A. Goldsmith, and J. Yang. 2009. Upwelling on the continental slope of the Alaskan Beaufort Sea: storms, ice, and oceanographic response. J. Geophys. Res. 114:C00A13. DOI: dx.doi.org/10.1029/2008JC005009 .

Quakenbush, L. 2003. Summer movements of beluga whales captured in the Kvichak River in May 2002 and 2003. Alaska Beluga Whale Committee Report 03-03. 15 p.

Shelden, K. E. W., K. T. Goetz, D. J. Rugh, D. G. Calkins, B. A. Mahoney, and R. C. Hobbs. 2015. Spatio-temporal changes in beluga whale, *Delphinapterus leucas*, distribution: results from aerial surveys (1977-2014), opportunistic sightings (1975-2014), and satellite tagging (1999-2003) in Cook Inlet, Alaska. Mar. Fish. Rev. 77(2):1-31 + appendices. DOI: dx.doi.org/10.7755/MFR.77.2.1 .

Stafford, K. M., J. J. Citta, S. R. Okkonen, and R. S. Suydam. 2016. Wind-dependent beluga whale dive behavior in Barrow Canyon, Alaska. Deep Sea Res. II 118:57-65. DOI: dx.doi.org/10.1016/j.dsr.2016.10.006 .

Suydam, R. S. 2009. Age, growth, reproduction, and movements of beluga whales (*Delphinapterus leucas*) from the eastern Chukchi Sea. Ph.D. Dissertation University of Washington, School of Aquatic and Fishery Sciences, Seattle, WA.

Suydam, R. S., L. F. Lowry, K. J. Frost, G. M. O'Corry-Crowe, and D. Pikok, Jr. 2001. Satellite tracking of Eastern Chukchi Sea beluga whales into the Arctic Ocean. Arctic 54(3):237-243.

Suydam, R. S., L. F. Lowry, and K. J. Frost. 2005. Distribution and movements of beluga whales from the Eastern Chukchi Sea stock during summer and early autumn. OCS Study MMS 2005-035 Final Report. 48 p.

Young, N. C., B. J. Delean, V. T. Helker, J. C. Freed, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2014-2018. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-413, 142 p.

Revised 12/30/2017

BELUGA WHALE (*Delphinapterus leucas*): Eastern Bering Sea Stock

NOTE – April 2022: Following consultation with the Alaska Beluga Whale Committee, NMFS withdrew the final 2020 Eastern Bering Sea beluga whale Stock Assessment Report. It is replaced here with the most recently published final Stock Assessment Report for this stock, last revised in 2017.

STOCK DEFINITION AND GEOGRAPHIC RANGE

Beluga whales are distributed throughout seasonally ice-covered arctic and subarctic waters of the Northern Hemisphere (Gurevich 1980) and are closely associated with open leads and polynyas in ice-covered regions (Hazard 1988). In Alaska, depending on season and region, beluga whales may occur in both offshore and coastal waters, with summer concentrations in upper Cook Inlet, Bristol Bay, the eastern Bering Sea (i.e., Yukon Delta and Norton Sound), eastern Chukchi Sea, and Beaufort Sea (Mackenzie River Delta) (Hazard 1988, O'Corry-Crowe et al. 1997) (Fig. 1). Seasonal distribution is affected by ice cover, tidal conditions, access to prey, temperature, and human interaction (Lowry 1985). Data from satellite transmitters attached to a few whales from the Beaufort Sea, Eastern Chukchi Sea, and Eastern Bering Sea stocks show ranges that are relatively distinct month to month for these populations' summering areas and autumn migratory routes (e.g., Hauser et al. 2014, Citta et al. 2017). The few transmitters that lasted through the winter showed that beluga whales from these summering areas overwinter in the Bering Sea; the stocks may use separate wintering locations and probably remain separated through the winter (Suydam 2009, Citta et al. 2017).

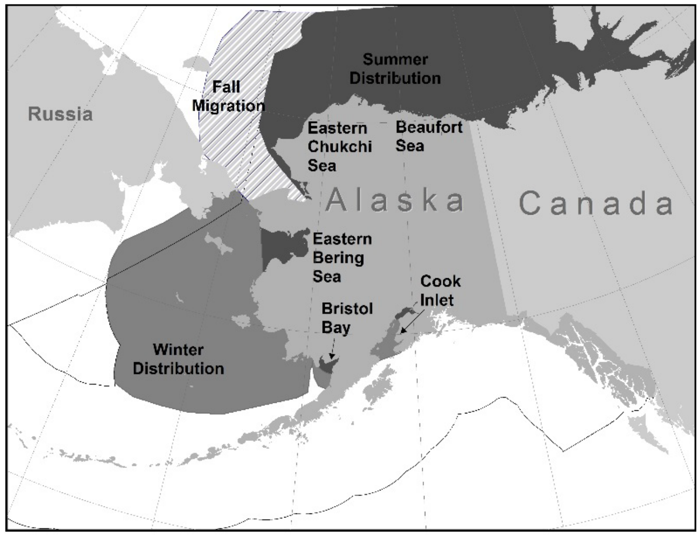

Figure 1. Approximate distribution for all five beluga whale stocks. Summering areas are dark gray, wintering areas are lighter gray, and the hashed area is a region used by the Eastern Chukchi Sea and Beaufort Sea stocks for autumn migration.

The Beaufort Sea and Eastern Chukchi Sea stocks of beluga whales migrate between the Bering and Beaufort seas. Beaufort Sea beluga whales depart from the Bering Sea in early spring, through the Chukchi Sea and into the Canadian waters of the Beaufort Sea where they remain in the summer and fall, returning to the Bering Sea in late fall. Eastern Chukchi Sea beluga whales migrate out of the Bering Sea in late spring and early summer, into the Chukchi Sea and western Beaufort Sea where they remain in the summer, returning to the Bering Sea in the fall. The Eastern Bering Sea stock remains in the Bering Sea but moves south near Bristol Bay in winter and returns north to Norton Sound and the mouth of the Yukon River in summer (Suydam 2009, Hauser et al. 2014, Citta et al. 2017). Beluga whales found in Bristol Bay (Quakenbush 2003; Citta et al. 2016, 2017) and Cook Inlet (Hobbs et al. 2005, Goetz et al. 2012, Shelden et al. 2015) remain in those areas throughout the year, showing only small seasonal shifts in distribution.

Two beluga whales from the Eastern Bering Sea stock were tagged with satellite transmitters in 2012 near Nome. The beluga whales moved south from Nome through ice covered shelf waters during the winter, swimming south near Hagemeister Island and the Walrus Islands in Bristol Bay, before returning to Norton Sound in the spring (Citta et al. 2017). A beluga whale tagged near Nome in September 2016 has remained in the vicinity of Nome and Norton Sound through mid-January 2017 due to low ice cover in the Bering Sea (Alaska Beluga Whale Committee, unpubl. data).

The following information was considered in classifying beluga whale stock structure based on the Dizon et al. (1992) phylogeographic approach: 1) Distributional data: geographic distribution discontinuous in summer (Frost and Lowry 1990); 2) Population response data: distinct population trends among regions occupied in summer;

3) Phenotypic data: unknown; and 4) Genotypic data: mitochondrial DNA analyses indicate distinct differences among the five summering areas (O'Corry-Crowe et al. 1997). Based on this information, five beluga whale stocks are recognized within U.S. waters: 1) Cook Inlet, 2) Bristol Bay, 3) Eastern Bering Sea, 4) Eastern Chukchi Sea, and 5) Beaufort Sea (Fig. 1).

POPULATION SIZE

The Alaska Beluga Whale Committee (ABWC) has been working to develop a population estimate for the Eastern Bering Sea stock since the first systematic aerial surveys of the Norton Sound/Yukon Delta region during May, June, and September 1992 and June 1993-1995 (Lowry et al. 1999). Beluga whale density estimates were calculated for the June 1992 surveys using strip-transect methods, and for the June 1993-1995 surveys using line-transect methods. Correction factors were applied to account for whales that were missed during the surveys (those below the surface and not visible and dark colored neonates). Lowry et al. (1999) concluded that the best abundance estimate for the Eastern Bering Sea stock was 17,675 beluga whales (95% CI: 9,056-34,515, not accounting for variance in correction factors), based on counts made in early June 1995. Additional aerial surveys of the Norton Sound/Yukon Delta region were conducted in June 1999 and 2000 (Lowry et al. 2017). Unlike previous survey years, in 1999 sea ice persisted in western Norton Sound resulting in a much different distribution of beluga whales, and the data were not used for population estimation. In 2000, systematic transect lines were flown covering the entire study region, and the data were analyzed using a covariate line-transect model. Results indicate 3,497 beluga whales (CV = 0.37) were seen at the surface in the study area (Lowry et al. 2017). If this estimate were doubled to correct for the proportion of whales that were diving, and thus not visible at the surface, the total abundance for the Eastern Bering Sea stock would be 6,994 beluga whales (95% CI: 3,162-15,472).

Minimum Population Estimate

For the Eastern Bering Sea stock of beluga whales, the minimum population estimate (N_{MIN}) is calculated according to Equation 1 from the potential biological removal (PBR) guidelines (Wade and Angliss 1997): $N_{MIN} = N/\exp(0.842\times[\ln(1+[CV(N)]^2)]^{½})$. Using the population estimate (N) of 6,994 and an associated coefficient of variation CV(N) of 0.37, N_{MIN} for this stock is 5,173 beluga whales. However, because the survey data are more than 8 years old, it is not considered a reliable minimum population estimate for calculating a PBR, and N_{MIN} is considered unknown.

Current Population Trend

Surveys to estimate population abundance in Norton Sound were not conducted prior to 1992. Annual estimates of population size from surveys flown in 1992-1995 and 1999-2000 have varied widely, due partly to differences in survey coverage and conditions between years. Available data do not allow an evaluation of population trend for the Eastern Bering Sea stock.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate is unavailable for the Eastern Bering Sea stock of beluga whales. Lowry et al. (2008) estimated the rate of increase of the Bristol Bay beluga whale stock was 4.8% per year (95% CI = 2.1%-7.5%) over a 12-year period. However, until additional data become available specific to the Eastern Bering Sea stock, the cetacean maximum theoretical net productivity rate (R_{MAX}) of 4% will be used for this stock (Wade and Angliss 1997).

POTENTIAL BIOLOGICAL REMOVAL

PBR is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: $PBR = N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 1.0, the value for cetacean stocks that are thought to be stable in the presence of a subsistence harvest (Wade and Angliss 1997). However, the 2016 guidelines for preparing Stock Assessment Reports (NMFS 2016) state that abundance estimates older than 8 years should not be used to calculate PBR due to a decline in confidence in the reliability of an aged abundance estimate. Therefore, the PBR for the Eastern Bering Sea stock of beluga whales is considered undetermined.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Detailed information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals in 2011-2015 is listed, by marine mammal stock, in Helker et al. (2017); however, only the mortality and serious injury data are included in the Stock Assessment Reports. The total estimated annual level of human-caused mortality and serious injury for Eastern Bering Sea beluga whales in 2011-2015 is 206 beluga whales: 0.2 in U.S. commercial fisheries and 206 in subsistence takes by Alaska Natives; however, a reliable estimate of mortality and serious injury in U.S. commercial fisheries is not available because there has never been an observer program for nearshore commercial fisheries in the eastern Bering Sea region. Assignment of mortality and serious injury to the Eastern Chukchi Sea, Eastern Bering Sea, and Bristol Bay stocks when stock is unknown, and the event occurred at a time and in an area where the three stocks could occur, may result in overestimating stock specific mortality and serious injury in federal commercial fisheries. Potential threats most likely to result in direct human-caused mortality or serious injury of this stock include entanglement in fishing gear.

Fisheries Information

Detailed information (including observer programs, observer coverage, and observed incidental takes of marine mammals) for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is presented in Appendices 3-6 of the Alaska Stock Assessment Reports.

During 2011-2015, one beluga whale mortality occurred in the Bering Sea/Aleutian Islands pollock trawl fishery (Table 1; Breiwick 2013; MML, unpubl. data). A genetics sample was collected but has not been analyzed. Since the stock of the beluga whale is unknown, and the event occurred at a time and in an area where the Eastern Chukchi Sea, Eastern Bering Sea, and Bristol Bay stocks could occur, this mortality has been assigned to all three stocks (NMFS 2016).

Table 1. Summary of incidental mortality and serious injury of Eastern Bering Sea beluga whales due to U.S. commercial fisheries in 2011-2015 and calculation of the mean annual mortality and serious injury rate (Breiwick 2013; MML, unpubl. data). Methods for calculating percent observer coverage are described in Appendix 6 of the Alaska Stock Assessment Reports.

Fishery name	Years	Data type	Percent observer coverage	Observed mortality	Estimated mortality	Mean estimated annual mortality
Bering Sea/Aleutian Is. pollock trawl	2011 2012 2013 2014 2015	obs data	98 98 97 98 99	0 0 1 0 0	0 0 1.0 0 0	0.2 (CV = 0.09)
Minimum total estimated annual mortality						0.2 (CV = 0.16)

In the nearshore waters of the Eastern Bering Sea, substantial effort occurs in commercial and subsistence fisheries, mostly for salmon and herring. The salmon fishery uses gillnet gear similar to that used in Bristol Bay, where it is known that beluga whales have been incidentally taken (Frost et al. 1984). However, there are no useful data on beluga whale incidental takes from this stock because there have never been observer programs in these commercial fisheries and there is no reporting requirement for takes in personal use fisheries. NMFS assumes that all beluga whales killed in these fisheries are used for subsistence, regardless of the method of harvest, and are reported to the ABWC. These subsistence takes are included in the Alaska Native Subsistence/Harvest Information section, below.

The minimum mean annual mortality and serious injury rate incidental to U.S. commercial fisheries in 2011-2015 is 0.2 beluga whales from this stock. However, because there has never been an observer program for state-managed nearshore commercial fisheries in the eastern Bering Sea region, a reliable estimate of the mortality and serious injury incidental to U.S. commercial fisheries is not available.

Alaska Native Subsistence/Harvest Information

The subsistence take of beluga whales from the Eastern Bering Sea stock is provided by the ABWC. The most recent subsistence harvest estimates for the stock are provided in Table 2 (ABWC, unpubl. data, 2016). Beluga whales harvested in Kuskokwim villages are included in the total harvest for the Eastern Bering Sea beluga whale stock. The annual subsistence take by Alaska Native villages averaged 206 beluga whales landed from the Eastern Bering Sea stock in 2011-2015

Table 2. Summary of Eastern Bering Sea beluga whales landed by Alaska Native subsistence hunters in 2011-2015 (ABWC, unpubl. data, 2016). These are minimum estimates of the total number of beluga whales taken, since struck and lost data are not consistently provided.

Year	Reported total number landed
2011	205
2012	181
2013	216
2014	237
2015	193
Mean annual number landed	206

STATUS OF STOCK

A minimum estimate of the mean annual mortality and serious injury rate incidental to U.S. commercial fisheries is 0.2 whales. Because the PBR is undetermined, the mean annual U.S. commercial fishery-related mortality and serious injury rate that can be considered insignificant and approaching zero mortality and serious injury rate is unknown. The total estimated annual level of human-caused mortality and serious injury is 206 beluga whales. Eastern Bering Sea beluga whales are not designated as depleted under the MMPA or listed as threatened or endangered under the Endangered Species Act. Therefore, the Eastern Bering Sea stock of beluga whales is classified as a non-strategic stock.

There are some key uncertainties in the assessment of the Eastern Bering Sea stock of beluga whales. The abundance is based on a line-transect survey; the resulting estimate is doubled to account for the proportion of whales that are diving and thus missed by the observers. It is not known whether doubling the estimate accurately accounts for whales missed. The population rate of increase is unknown. Coastal commercial fisheries that overlap with this stock have either never been observed or have not been observed recently, so mortality and serious injury of Eastern Bering Sea beluga whales in commercial fisheries could be underestimated. Coastal subsistence fisheries for fish will occasionally cause incidental mortality or serious injury of a beluga whale; these incidental takes used for subsistence purposes are not always reported to the ABWC and included in the estimate of subsistence harvest for the stock.

HABITAT CONCERNS

Evidence indicates that the arctic climate is changing significantly and that one result of the change is a reduction in the extent and duration of sea ice in most regions of the Arctic (ACIA 2004, Johannessen et al. 2004). These changes are likely to affect marine mammal species in the Arctic. Ice-associated animals, such as the beluga whale, are sensitive to changes in arctic weather, sea-surface temperatures, and ice extent, and the concomitant effect on prey availability. Decreases in seasonal sea ice may also increase the risk of killer whale predation (O'Corry-Crowe et al. 2016). It is unknown whether Eastern Bering Sea beluga whales have changed their areas of use in the winter; however, information from the Beaufort Sea and Eastern Chukchi Sea populations (Hauser et al. 2017), where tag data are more extensive, suggest that changes in timing of migration and winter distribution may have occurred. There are insufficient data to make reliable predictions of the effects of arctic climate change on beluga whales; however, Laidre et al. (2008) and Heide-Jørgensen et al. (2010) concluded that on a worldwide basis beluga whales were likely to be less sensitive to climate change than other arctic cetaceans because of their wide distribution and flexible behavior. Increased human activity in the Arctic, including increased oil and gas exploration and development and increased nearshore development, has the potential to impact habitat for beluga whales (Moore et al. 2000, Lowry et al. 2006); however, predicting the type and magnitude of the impacts is difficult.

CITATIONS

Arctic Climate Impact Assessment (ACIA). 2004. Impacts of a Warming Arctic: Arctic Climate Impact Assessment. Cambridge University Press, Cambridge, UK.

Breiwick, J. M. 2013. North Pacific marine mammal bycatch estimation methodology and results, 2007-2011. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-260, 40 p.

Citta, J. J., L. T. Quakenbush, K. J. Frost, L. Lowry, R. C. Hobbs, and H. Aderman. 2016. Movements of beluga whales (*Delphinapterus leucas*) in Bristol Bay, Alaska. Mar. Mammal Sci. 32:1272-1298. DOI: dx.doi.org/10.1111/mms.12337 .

Citta, J. J., P. Richard, L. F. Lowry, G. O'Corry-Crowe, M. Marcoux, R. Suydam, L. T. Quakenbush, R. C. Hobbs, D. I. Litovka, K. J. Frost, T. Gray, J. Orr, B. Tinker, H. Aderman, and M. L. Druckenmiller. 2017. Satellite telemetry reveals population specific winter ranges of beluga whales in the Bering Sea. Mar. Mammal Sci. 33:236-250. DOI: dx.doi.org/10.1111/mms.12357 .

Dizon, A. E., C. Lockyer, W. F. Perrin, D. P. DeMaster, and J. Sisson. 1992. Rethinking the stock concept: a phylogeographic approach. Conserv. Biol. 6:24-36.

Frost, K. J., and L. F. Lowry. 1990. Distribution, abundance, and movements of beluga whales, *Delphinapterus leucas*, in coastal waters of western Alaska, p. 39-57. *In* T. G. Smith, D. J. St. Aubin, and J. R. Geraci (eds.), Advances in research on the beluga whale, *Delphinapterus leucas*. Can. Bull. Fish. Aquat. Sci. 224.

Frost, K. J., L. F. Lowry, and R. R. Nelson. 1984. Belukha whale studies in Bristol Bay, Alaska, p. 187-200. *In* Proceedings of the workshop on biological interactions among marine mammals and commercial fisheries in the southeastern Bering Sea, October 18-21, 1983, Anchorage AK. Alaska Sea Grant Report 84-1.

Goetz, K. T., P. W. Robinson, R. C. Hobbs, K. L. Laidre, L. A. Huckstadt, and K. E. W. Shelden. 2012. Movement and dive behavior of beluga whales in Cook Inlet, Alaska. AFSC Processed Rep. 2012-03, 40 p. Alaska Fisheries Science Center, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Gurevich, V. S. 1980. Worldwide distribution and migration patterns of the white whale (beluga), *Delphinapterus leucas*. Rep. Int. Whal. Comm. 30:465-480.

Hauser, D. D. W., K. L. Laidre, R. S. Suydam, and P. R. Richard. 2014. Population-specific home ranges and migration timing of Pacific Arctic beluga whales (*Delphinapterus leucas*). Polar Biol. 37:1171-1183. DOI: dx.doi.org/10.1007/s00300-014-1510-1 .

Hauser, D. D. W., K. L. Laidre, K. M. Stafford, H. L. Stern, R. S. Suydam, and P. R. Richard. 2017. Decadal shifts in autumn migration timing by Pacific Arctic beluga whales are related to delayed annual sea ice formation. Glob. Change Biol. 23:2206-2217. DOI: dx.doi.org/10.1111/gcb.13564 .

Hazard, K. 1988. Beluga whale, *Delphinapterus leucas*, p. 195-235. *In* J. W. Lentfer (ed.), Selected Marine Mammals of Alaska. Species Accounts with Research and Management Recommendations. Marine Mammal Commission, Washington, DC.

Heide-Jørgensen, M., K. Laidre, D. Borchers, T. Marques, H. Stern, and M. Simon. 2010. The effect of sea-ice loss on beluga whales (*Delphinapterus leucas*) in West Greenland. Polar Res. 29:198-208. DOI: dx.doi.org/10.1111/j.1751-8369.2009.00142.x .

Helker, V. T., M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. 2017. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2011-2015. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-354, 112 p.

Hobbs, R. C., K. L. Laidre, D. J. Vos, B. A. Mahoney, and M. Eagleton. 2005. Movements and area use of belugas, *Delphinapterus leucas*, in a subarctic Alaskan estuary. Arctic 58(4):331-340.

Johannessen, O. M., L. Bengtson, M. W. Miles, S. I. Kuzmina, V. A. Semenov, G. V. Alexseev, A. P. Nagurnyi, V. F. Zakharov, L. P. Bobylev, L. H. Pettersson, K. Hasselmann, and H. P. Cattle. 2004. Arctic climate change: observed and modeled temperature and sea-ice variability. Tellus 56A:328-341.

Laidre, K. L., I. Stirling, L. Lowry, Ø. Wiig, M. P. Heide-Jørgensen, and S. Ferguson. 2008. Quantifying the sensitivity of arctic marine mammals to climate-induced habitat change. Ecol. Appl. 18(2):S97-S125.

Lowry, L. F. 1985. The belukha whale (*Delphinapterus leucas*), p. 3-13. *In* J. J. Burns, K. J. Frost, and L. F. Lowry (eds.), Marine mammals species accounts. Alaska Department of Fish and Game, Game Tech. Bull. 7.

Lowry, L. F., D. P. DeMaster, and K. J. Frost. 1999. Alaska Beluga Whale Committee surveys of beluga whales in the eastern Bering Sea, 1992-1995. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/51/SM34). 22 p.

Lowry, L., G. O'Corry-Crowe, and D. Goodman, D. 2006. *Delphinapterus leucas* (Cook Inlet population). In IUCN 2006. 2006 IUCN Red List of Threatened Species.

Lowry, L. F., K. J. Frost, A. Zerbini, D. DeMaster, and R. R. Reeves. 2008. Trend in aerial counts of beluga or white whales (*Delphinapterus leucas*) in Bristol Bay, Alaska. J. Cetacean Res. Manage. 10(3):201-207.

Lowry, L. F., A. Zerbini, K. J. Frost, D. P. DeMaster, and R. C. Hobbs. 2017. Development of an abundance estimate for the Eastern Bering Sea stock of beluga whales (*Delphinapterus leucas*). J. Cetacean Res. Manage. 16:39-47.

Moore, S. E., K. E. W. Shelden, L. K. Litzky, B. A. Mahoney, and D. J. Rugh. 2000. Beluga, *Delphinapterus leucas*, habitat associations in Cook Inlet, Alaska. Mar. Fish. Rev. 62(3):60-80.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed June 2018.

O'Corry-Crowe, G. M., R. S. Suydam, A. Rosenberg, K. J. Frost, and A. E. Dizon. 1997. Phylogeography, population structure and dispersal patterns of the beluga whale *Delphinapterus leucas* in the western Nearctic revealed by mitochondrial DNA. Mol. Ecol. 6:955-970.

O'Corry-Crowe G., A. R. Mahoney, R. Suydam, L. Quakenbush A. Whiting, L. Lowry, and L. Harwood. 2016. Genetic profiling links changing sea-ice to shifting beluga whale migration patterns. Biol. Lett. 12:20160404. DOI: dx.doi.org/10.1098/rsbl.2016.0404 .

Quakenbush, L. 2003. Summer movements of beluga whales captured in the Kvichak River in May 2002 and 2003. Alaska Beluga Whale Committee Report 03-03. 15 p.

Shelden, K. E. W., K. T. Goetz, D. J. Rugh, D. G. Calkins, B. A. Mahoney, and R. C. Hobbs. 2015. Spatio-temporal changes in beluga whale, *Delphinapterus leucas*, distribution: results from aerial surveys (1977-2014), opportunistic sightings (1975-2014), and satellite tagging (1999-2003) in Cook Inlet, Alaska. Mar. Fish. Rev. 77(2):1-31 + appendices. DOI: dx.doi.org/10.7755/MFR.77.2.1 .

Suydam, R. S. 2009. Age, growth, reproduction, and movements of beluga whales (*Delphinapterus leucas*) from the eastern Chukchi Sea. Ph.D. Dissertation, University of Washington, School of Aquatic and Fishery Sciences, Seattle, WA.

Wade, P. R., and R. Angliss. 1997. Guidelines for assessing marine mammal stocks: report of the GAMMS Workshop April 3-5, 1996, Seattle, Washington. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-OPR-12, 93 p.

BELUGA WHALE (*Delphinapterus leucas*): Bristol Bay Stock

NOTE – April 2022: NMFS is evaluating whether scientific issues raised by co-management partners in November 2021 concerning the Eastern Bering Sea beluga whale Stock Assessment Report may also be applicable to the Bristol Bay beluga whale Stock Assessment Report. Any resulting changes will be reflected in a future Stock Assessment Report.

STOCK DEFINITION AND GEOGRAPHIC RANGE

Beluga whales are distributed throughout seasonally ice-covered arctic and subarctic waters of the Northern Hemisphere (Gurevich 1980). In ice-covered regions, they are closely associated with open leads and polynyas (Hazard 1988). In Alaska, depending on season and region, beluga whales may occur in both offshore and coastal waters, with summer concentrations in upper Cook Inlet, Bristol Bay, eastern Bering Sea (i.e., Yukon River Delta, Norton Sound), eastern Chukchi Sea, and Beaufort Sea (Mackenzie River Delta) (Hazard 1988, O'Corry-Crowe et al. 2018) (Fig. 1). Seasonal distribution is affected by ice cover, tidal conditions, access to prey, temperature, and human interaction (Lowry 1985). Data from satellite transmitters attached to beluga whales from the Beaufort Sea, Eastern Chukchi Sea, Eastern Bering Sea, and Bristol Bay stocks identify ranges that are relatively distinct month to month for these stocks' summering areas and autumn migratory routes (e.g., Hauser et al. 2014, Citta et al. 2017, Lowry et al. 2019). Transmitters that lasted through the winter showed that beluga whales from these summering areas overwinter in the Bering Sea; these stocks are not known to overlap in space and time (Suydam 2009, Citta et al. 2017, Lowry et al. 2019).

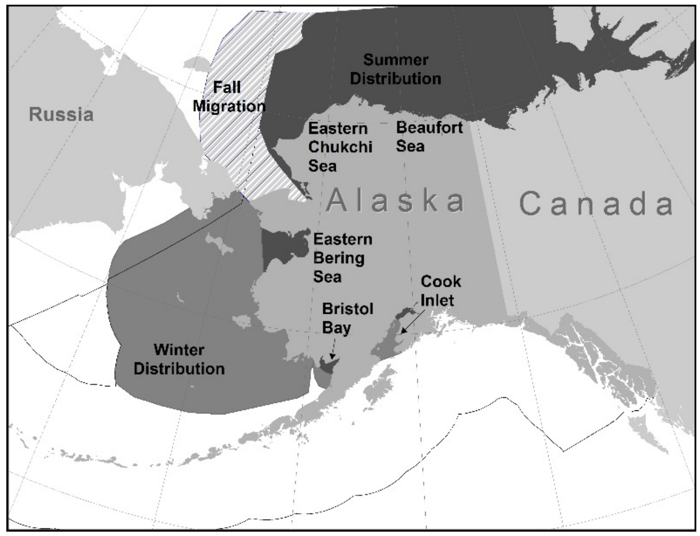

Figure 1. Approximate distribution for all five beluga whale stocks. The Beaufort Sea, Eastern Chukchi Sea, Eastern Bering Sea, and Bristol Bay beluga whale stocks summer in the Beaufort Sea (Beaufort Sea and Eastern Chukchi Sea stocks) and Bering Sea (Eastern Bering Sea and Bristol Bay stocks); they overwinter in the Bering Sea. The Bristol Bay and Cook Inlet beluga whale stocks show only small seasonal shifts in distribution, remaining in Bristol Bay and Cook Inlet, respectively, throughout the year. Summering areas are dark gray, wintering areas are lighter gray, and the hashed area is a region used by the Eastern Chukchi Sea and Beaufort Sea stocks for autumn migration. The U.S. Exclusive Economic Zone is delineated by a black line.

New genetic analyses have further defined five of the summering aggregations in the Bering, Chukchi, and Beaufort seas as follows: Bristol Bay, eastern Bering Sea (Norton Sound), eastern Chukchi Sea (Kasegaluk Lagoon), eastern Beaufort Sea (Mackenzie-Amundsen), and Gulf of Anadyr (Anadyr Bay) (O'Corry-Crowe et al. 2018). These genetic analyses, combined with new telemetry data, demonstrate that the demographically distinct summering aggregations return to discrete wintering areas and disperse and interbreed over limited distances but do not appear to interbreed extensively (O'Corry-Crowe et al. 2018).

The Beaufort Sea and Eastern Chukchi Sea stocks of beluga whales migrate between the Bering and Beaufort seas. Beaufort Sea beluga whales depart the Bering Sea in early spring, migrate through the Chukchi Sea and into the Canadian waters of the Beaufort Sea where they remain in the summer and fall, returning to the Bering Sea in late fall. Eastern Chukchi Sea beluga whales depart the Bering Sea in late spring and early summer, migrate through the Chukchi Sea and into the western Beaufort Sea where they remain in the summer, returning to the Bering Sea in the fall. The Eastern Bering Sea beluga whale stock remains in the Bering Sea but migrates south

near Bristol Bay in winter and returns north to Norton Sound and the mouth of the Yukon River in summer (Suydam 2009, Hauser et al. 2014, Citta et al. 2017, Lowry et al. 2019). Beluga whales tagged in Bristol Bay (Quakenbush 2003; Citta et al. 2016, 2017) and Cook Inlet (Goetz et al. 2012; Shelden et al. 2015, 2018; Lowry et al. 2019) remain in those areas throughout the year, showing only small seasonal shifts in distribution.

Summer movement patterns of Bristol Bay beluga whales were determined from satellite-linked tags deployed on 10 animals in the Kvichak River in 2002 and 2003 and 22 whales in the Nushagak River from 2006 to 2011 (Citta et al. 2016). Those whales used the shallow upper portions of Kvichak and Nushagak bays between May and August (Quakenbush 2003) and remained in the nearshore waters of Bristol Bay throughout September and October (Quakenbush and Citta 2006). Data from two beluga whales whose tags transmitted into December and January showed they were in Nushagak and Kvichak bays, suggesting that some beluga whales do not leave the nearshore waters of Bristol Bay during the winter (Citta et al. 2017). Tags attached to whales in 2012, 2013, 2014, and 2016 confirmed these movement observations (NMFS and Alaska SeaLife Center, unpubl. data; https://www.fisheries.noaa.gov/resource/document/2014-cook-inlet-beluga-whale-science-conference-presentations, accessed December 2020).

The following information was considered in classifying beluga whale stock structure based on the Dizon et al. (1992) phylogeographic approach: 1) Distributional data: geographic distribution discontinuous in summer (Frost and Lowry 1990); 2) Population response data: distinct population trends among regions occupied in summering areas (O'Corry-Crowe et al. 2018); 3) Phenotypic data: unknown; and 4) Genotypic data: mitochondrial DNA analyses indicate distinct differences among the five summering areas (O'Corry-Crowe et al. 2018). Based on this information, five beluga whale stocks are recognized within U.S. waters: 1) Cook Inlet, 2) Bristol Bay (Fig. 1), 3) Eastern Bering Sea, 4) Eastern Chukchi Sea, and 5) Beaufort Sea.

POPULATION SIZE

The sources of information to estimate abundance for beluga whales in the waters of western and northern Alaska have included both opportunistic and systematic observations. Frost and Lowry (1990) compiled data collected from aerial surveys conducted in Bristol Bay between 1978 and 1987 that were specifically designed to estimate the beluga whale population. Surveys focused on areas where beluga whales had been found to aggregate during the summer. Frost and Lowry (1990) reported an estimate of 1,000-1,500 whales for Bristol Bay, similar to that reported by Seaman et al. (1985). In 1994, the abundance was estimated at 1,555 beluga whales (Lowry and Frost 1998). That estimate was based on a maximum count of 503 whales, which was corrected using radio-telemetry data for the proportion of whales that were diving and thus not visible at the surface (2.62: Frost and Lowry 1995) and for the proportion of newborns and yearlings not observed due to their small size and dark coloration (1.18: Brodie 1971). The Alaska Department of Fish and Game and the Alaska Beluga Whale Committee (ABWC) conducted aerial beluga whale surveys in Bristol Bay in 1999, 2000, 2004, 2005, and 2016, with average counts of 444, 421, 609, 637, and 660 whales, respectively (Lowry et al. 2008, Lowry et al. 2019). The data from the 2004 and 2005 surveys result in an average count of 623 (coefficient of variation (CV) = 0.25) and, using the correction values above, a population estimate of 1,926 beluga whales (623 × 2.62 × 1.18). Using the count from the 2016 surveys and the correction values that have been applied in the past yields an estimated abundance of 2,040 beluga whales (CV = 0.26) in 2016 (660 × 2.62 × 1.18).

The Bristol Bay stock of beluga whales is genetically distinct. Citta et al. (2018) used a POPAN Jolly-Seber model to estimate abundance using genetic mark-recapture methods. Of the 516 individual whales identified from skin biopsies collected between 2002 and 2011, 75 beluga whales were identified (recaptured) in separate years, resulting in an estimate of 1,928 beluga whales (95% CI: 1,611-2,337), not including calves, which were not sampled (Citta et al. 2018).

Minimum Population Estimate

The survey technique used for estimating the abundance of beluga whales in this stock is a direct count which incorporates correction factors for submerged whales and calves. The abundance estimate is thought to be conservative because no correction was made for whales that were at the surface but were missed by the observers (Lowry and Frost 1998). The minimum population estimate (N_{MIN}) for the Bristol Bay beluga whale stock is calculated according to Equation 1 from the potential biological removal (PBR) guidelines (NMFS 2016): $N_{MIN} = N/\exp(0.842\times[\ln(1+[CV(N)]^2)]^{½})$. Using the population estimate (N) from the 2016 surveys of 2,040 and the CV of 0.26, N_{MIN} for the Bristol Bay stock is 1,645 beluga whales.

Current Population Trend

After a period of growth observed during surveys conducted from 1993 to 2005 where the population increased by 65% (Lowry et al. 2008), the estimate obtained from a survey conducted in 2016 was similar to those in 2004 and 2005 (Citta et al. 2019). Citta et al. (2019) concluded that population growth has now slowed or ceased entirely.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

The estimated rate of increase in beluga whale abundance in Bristol Bay from 1993 to 2005 was 4.8% per year (95% CI: 2.1%-7.5%: Lowry et al. 2008); however, because this estimate has a large CV, the default cetacean maximum net productivity rate (R_{MAX}) of 4% (NMFS 2016) will be used for this stock. It is not clear why the stock increased at this rate between 1993 and 2005, but possibilities include recovery from research kills in the 1960s, a reduction in subsistence harvests, and a delayed response to increases in salmon stocks (Lowry et al. 2008). Genetic mark-recapture estimates that include whales sampled between 2002 and 2011 and the most recent aerial estimate from 2016 suggest the population growth previously observed has slowed or ceased (Citta et al. 2019, Lowry et al. 2019).

POTENTIAL BIOLOGICAL REMOVAL

PBR is defined as the product of the minimum population estimate, one-half the maximum estimated net productivity rate, and a recovery factor: PBR = $N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 1.0, a value that may be used for stocks that are not known to be decreasing and are taken primarily by aboriginal subsistence hunters, provided there have not been recent increases in the levels of takes (NMFS 2016, Lowry et al. 2019). Using the N_{MIN} of 1,645, calculated from the 2016 aerial survey estimate of 2,040 (CV = 0.26), PBR for this stock is 33 beluga whales (1,645 × 0.02 × 1.0).

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2014 and 2018 is listed, by marine mammal stock, in Young et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports. The minimum estimated mean annual level of human-caused mortality and serious injury for Bristol Bay beluga whales between 2014 and 2018 is 19 beluga whales: 19 in subsistence takes by Alaska Natives (including one take in a subsistence salmon set gillnet fishery), and 0.2 incidental to Marine Mammal Protection Act (MMPA)-authorized research. Estimates of mortality and serious injury incidental to Bristol Bay fisheries are likely to be underestimated because observers have never monitored the Bristol Bay commercial salmon set gillnet and drift gillnet fisheries, there is substantial participation in the subsistence salmon gillnet fishery in Bristol Bay but no established protocol for reporting incidental takes in non-commercial fisheries to NMFS, and beluga whales taken incidental to personal-use or commercial salmon fisheries may be used by Alaska Natives for subsistence purposes and may be reported as subsistence takes. Potential threats most likely to result in direct human-caused mortality or serious injury of this stock include entanglement in fishing gear.

Fisheries Information

Information for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is available in Appendix 3 of the Alaska Stock Assessment Reports (observer coverage) and in the NMFS List of Fisheries (LOF) and the fact sheets linked to fishery names in the LOF (observer coverage and reported incidental takes of marine mammals: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-list-fisheries, accessed December 2020).

No beluga whale mortality or serious injury was observed incidental to U.S. commercial fisheries in Alaska between 2014 and 2018.

The Bristol Bay commercial salmon set gillnet and drift gillnet fisheries combined had 2,841 active permits listed in the NMFS 2019 LOF (https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-list-fisheries, accessed December 2020). These fisheries are known to have caused mortality of Bristol Bay beluga whales (Frost et al. 1984). However, complete data on incidental takes of this stock are not available because there have never been observer programs in these commercial fisheries, and there is no reporting requirement for takes in personal-use fisheries.

It should be noted that in western Alaska, beluga whales taken incidental to personal-use or commercial salmon fisheries may be used by Alaska Natives for subsistence purposes and may be included in the subsistence harvest data reported below. For example, one beluga whale that entangled in a Bristol Bay subsistence salmon set gillnet in 2014 was known to be used for subsistence purposes and is included in the subsistence harvest data for 2014-2018 (Table 1; ABWC, unpubl. data; Young et al. 2020).

The minimum mean annual mortality and serious injury rate incidental to U.S. commercial fisheries between 2014 and 2018 is zero beluga whales from this stock; however, a reliable estimate of the mortality rate incidental to U.S. commercial fisheries is not available because most coastal commercial fisheries that overlap with this stock have never been observed.

Alaska Native Subsistence/Harvest Information

NMFS signed an agreement with the ABWC (2000) to co-manage western Alaska beluga whale populations in the Bering Sea (including Bristol Bay), Chukchi Sea, and Beaufort Sea. This co-management agreement promotes full and equal participation by Alaska Natives in decisions affecting the subsistence management of beluga whales (to the maximum extent allowed by law) as a tool for conserving beluga whale populations in Alaska (https://www.fisheries.noaa.gov/alaska/marine-mammal-protection/co-management-marine-mammals-alaska, accessed December 2020).

The subsistence take of Bristol Bay beluga whales is reported by the ABWC. The most recent subsistence harvest estimates for the Bristol Bay stock are provided in Table 1 (ABWC, unpubl. data, 2020). The annual subsistence take by Alaska Native hunters averaged 19 Bristol Bay beluga whales landed between 2014 and 2018.

Table 1. Summary of Bristol Bay beluga whales landed by Alaska Native subsistence hunters between 2014 and 2018 (ABWC, unpubl. data, 2020). These are minimum estimates of the total number of beluga whales taken, because not all landed whales and struck and lost whales are consistently reported.

Year	**Number landed**	**Number struck and lost**	**Total (landed + struck and lost)**
2014	27	0	27
2015	22	2	24
2016	19	1	20
2017	10	no data	10
2018	11	2	13
Mean annual number (landed + struck and lost)			19

Other Mortality

Mortality and serious injury may occasionally occur incidental to marine mammal research activities authorized under MMPA permits issued to a variety of government, academic, and other research organizations. In 2016 there was a report of one beluga whale mortality incidental to research on the Bristol Bay stock (Table 2; Young et al. 2020), resulting in a mean annual mortality and serious injury rate of 0.2 beluga whales from this stock between 2014 and 2018.

Table 2. Summary of Bristol Bay beluga whale mortality and serious injury, by year and type, reported to the NMFS Office of Protected Resources between 2014 and 2018 (Young et al. 2020). Beluga whales with non-serious injuries were excluded.

Cause of Injury	**2014**	**2015**	**2016**	**2017**	**2018**	**Mean annual mortality**
Incidental to MMPA-authorized research	0	0	1	0	0	0.2
Total incidental to MMPA-authorized research						0.2

STATUS OF STOCK

No fishery-related mortality or serious injury has been reported for the Bristol Bay beluga whale stock between 2014 to 2018; therefore, the mean annual mortality and serious injury rate incidental to U.S. commercial fisheries can be considered insignificant and approaching a zero mortality and serious injury rate. Bristol Bay beluga whales are not designated as depleted under the MMPA or listed as threatened or endangered under the

Endangered Species Act. Because the minimum estimate of the mean annual human-caused mortality and serious injury rate (19 beluga whales) is less than the PBR (33), the Bristol Bay stock of beluga whales is not classified as a strategic stock. However, as noted previously, the estimate of fisheries-related mortality and serious injury is likely underestimated.

There are key uncertainties in the assessment of the Bristol Bay stock of beluga whales. The abundance is based on count data that are corrected for the proportion of whales that are diving and the proportion of newborns and yearlings not observed because of their size and coloration; however, the counts are not corrected for whales which are at the surface but missed by the observers. Although the apparent population rate of increase was quite high from 1993 to 2005, which may indicate that the population was depleted and reduced human-related mortality and serious injury allowed an increase, most coastal commercial fisheries that overlap with this stock have never been observed. Therefore, the mortality and serious injury of Bristol Bay beluga whales in commercial fisheries could be underestimated. Coastal subsistence fisheries for salmon will occasionally cause incidental mortality or serious injury of a beluga whale; these incidental takes used for subsistence purposes may not always be reported to the ABWC for inclusion in the subsistence harvest estimates for this stock.

HABITAT CONCERNS

Evidence indicates that climate is changing significantly in the Bristol Bay region. One result of the change is a reduction in the extent and duration of sea ice in the winter (ACIA 2004, Johannessen et al. 2004). These changes are likely to affect marine mammal species in Bristol Bay. Ice-associated animals, such as the beluga whale, are sensitive to changes in weather, sea-surface temperatures, and sea-ice extent, and the concomitant effect on prey availability. Decreases in seasonal sea ice may also increase the risk of killer whale predation (O'Corry-Crowe et al. 2016). There are insufficient data to make reliable predictions of the effects of climate change on beluga whales; however, Laidre et al. (2008) and Heide-Jørgensen et al. (2010) concluded that on a worldwide basis beluga whales were likely to be less sensitive to climate change in general than other arctic cetaceans because of their wide distribution and flexible behavior. However, local changes in distribution and seasonal behavior are likely to occur (Hauser et al. 2017). Increased human activity in the Bristol Bay region, including increased oil and gas exploration and development and increased nearshore development and mining activities near large tributaries, has the potential to impact habitat for beluga whales (Lowry et al. 2006, Norman et al. 2015). However, predicting the type and magnitude of these impacts is difficult.

In all cases, increased human activities in or near coastal areas of Bristol Bay will increase anthropogenic noise in the water, which has been shown to have negative impacts on cetacean feeding and communication (Norman et al. 2015, Small et al. 2017). Studies of beluga whales in Bristol Bay found that some individuals have "sensitive hearing that approaches the lower levels of noise within their habitat" (Mooney et al. 2018). This may be a result of living in an acoustically quiet environment, which allows for a large dynamic range of hearing. However, if the ambient noise were to increase due to increased anthropogenic activities, masking of calls may occur. This is a particular concern for cow/calf pairs because calves have been shown to vocalize at lower amplitudes than their mothers (Vergara 2019). If ambient or anthropogenic noise levels increase, cow/calf pairs may lose the ability to communicate effectively. Additionally, masking can reduce the range of acoustic detection of prey and communication in cooperative feeding.

CITATIONS

Arctic Climate Impact Assessment (ACIA). 2004. Impacts of a Warming Arctic: Arctic Climate Impact Assessment. Cambridge University Press, Cambridge, UK.

Brodie, P. F. 1971. A reconsideration of aspects of growth, reproduction, and behavior of the white whale with reference to the Cumberland Sound, Baffin Island, population. J. Fish. Res. Bd. Can. 28:1309-1318.

Citta, J. J., L. T. Quakenbush, K. J. Frost, L. Lowry, R. C. Hobbs, and H. Aderman. 2016. Movements of beluga whales (*Delphinapterus leucas*) in Bristol Bay, Alaska. Mar. Mammal Sci. 32:1272-1298. DOI: dx.doi.org/10.1111/mms.12337 .

Citta, J. J., P. Richard, L. F. Lowry, G. O'Corry-Crowe, M. Marcoux, R. Suydam, L. T. Quakenbush, R. C. Hobbs, D. I. Litovka, K. J. Frost, T. Gray, J. Orr, B. Tinker, H. Aderman, and M. L. Druckenmiller. 2017. Satellite telemetry reveals population specific winter ranges of beluga whales in the Bering Sea. Mar. Mammal Sci. 33:236-250. DOI: dx.doi.org/10.1111/mms.12357 .

Citta, J. J., G. O'Corry-Crowe, L. T. Quakenbush, A. L. Bryan, T. Ferrer, M. J. Olson, R. C. Hobbs, and B. Potgieter. 2018. Assessing the abundance of Bristol Bay belugas with genetic mark-recapture methods. Mar. Mammal Sci. 34(3):666-686.

Citta, J. J., K. J. Frost, and L. Quakenbush. 2019. Aerial surveys of Bristol Bay beluga whales, *Delphinapterus leucas*, in 2016. Mar. Fish. Rev. 81(3-4):98-104.

Dizon, A. E., C. Lockyer, W. F. Perrin, D. P. DeMaster, and J. Sisson. 1992. Rethinking the stock concept: a phylogeographic approach. Conserv. Biol. 6:24-36.

Frost, K. J., and L. F. Lowry. 1990. Distribution, abundance, and movements of beluga whales, *Delphinapterus leucas*, in coastal waters of western Alaska, p. 39-57. *In* T. G. Smith, D. J. St. Aubin, and J. R. Geraci (eds.), Advances in research on the beluga whale, *Delphinapterus leucas*. Can. Bull. Fish. Aquat. Sci. 224.

Frost, K. J., and L. F. Lowry. 1995. Radio tag based correction factors for use in beluga whale population estimates. Working paper for Alaska Beluga Whale Committee Scientific Workshop, Anchorage, AK, 5-7 April 1995. 12 p.

Frost, K. J., L. F. Lowry, and R. R. Nelson. 1984. Belukha whale studies in Bristol Bay, Alaska, p. 187-200. *In* Proceedings of the workshop on biological interactions among marine mammals and commercial fisheries in the southeastern Bering Sea, October 18-21, 1983, Anchorage AK. Alaska Sea Grant Report 84-1.

Goetz, K. T., P. W. Robinson, R. C. Hobbs, K. L. Laidre, L. A. Huckstadt, and K. E. W. Shelden. 2012. Movement and dive behavior of beluga whales in Cook Inlet, Alaska. AFSC Processed Rep. 2012-03, 40 p. Alaska Fisheries Science Center, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Gurevich, V. S. 1980. Worldwide distribution and migration patterns of the white whale (beluga), *Delphinapterus leucas*. Rep. Int. Whal. Comm. 30:465-480.

Hauser, D. D. W., K. L. Laidre, R. S. Suydam, and P. R. Richard. 2014. Population-specific home ranges and migration timing of Pacific Arctic beluga whales (*Delphinapterus leucas*). Polar Biol. 37:1171-1183. DOI: dx.doi.org/10.1007/s00300-014-1510-1 .

Hauser, D. D. W., K. L. Laidre, K. M. Stafford, H. L. Stern, R. S. Suydam, and P. R. Richard. 2017. Decadal shifts in autumn migration timing by Pacific Arctic beluga whales are related to delayed annual sea ice formation. Glob. Change Biol. 23:2206-2217. DOI: dx.doi.org/10.1111/gcb.13564 .

Hazard, K. 1988. Beluga whale, *Delphinapterus leucas*, p. 195-235. *In* J. W. Lentfer (ed.), Selected Marine Mammals of Alaska. Species Accounts with Research and Management Recommendations. Marine Mammal Commission, Washington, DC.

Heide-Jørgensen, M., K. Laidre, D. Borchers, T. Marques, H. Stern, and M. Simon. 2010. The effect of sea-ice loss on beluga whales (*Delphinapterus leucas*) in West Greenland. Polar Res. 29:198-208. DOI: dx.doi.org/10.1111/j.1751-8369.2009.00142.x .

Johannessen, O. M., L. Bengtson, M. W. Miles, S. I. Kuzmina, V. A. Semenov, G. V. Alexseev, A. P. Nagurnyi, V. F. Zakharov, L. P. Bobylev, L. H. Pettersson, K. Hasselmann, and H. P. Cattle. 2004. Arctic climate change: observed and modeled temperature and sea-ice variability. Tellus 56A:328-341.

Laidre, K. L., I. Stirling, L. Lowry, Ø. Wiig, M. P. Heide-Jørgensen, and S. Ferguson. 2008. Quantifying the sensitivity of arctic marine mammals to climate-induced habitat change. Ecol. Appl. 18(2):S97-S125.

Lowry, L. F. 1985. The belukha whale (*Delphinapterus leucas*), p. 3-13. *In* J. J. Burns, K. J. Frost, and L. F. Lowry (eds.), Marine mammals species accounts. Alaska Department of Fish and Game, Game Tech. Bull. 7.

Lowry, L. F., and K. J. Frost. 1998. Alaska Beluga Whale Committee surveys of beluga whales in Bristol Bay, Alaska, 1993-1994. Alaska Beluga Whale Committee Report 98-3. 13 p.

Lowry, L., G. O'Corry-Crowe, and D. Goodman. 2006. *Delphinapterus leucas* (Cook Inlet population). *In* IUCN 2006. 2006 IUCN Red List of Threatened Species.

Lowry, L. F., K. J. Frost, A. Zerbini, D. DeMaster, and R. R. Reeves. 2008. Trend in aerial counts of beluga or white whales (*Delphinapterus leucas*) in Bristol Bay, Alaska, 1993-2005. J. Cetacean Res. Manage. 10(3):201-207.

Lowry, L. F., J. J. Citta, G. O'Corry-Crowe, L. T. Quakenbush, K. J. Frost, R. Suydam, R. C. Hobbs, and T. Gray. 2019. Distribution, abundance, harvest, and status of western Alaska beluga whale, *Delphinapterus leucas*, stocks. Mar. Fish. Rev. 81(3-4):54-71.

Mooney, T. A., M. Castellote, I. T. Jones, L. Quakenbush, R. Hobbs, E. Gaglione, and C. Goertz. 2018. Local acoustic habitat relative to hearing sensitivities in beluga whales (*Delphinapterus leucas*). J. Ecoacoustics 2:#QZD9Z5.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2020.

Norman, S. A., R. C. Hobbs, C. E. C. Goertz, K. A. Burek-Huntington, K. E. W. Shelden, W. A. Smith, and L. A. Beckett. 2015. Potential natural and anthropogenic impediments to the conservation and recovery of Cook Inlet beluga whales, *Delphinapterus leucas*. Mar. Fish. Rev. 77(2):89-105. DOI: dx.doi.org/10.7755/MFR.77.2.5 .

O'Corry-Crowe, G., A. R. Mahoney, R. Suydam, L. Quakenbush, A. Whiting, L. Lowry, and L. Harwood. 2016. Genetic profiling links changing sea-ice to shifting beluga whale migration patterns. Biol. Lett. 12:20160404. DOI: dx.doi.org/10.1098/rsbl.2016.0404 .

O'Corry-Crowe, G., R. Suydam, L. Quakenbush, B. Potgieter, L. Harwood, D. Litovka, T. Ferrer, J. Citta, V. Burkanov, K. Frost, and B. Mahoney. 2018. Migratory culture, population structure and stock identity in North Pacific beluga whales (*Delphinapterus leucas*). PLoS ONE 13(3):e0194201.

Quakenbush, L. 2003. Summer movements of beluga whales captured in the Kvichak River in May 2002 and 2003. Alaska Beluga Whale Committee Report 03-03. 15 p.

Quakenbush, L., and J. Citta. 2006. Fall movements of beluga whales captured in the Nushagak River, in September 2006. Alaska Beluga Whale Committee Report. 9 p.

Seaman, G. A., K. J. Frost, and L. F. Lowry. 1985. Investigations of belukha whales in coastal waters of western and northern Alaska. Part I. Distribution, abundance and movements. U.S. Dep. Commer., NOAA, OCSEAP Final Report 56:153-220. Available from NOAA-OMA-OAD, Alaska Office, 701 C. Street, P.O. Box 56, Anchorage, AK 99513.

Shelden, K. E. W., K. T. Goetz, D. J. Rugh, D. G. Calkins, B. A. Mahoney, and R. C. Hobbs. 2015. Spatio-temporal changes in beluga whale, *Delphinapterus leucas*, distribution: results from aerial surveys (1977-2014), opportunistic sightings (1975-2014), and satellite tagging (1999-2003) in Cook Inlet, Alaska. Mar. Fish. Rev. 77(2):1-31 + appendices. DOI: dx.doi.org/10.7755/MFR.77.2.1 .

Shelden, K. E. W., K. T. Goetz, R. C. Hobbs, L. K. Hoberecht, K. L. Laidre, B. A. Mahoney, T. L. McGuire, S. A. Norman, G. O'Corry-Crowe, D. J. Vos, G. M. Ylitalo, S. A. Mizroch, S. Atkinson, K. A. Burek-Huntington, and C. Garner. 2018. Beluga whale, *Delphinapterus leucas*, satellite-tagging and health assessments in Cook Inlet, Alaska, 1999 to 2002. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-369, 227 p.

Small, R. J., B. Brost, M. Hooten, M. Castellote, and J. Mondragon. 2017. Potential for spatial displacement of Cook Inlet beluga whales by anthropogenic noise in critical habitat. Endang. Species Res. 32:43-57. DOI: dx.doi.org/10.3354/esr00786 .

Suydam, R. S. 2009. Age, growth, reproduction, and movements of beluga whales *(Delphinapterus leucas*) from the eastern Chukchi Sea. Ph.D. Dissertation, University of Washington, School of Aquatic and Fishery Sciences, Seattle, WA.

Vergara, V., J. Wood, A. Ames, M. Mikus, V. Lesage, and R. Michaud. 2019. Mom, can you hear me? Impacts of underwater noise on mother-calf contact calls in endangered belugas (*Delphinapterus leucas*). Presented at the World Marine Mammal Conference, 9-12 December 2019, Barcelona, Spain.

Young, N. C., B. Delean, V. T. Helker, J. C. Freed, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2014-2018. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-413, 142 p.

BELUGA WHALE (*Delphinapterus leucas*): Cook Inlet Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Beluga whales are distributed throughout seasonally ice-covered arctic and subarctic waters of the Northern Hemisphere (Gurevich 1980). In ice-covered regions, they are closely associated with open leads and polynyas (Hazard 1988). In Alaska, depending on season and region, beluga whales may occur in both offshore and coastal waters, with summer concentrations in upper Cook Inlet, Bristol Bay, eastern Bering Sea (i.e., Yukon River Delta, Norton Sound), eastern Chukchi Sea, and Beaufort Sea (Mackenzie River Delta) (Hazard 1988, O'Corry-Crowe et al. 2018) (Fig. 1). Seasonal distribution is affected by ice cover, tidal conditions, access to prey, temperature, and human interaction (Lowry 1985). Data from satellite transmitters attached to beluga whales from the Beaufort Sea, Eastern Chukchi Sea, Eastern Bering Sea, and Bristol Bay stocks identify ranges that are relatively distinct month to month for these stocks' summering areas and autumn migratory routes (e.g., Hauser et al. 2014, Citta et al. 2017, Lowry et al. 2019). Transmitters that lasted through the winter showed that beluga whales from these summering areas overwinter in the Bering Sea; these stocks are not known to overlap in space and time in the Bering Sea (Suydam 2009, Citta et al. 2017, Lowry et al. 2019).

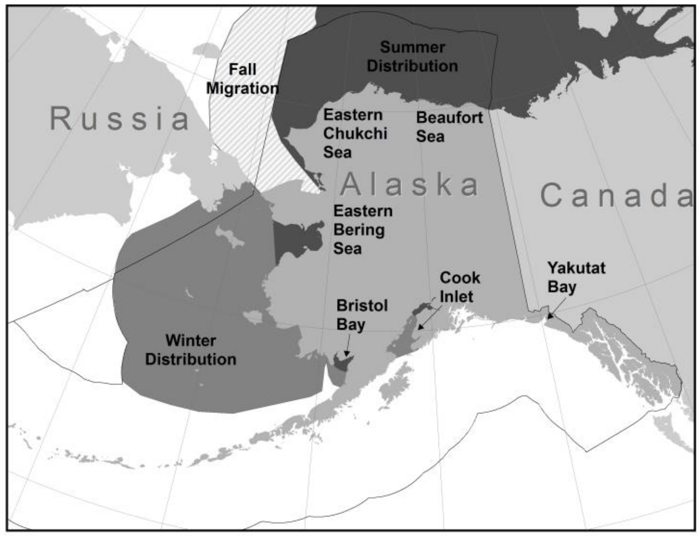

Figure 1. Approximate distribution for all five beluga whale stocks. The Beaufort Sea, Eastern Chukchi Sea, Eastern Bering Sea, and Bristol Bay beluga whale stocks summer in the Beaufort Sea (Beaufort Sea and Eastern Chukchi Sea stocks) and Bering Sea (Eastern Bering Sea and Bristol Bay stocks); they overwinter in the Bering Sea. The Bristol Bay and Cook Inlet beluga whale stocks show only small seasonal shifts in distribution, remaining in Bristol Bay and Cook Inlet, respectively, throughout the year. Summering areas are dark gray, wintering areas are lighter gray, and the hashed area is a region used by the Eastern Chukchi Sea and Beaufort Sea stocks for autumn migration. The U.S. Exclusive Economic Zone is delineated by a black line.

The Beaufort Sea and Eastern Chukchi Sea stocks of beluga whales migrate between the Bering and Beaufort seas. Beaufort Sea beluga whales depart the Bering Sea in early spring, migrate through the Chukchi Sea and into the Canadian waters of the Beaufort Sea where they remain in the summer and fall, returning to the Bering Sea in late fall. Eastern Chukchi Sea beluga whales depart the Bering Sea in late spring and early summer, migrate through the Chukchi Sea and into the western Beaufort Sea where they remain in the summer, returning to the Bering Sea in the fall. The Eastern Bering Sea beluga whale stock remains in the Bering Sea but migrates south near Bristol Bay in winter and returns north to Norton Sound and the mouth of the Yukon River in summer (Suydam 2009, Hauser et al. 2014, Citta et al. 2017, Lowry et al. 2019). Beluga whales tagged in Bristol Bay (Quakenbush 2003; Citta et al. 2016, 2017) and Cook Inlet (Goetz et al. 2012a; Shelden et al. 2015, 2018; Lowry et al. 2019) remain in those areas throughout the year, showing only small seasonal shifts in distribution.

The following information was considered in classifying beluga whale stock structure based on the Dizon et al. (1992) phylogeographic approach: 1) Distributional data: geographic distribution discontinuous in summer (Frost and Lowry 1990); 2) Population response data: distinct population trends among regions occupied in summering areas (O'Corry-Crowe et al. 2018); 3) Phenotypic data: unknown; and 4) Genotypic data: mitochondrial DNA analyses indicate distinct differences among the five summering areas (O'Corry-Crowe et al. 2018). Based on this information, five beluga whale stocks are recognized within U.S. waters: 1) Cook Inlet (Fig. 1), 2) Bristol Bay, 3) Eastern Bering Sea, 4) Eastern Chukchi Sea, and 5) Beaufort Sea.

During ice-free months, Cook Inlet beluga whales are often concentrated near river mouths (Shelden et al. 2015). The fall-winter-spring distribution of this stock is not fully understood; however, there is evidence that most whales in this population inhabit upper Cook Inlet year-round (Lammers et al. 2013, Castellote et al. 2015, Shelden et al. 2015). From 1999 to 2002, satellite tags were attached to a total of 18 Cook Inlet beluga whales to determine their movement patterns (Goetz et al. 2012a; Shelden et al. 2015, 2018). All tagged beluga whales remained in Cook Inlet, primarily in the upper inlet north of East and West Forelands, with brief trips to the lower inlet (Shelden et al. 2015, 2018).

A review of all marine mammal surveys and anecdotal sightings in the northern Gulf of Alaska between 1936 and 2000 found only 28 beluga whale sightings, indicating that very few beluga whales occurred in the Gulf of Alaska outside Cook Inlet (Laidre et al. 2000). Yakutat Bay is the only area in the Gulf of Alaska outside of Cook Inlet where multiple beluga whale sightings have occurred (Laidre et al. 2000, Lucey et al. 2015, O'Corry-Crowe et al. 2015). Based on genetic analyses, traditional ecological knowledge, and observations by fishermen and others, the Yakutat Bay beluga whales likely represent a small, resident group (fewer than 20 whales) that has been observed year round and is reproductively separated from Cook Inlet (Lucey et al. 2015, O'Corry-Crowe et al. 2015). Furthermore, this group in Yakutat Bay appears to be showing signs of inbreeding and low diversity due to their isolation and small numbers (O'Corry-Crowe et al. 2015). Although the beluga whales in Yakutat Bay are not included in the Cook Inlet Distinct Population Segment (DPS) of beluga whales under the Endangered Species Act (ESA), they are considered part of the depleted Cook Inlet stock under the Marine Mammal Protection Act (MMPA) (50 CFR 216.15; 75 FR 12498, 16 March 2010) because insufficient information was available to identify Yakutat Bay beluga whales as a separate population when Cook Inlet beluga whales were designated as depleted under the MMPA. Thus, Yakutat Bay beluga whales remain part of the Cook Inlet stock, are designated as depleted, and are provided the same protections as the Cook Inlet stock, including hunting regulations/restrictions.

POPULATION SIZE

Aerial surveys during June documented the distribution and abundance of Cook Inlet beluga whales and were conducted by NMFS each year from 1994 to 2012 (Rugh et al. 2000, 2005; Shelden et al. 2013), after which NMFS began biennial surveys in 2014 (Shelden et al. 2019) (Fig. 2). NMFS changed to a biennial survey schedule after analysis showed there would be little reduction in the ability to detect a trend given the current growth rate of the population (Hobbs 2013).

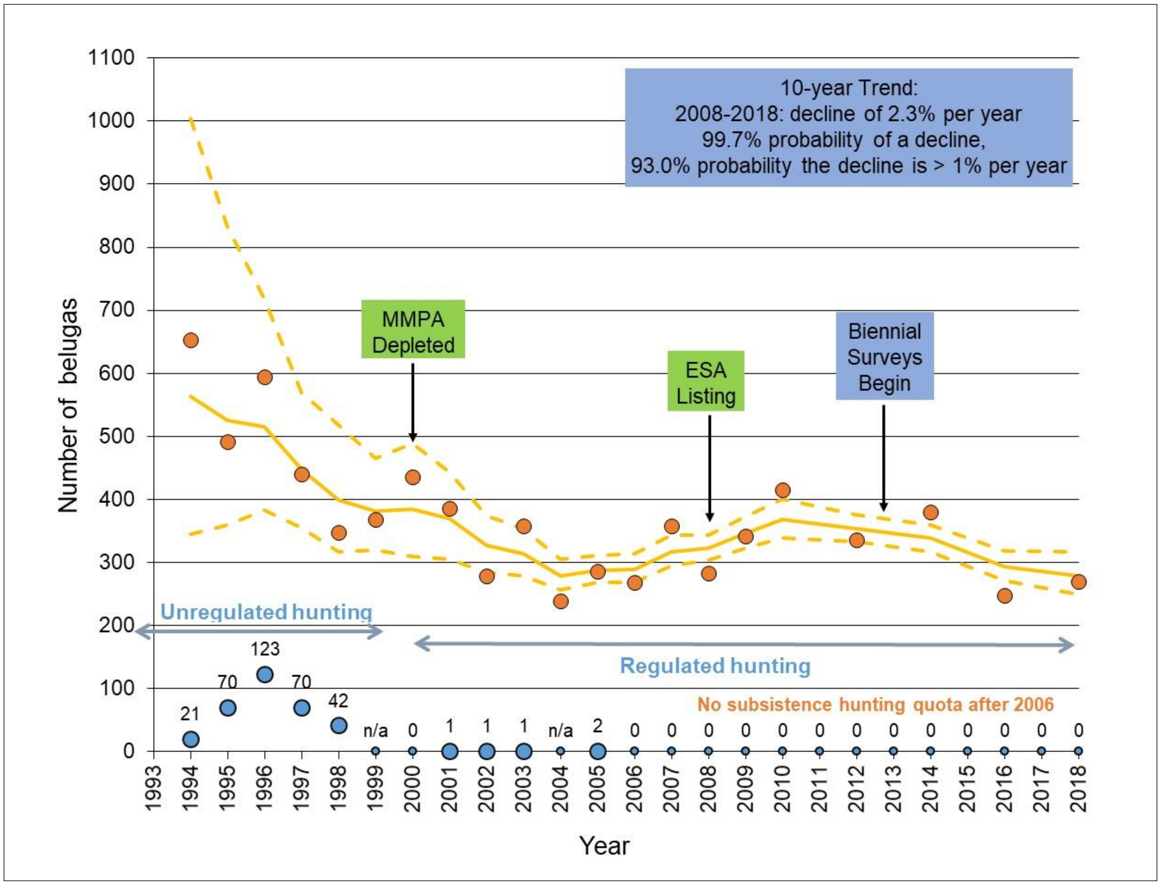

Figure 2. Annual abundance estimates (orange circles) of beluga whales in Cook Inlet, Alaska, 1994-2018 (Hobbs et al. 2015a, Wade et al. 2019). Blue circles show reported removals (landed plus struck and lost) during the Alaska Native subsistence harvest. A struck and lost average was calculated by the Cook Inlet Marine Mammal Council (CIMMC) for 1996, 1997, and 1998. The solid line is a weighted moving average of the abundance estimates that represents the smoothed trend of the population through time. Dashed lines above and below the solid line are 95% probability intervals around the smoothed trend line.

The survey covers all coastal areas and all river mouths and deltas in Cook Inlet in early June. The surveys are designed with the intention of detecting all substantially-sized beluga whale groups in the upper inlet. When beluga whale groups are detected, the group sizes are estimated by visual counts by observers or from video data recorded of the groups. The group-size estimates are summed across all detected groups to calculate an abundance estimate from each day's survey. Daily estimates from all survey days considered acceptable are combined to form an annual estimate of abundance for the population.

The method used for estimating group size from video data requires estimating multiple correction factors for visibility bias (Hobbs et al. 2000, 2015a). Following the June 2016 abundance survey, a major revision was made to the methods used to estimate group sizes from the survey data (Boyd et al. 2019). A new method was developed using a Bayesian statistical approach to group-size estimation; this new method was then applied to the 2004-2016 time series (Boyd et al. 2019). Wade et al. (2019) applied the same methodology to the 2018 survey data to estimate abundance for the 2018 survey. The new approach was designed to address the same four types of bias in the group-size estimation process as previous methods: 1) availability bias due to diving behavior; 2) proximity bias due to individuals concealed by another individual in the video data; 3) perception bias due to individuals not detected because of small image size in the video data; and 4) individual observer bias in visual estimates of group size (see Boyd et al. 2019 for a complete description of methods). The main advantages to the change in group-size estimation methods are as follows: (a) the Bayesian methods allow the variance in the parameter estimates to be fully propagated through the analysis (unlike the previous methods), and also allows for specification of distributions for some parameters, rather than just single values, to more completely consider uncertainty in the analyses; (b) for estimating the visibility bias correction factors (availability, proximity bias, and perception), the important assumption was added that the true group size was the same for all video passes of the same group (this assumption was not previously used in the analysis); (c) for availability bias, a prior distribution is specified for mean dive time for a beluga group; previously this was fixed at the single value of 24.1 seconds; and (d) for perception bias, the analysis now simultaneously estimates two distributions as part of the integrated analysis: 1) detection probability as a function of image size, and 2) the distribution of image sizes for all individuals; previously, this was done as a separate ad hoc analysis (Wade et al. 2019).

In addition to the new group-size estimation method, the revised abundance method controls for possible strong positive and negative outliers on single days (Wade et al. 2019). Strong negative outliers (days with very low abundance) can potentially happen when some groups are not seen. Strong positive outliers (days with very high abundance) can potentially happen when the whales occur in one or more very large groups, and the video group-size estimation process becomes difficult, with wide probability intervals. Previously (i.e., Hobbs et al. 2015a), the annual estimate of abundance was calculated as the average of three or more days, excluding a day's estimate if it was less than approximately 60% of the highest day. Now, the annual abundance is calculated as the median of all the daily abundance estimates, using all days with an acceptable survey day, defined objectively by weather/sighting conditions and spatial coverage. Using the median lessens the influence of strong positive and negative outliers.

The point estimate of abundance for 2018, based on the median of all acceptable daily estimates in 2018, is 269 beluga whales (coefficient of variation (CV) = 0.103, 95% probability interval (PI): 227 to 333). The best estimate of current abundance is based on a weighted average from the last three annual abundance estimates (2014, 2016, and 2018), giving more weight to the more recent estimates. From that weighted average, the best estimate of abundance for the Cook Inlet beluga population in 2018 is 279 (CV = 0.061, 95% PI: 250 to 317) (Wade et al. 2019).

Minimum Population Estimate

The minimum population estimate (N_{MIN}) is calculated as the 20th percentile of the best abundance estimate, according to the potential biological removal (PBR) guidelines (NMFS 2016a). In this case, N_{MIN} is calculated as the 20th percentile of the posterior distribution of the best estimate of abundance in 2018, which is 267 (Wade et al. 2019). Therefore, N_{MIN} for the Cook Inlet beluga whale stock is 267 beluga whales.

Current Population Trend

The annual abundance estimates for 1994 to 2018 are shown in Figure 2, along with a weighted moving average to show the smoothed trend over time. The population declined substantially during the period of unregulated hunting, with the peak hunting mortality reported in 1996 (123 whales) and the last year of substantial hunting mortality in 1998 (42 whales). Although only five whales were reported killed from hunting from 1999 to 2005, the population continued to decline until about 2004. The population showed an increase from 2005 to 2010 but has apparently declined since 2010. During the most recent 10-year time period (2008-2018), the estimated exponential trend in the abundance estimates is a decline of 2.3% per year (95% PI: -4.1% to -0.6%), with a 99.7% probability of a decline, and a 93.0% probability of a decline that is more than 1% per year (Wade et al. 2019).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate (R_{MAX}) is not available for the Cook Inlet beluga whale stock. Until additional data become available, the default cetacean maximum theoretical net productivity rate of 4% will be used for this stock (NMFS 2016a).

POTENTIAL BIOLOGICAL REMOVAL

PBR is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: PBR = $N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 0.1, the value for cetacean stocks that are listed as endangered (NMFS 2016a). Using the N_{MIN} of 267 beluga whales, the calculated PBR for this stock is 0.53 beluga whales ($267 \times 0.02 \times 0.1$).

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2014 and 2018 is listed, by marine mammal stock, in Young et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports. No human-caused mortality or serious injury of Cook Inlet beluga whales was confirmed between 2014 and 2018.

There are no observers in Cook Inlet fisheries, so the mean annual mortality and serious injury in commercial fisheries is unknown, although likely low, given that an observer program conducted in Cook Inlet in 1999-2000 did not observe mortality or serious injury of beluga whales (Manly 2006). Other potential threats most likely to result in direct human-caused mortality or serious injury of this stock include ship strikes.

Fisheries Information

Information for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is available in Appendix 3 of the Alaska Stock Assessment Reports (observer coverage) and in the NMFS List of Fisheries (LOF) and the fact sheets linked to fishery names in the LOF (observer coverage and reported incidental takes of marine mammals: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-list-fisheries, accessed December 2020).

Alaska Native Subsistence/Harvest Information

Subsistence harvest of Cook Inlet beluga whales is important to the Native Village of Tyonek and the Alaska Native subsistence hunter community in Anchorage. Between 1993 and 1998, the annual subsistence take ranged from 17 to more than 123 beluga whales (Fig. 2), including struck and lost whales (NMFS 2016b).

Following a significant decline in Cook Inlet beluga whale abundance estimates between 1994 and 1998, the Cook Inlet hunters voluntarily stopped hunting in 1999 and the Federal government took actions to conserve, protect, and prevent further declines in the abundance of these whales. Public Laws 106-31 (1999) and 106-553 (2000) established a moratorium on Cook Inlet beluga whale harvests unless such taking occurs pursuant to a cooperative agreement between NMFS and affected Alaska Native organizations. A cooperative agreement, also referred to as a co-management agreement, was not signed in 1999 and 2004. In December 2000, an administrative hearing was held to create interim harvest regulations for 2001 through 2004 (69 FR 17973, 6 April 2004). Three Cook Inlet beluga whales were killed under this interim harvest plan (2001-2004). In August 2004, an administrative hearing was held to create a long-term harvest plan, which allowed up to eight whales to be harvested between 2005 and 2009 (NMFS 2008). Two whales were harvested in 2005 and no whales were harvested in 2006. The long-term harvest plan was signed in 2008 and established a harvest level for a 5-year period, based on the average abundance in the previous 5-year period and the growth rate during the previous 10-year period (NMFS 2008). A harvest is not allowed if the previous 5-year average abundance is less than 350 beluga whales. Under the long-term harvest plan, the 5-year average abundance during the first review period (2003-2007) was 336 whales and, therefore, a harvest was not allowed during the subsequent 5-year period (2008-2012) (73 FR 60976, 15 October 2008). The average abundance of Cook Inlet beluga whales remained below 350 whales during the second review period (2008-2012); therefore, a harvest was not allowed for the subsequent 5-year period (2013-2017). NMFS changed to a biennial survey schedule after 2012, therefore, the 5-year average abundance is now based on either two or three surveys in a 5-year period. Hobbs (2013) showed that biennial rather than annual surveys may lead to higher variation in allowable harvest levels, but it is not expected to change the probability of recovery while using the algorithm that determines the allowable harvest level. The average abundance for a third review period (2013-2017), using the 2014 and 2016 estimates, is still below 350 whales (Wade et al. 2019), so a harvest is not allowed for the subsequent 5-year period (2018-2022).

Other Mortality

Reports from the NMFS Alaska Region marine mammal stranding network provide additional information on beluga whale mortality. Mortality related to live stranding events, where a beluga whale group strands as the tide recedes, has been regularly observed in upper Cook Inlet (Table 1). Improved reports include the number of live stranded beluga whales, as well as floating and beachcast carcasses (NMFS 2016b; https://www.fisheries.noaa.gov/resource/document/alaska-region-marine-mammal-stranding-summary, accessed December 2020). Most whales involved in live stranding events survive, although some associated deaths may not be observed if whales die later from live-stranding-related injuries (Vos and Shelden 2005, Burek-Huntington et al. 2015). Between 2014 and 2018, there were reports of approximately 79 beluga whales involved in three known live stranding events plus one suspected live stranding event with two associated deaths (Table 1; NMFS 2016b; NMFS, unpubl. data). A beluga whale calf that stranded alive in 2017 was sent to the Alaska SeaLife Center for rehabilitation and then transferred to SeaWorld in San Antonio, Texas, in 2018. In 2014, necropsy results from two whales found in Turnagain Arm suggested that a live stranding event contributed to their deaths as both had aspirated mud and water. No live stranding events were reported prior to the discovery of these dead whales, suggesting that not all live stranding events are observed (Table 1). Most live strandings occur in Knik Arm and

Turnagain Arm, which are shallow and have large tidal ranges (Turnagain Arm has the largest tidal range in the U.S., with a mean of 30 ft), strong currents, and extensive mudflats.

Table 1. Cook Inlet beluga whale strandings investigated by NMFS between 2014 and 2018 (NMFS 2016b; NMFS, unpubl. data).

Year	Floating and beachcast carcasses	Number of beluga whales per live stranding event (number of associated known or suspected resulting deaths)
2014	11	unknown[a] (2), 76+ (0)
2015	3	2 (0)
2016	8	0
2017	12	1[b]
2018	7	0
Total	41	79+ (2)

[a]A live stranding was not observed but was suspected based on necropsy results from two beluga whales found in Turnagain Arm (NMFS 2016b).
[b]A beluga whale calf that stranded alive in 2017 was sent to the Alaska SeaLife Center for rehabilitation and then transferred to SeaWorld in San Antonio, Texas, in 2018.

Another source of beluga whale mortality in Cook Inlet is predation by transient-type (mammal-eating) killer whales. Killer whale sightings were not well documented and were likely rare in the upper inlet prior to the mid-1980s. From 1982 through 2018, NMFS received 31 reports of killer whale sightings in upper Cook Inlet (north of East and West Forelands). Up to 12 beluga whale deaths, inlet-wide, were suspected to be a direct result of killer whale predation (NMFS 2016b). The last confirmed killer whale predation of a Cook Inlet beluga whale occurred in 2008 in Turnagain Arm. From 2014 through 2018, NMFS received two separate killer whale sighting reports (both in 2015) in upper Cook Inlet, but there were no reports of predation attempts. Transient killer whale vocalizations have been detected on acoustic moorings in upper Cook Inlet (Castellote et al. 2016a) but only once in a 5-year period (Castellote et al. 2016b).

Between 1998 and 2013, 38 necropsies were performed on beluga whale carcasses (23% of the 164 known stranded carcasses) (Burek-Huntington et al. 2015). The sample included adults (n = 25), juveniles (n = 6), calves (n = 3), and aborted fetuses (n = 4). When possible, a primary cause of death was noted along with contributing factors. Cause of death was unknown for 29% of the necropsied carcasses. Other causes of death were attributed to various types of trauma (18%), caused by confirmed and suspected killer whale predation, blunt force, choking on a starry flounder, and entanglement in a setnet (although this individual was in poor health and it could not be determined if it died before or after entanglement); perinatal mortality (13%); mass stranding (13%); single stranding (11%); malnutrition (8%); or disease (8%). Several animals had mild to moderate pneumonia, kidney disease, and/or stomach ulcers that likely contributed to their deaths.

A photo-identification study (Kaplan et al. 2009) did not find any instances where Cook Inlet beluga whales appeared to have been entangled in, or to have otherwise interacted with, fishing gear. However, in 2010, a beluga whale with a rope entangled around its girth was observed and photographed from May through August. Based on how frequently this whale was seen between 2010 and 2013, and the abrupt cessation of sightings post-2013, it is assumed this whale died. It is also possible that it lost the rope and was no longer recognized in subsequent sightings; however, natural marks (i.e., marks other than the rope) were quite distinct on this whale, and it seems likely that it would still have been recognizable if it had been photographed without the rope (McGuire et al. 2018).

STATUS OF STOCK

The Cook Inlet beluga whale stock was designated as depleted under the MMPA in 2000 (65 FR 34590, 21 May 2000) and listed as endangered under the ESA in 2008 (73 FR 62919, 22 October 2008). Therefore, the Cook Inlet beluga whale stock is considered a strategic stock.

There are key uncertainties in the assessment of the Cook Inlet stock of beluga whales. The stock decline is well documented. While the early decline was likely due to unrestricted subsistence harvesting, it is unknown what has prevented recovery of this stock, because subsistence harvest has not been allowed since 2007, and the mortality and serious injury in commercial fisheries is likely low. PBR is designed to allow stocks to recover to, or remain above, the maximum net productivity level (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 Optimum Sustainable Population and that some surplus growth could be removed while still allowing recovery. However, the Cook Inlet beluga whale population is far below historical levels and yet, for unknown reasons, is not increasing. If the Cook Inlet beluga whale population was increasing at an expected rate of approximately 2 to 4%, it would currently be adding, on average, about 7 to 13 whales per year to the population. Although there is currently no known direct human-caused mortality (e.g., from fisheries bycatch, harvest, ship strikes, or other sources), even if the PBR level (~one whale every 2 years) was taken, it is clear this would have little consequence on the overall population trend given the unexplained lack of increase by 7 to 13 whales per year. Stranding data from Cook Inlet have shown that an average of approximately 10 beluga whales died per year between 1998 and 2013 (Burek-Huntington et al. 2015) due to non-human-related or unknown causes, but total mortality in the population is unknown without information on the carcass recovery rate. Individuals die from natural causes even in a growing population; for example, if the average survival rate was a relatively high 0.95, there would still be approximately 14 (0.05×279) deaths expected each year; therefore, it is hard to conclude anything definitive from an average of 10 observed deaths per year.

HABITAT CONCERNS

Critical habitat designated for the Cook Inlet DPS of beluga whales under the ESA includes two geographic areas of marine habitat in Cook Inlet that comprise 7,800 km^2 (3,013 mi^2), excluding waters of the Port of Alaska (76 FR 20180, 11 April 2011). Based on available information, beluga whales remain within the inlet year-round. Review of beluga whale presence data from aerial surveys, satellite tagging, protected species observers, citizen scientists, and opportunistic sightings collected in Cook Inlet from the late 1970s to 2018 shows their range has contracted remarkably since the 1970s (Shelden et al. 2019). Almost the entire population is found in northern Cook Inlet from late spring through the summer and into the fall. This differs markedly from surveys in the 1970s when beluga whales were found in, or would disperse to, lower Cook Inlet by midsummer. Since 2008, on average, 83% of the total population occupied the Susitna Delta (Beluga to Little Susitna rivers) in early June during the aerial survey period, compared to roughly 50% in the past (1978-1979, 1993-1997, 1998-2008). The 2009 to 2014 distribution was estimated to be only 25% of the range observed in 1978 and 1979 (Shelden et al. 2015). Rugh et al. (2000) first noted that whales had not dispersed to the lower inlet in July during surveys in the mid-1990s. This was also evident during aerial surveys conducted in July 2001 (Rugh et al. 2004). Whales transmitting locations from satellite tags during July in 1999 and 2002 also remained in the northern reaches of the upper inlet (Shelden et al. 2015). During surveys in the 1970s, large numbers of whales were scattered throughout the lower inlet in August (Shelden et al. 2015). This was not the case in 2001, when counts in the upper inlet in August were similar to those reported in June and July (Rugh et al. 2004). In August, only 2 of 10 tagged whales spent time in offshore waters and the lower inlet (Shelden et al. 2015). The number of whales observed in the upper inlet during the August calf index surveys, conducted from 2005 to 2012, was similar to the June surveys (Hobbs et al. 2015a), suggesting the contraction in range continued through the summer. While surveys were not conducted in September during the 1970s and 1980s, aerial surveys in 1993 showed some dispersal into lower inlet waters by late September (Shelden et al. 2015). However, surveys in September and October of 2001 resulted in counts that were similar to June (Rugh et al. 2004). With the exception of three whales that spent brief periods of time in the lower inlet during September and/or October, most whales transmitting locations in 1999, 2000, 2001, and 2002 remained in the upper inlet north of East and West Forelands (Shelden et al. 2015, 2018). Counts during aerial surveys in September 2008 were also similar to June (Shelden et al. 2015).

Goetz et al. (2012b) modeled habitat preferences using NMFS' 1994-2008 June abundance survey data. In large areas, such as the Susitna Delta and Knik Arm, there was a high probability that beluga whales were in larger group sizes. Beluga whale presence also increased closer to rivers with Chinook salmon (*Oncorhynchus tshawytscha*) runs, such as the Susitna River. Chinook salmon runs have been decreasing in many Alaska Rivers since 2007, including the Susitna River (https://www.adfg.alaska.gov/index.cfm?adfg=chinookinitiative.main, accessed December 2020). The Susitna Delta also supports two major spawning migrations of a small, schooling eulachon (*Thaleichthys pacificus*) in May and June (Goetz et al. 2012b).

The population appears to be consolidated into habitat in the upper-most reaches of Cook Inlet for much longer periods of time, in habitat that is most likely to be noisy (e.g., Moore et al. 2000, Lowry et al. 2006, Hobbs et al. 2015b, Kendall and Cornick 2015, Norman et al. 2015). An assessment of noise sources in Cook Inlet (Castellote et al. 2019) indicates that anthropogenic noise occurring in some of the most important habitat (i.e., Area 1 critical habitat: 76 FR 20180, 11 April 2011) has the potential to mask beluga communication and hearing, and the potential reduction of communication and echolocation range is considerable. Whether this contracted distribution is a result of changing habitat (Moore et al. 2000), prey concentration, or predator avoidance (Shelden et al. 2003) or can simply be explained as the contraction of a reduced population into small areas of preferred habitat (Goetz et al. 2007, 2012b) is unknown.

The Cook Inlet Beluga Recovery Plan (NMFS 2016b) identifies potential threats: 1) high concern: catastrophic events (e.g., natural disasters, spills, mass strandings), cumulative effects of multiple stressors, and noise; 2) medium concern: disease agents (e.g., pathogens, parasites, and harmful algal blooms), habitat loss or degradation, reduction in prey, and unauthorized take; and 3) low concern: pollution, predation, and subsistence harvest. The recovery plan did not treat climate change as a distinct threat but rather as a consideration in the threats of high and medium concern.

CITATIONS

Boyd, C., R. C. Hobbs, A. E. Punt, K. E. W. Shelden, C. L. Sims, and P. R. Wade. 2019. Bayesian estimation of group sizes for a coastal cetacean using aerial survey data. Mar. Mammal Sci. 35(4):1322-1346. DOI: dx.doi.org/10.1111/mms.12592 .

Burek-Huntington, K. A., J. Dushane, C. E. C. Goertz, L. Measures, C. Romero, and S. Raverty. 2015. Morbidity and mortality in stranded Cook Inlet beluga whales (*Delphinapterus leucas*). Dis. Aquat. Organ. 114(1):45-60. DOI: dx.doi.org/10.3354/dao02839 .

Castellote, M., R. J. Small, J. Mondragon, J. Jenniges, and J. Skinner. 2015. Seasonal distribution and foraging behavior of Cook Inlet belugas based on acoustic monitoring. ADF&G Final Report to Department of Defense.

Castellote, M., R. J. Small, M. O. Lammers, J. J. Jenniges, J. Mondragon, and S. Atkinson. 2016a. Dual instrument passive acoustic monitoring of belugas in Cook Inlet, Alaska. J. Acoust. Soc. Am. 139:2697. DOI: dx.doi.org/10.1121/1.4947427 .

Castellote, M., R. J. Small, J. Mondragon, J. Jenniges, and J. Skinner. 2016b. Seasonal distribution and foraging behavior of Cook Inlet belugas based on acoustic monitoring. Alaska Department of Fish and Game, Final Wildlife Research Report, ADF&G/DWS/WRR-2016-3, Juneau, AK.

Castellote, M., B. Thayre, M. Mahoney, J. Mondragon, M. O. Lammers, and R. J. Small. 2019. Anthropogenic noise and the endangered Cook Inlet beluga whale, *Delphinapterus leucas*: acoustic considerations for management. Mar. Fish. Rev. 80(3):63-88. DOI: dx.doi.org/10.7755/MFR.80.3.3 .

Citta, J. J., L. T. Quakenbush, K. J. Frost, L. Lowry, R. C. Hobbs, and H. Aderman. 2016. Movements of beluga whales (*Delphinapterus leucas*) in Bristol Bay, Alaska. Mar. Mammal Sci. 32:1272-1298. DOI: dx.doi.org/10.1111/mms.12337 .

Citta, J. J., P. Richard, L. F. Lowry, G. O'Corry-Crowe, M. Marcoux, R. Suydam, L. T. Quakenbush, R. C. Hobbs, D. I. Litovka, K. J. Frost, T. Gray, J. Orr, B. Tinker, H. Aderman, and M. L. Druckenmiller. 2017. Satellite telemetry reveals population specific winter ranges of beluga whales in the Bering Sea. Mar. Mammal Sci. 33:236-250. DOI: dx.doi.org/10.1111/mms.12357 .

Dizon, A. E., C. Lockyer, W. F. Perrin, D. P. DeMaster, and J. Sisson. 1992. Rethinking the stock concept: a phylogeographic approach. Conserv. Biol. 6:24-36.

Frost, K. J., and L. F. Lowry. 1990. Distribution, abundance, and movements of beluga whales, *Delphinapterus leucas*, in coastal waters of western Alaska, p. 39-57. *In* T. G. Smith, D. J. St. Aubin, and J. R. Geraci (eds.), Advances in research on the beluga whale, *Delphinapterus leucas*. Can. Bull. Fish. Aquat. Sci. 224.

Goetz, K. T., D. J. Rugh, A. J. Read, and R. C. Hobbs. 2007. Summer habitat preferences of beluga whales (*Delphinapterus leucas*) in Cook Inlet, Alaska. Mar. Ecol. Prog. Ser. 330:247-256.

Goetz, K. T., P. W. Robinson, R. C. Hobbs, K. L. Laidre, L. A. Huckstadt, and K. E. W. Shelden. 2012a. Movement and dive behavior of beluga whales in Cook Inlet, Alaska. AFSC Processed Rep. 2012-03, 40 p. Alaska Fisheries Science Center, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Goetz, K. T., R. A. Montgomery, J. M. Ver Hoef, R. C. Hobbs, and D. S. Johnson. 2012b. Identifying essential summer habitat of the endangered beluga whale *Delphinapterus leucas* in Cook Inlet, Alaska. Endang. Species Res. 16:135-147.

Gurevich, V. S. 1980. Worldwide distribution and migration patterns of the white whale (beluga), *Delphinapterus leucas*. Rep. Int. Whal. Comm. 30:465-480.

Hauser, D. D. W., K. L. Laidre, R. S. Suydam, and P. R. Richard. 2014. Population-specific home ranges and migration timing of Pacific Arctic beluga whales (*Delphinapterus leucas*). Polar Biol. 37(8):1171-1183. DOI: dx.doi.org/10.1007/s00300-014-1510-1 .

Hazard, K. 1988. Beluga whale, *Delphinapterus leucas*, p. 195-235. *In* J. W. Lentfer (ed.), Selected Marine Mammals of Alaska: Species Accounts with Research and Management Recommendations. Marine Mammal Commission, Washington, DC.

Hobbs, R. C. 2013. Detecting changes in population trends for Cook Inlet beluga whales (*Delphinapterus leucas*) using alternative schedules for aerial surveys. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-252, 25 p.

Hobbs, R. C., J. M. Waite, and D. J. Rugh. 2000. Beluga, *Delphinapterus leucas*, group sizes in Cook Inlet, Alaska, based on observer counts and aerial video. Mar. Fish. Rev. 62(3):46-59.

Hobbs, R. C., K. E. W. Shelden, D. J. Rugh, C. L. Sims, and J. M. Waite. 2015a. Estimated abundance and trend in aerial counts of beluga whales, *Delphinapterus leucas*, in Cook Inlet, Alaska, 1994-2012. Mar. Fish. Rev. 77(1):11-31. DOI: dx.doi.org/10.7755/MFR.77.1.2 .

Hobbs, R. C., P. R. Wade, and K. E. W. Shelden. 2015b. Viability of a small, geographically-isolated population of beluga whales, *Delphinapterus leucas*: effects of hunting, predation, and mortality events in Cook Inlet, Alaska. Mar. Fish. Rev. 77(2):59-88. DOI: dx.doi.org/10.7755/MFR.77.2.4 .

Kaplan, C. C., T. L. McGuire, M. K. Blees, and S. W. Raborn. 2009. Longevity and causes of marks seen on Cook Inlet beluga whales, Chapter 1. *In* Photo-identification of beluga whales in Upper Cook Inlet, Alaska: mark analysis, mark-resight estimates, and color analysis from photographs taken in 2008. Report prepared by LGL Alaska Research Associates, Inc., Anchorage, AK, for National Fish and Wildlife Foundation, Chevron, and ConocoPhillips Alaska, Inc. 32 p.

Kendall, L. S., and L. A. Cornick. 2015. Behavior and distribution of Cook Inlet beluga whales, *Delphinapterus leucas*, before and during pile driving activity. Mar. Fish. Rev. 77(2):106-114. DOI: dx.doi.org/10.7755/MFR.77.2.6 .

Laidre, K. L., K. E. W. Shelden, D. J. Rugh, and B. Mahoney. 2000. Beluga, *Delphinapterus leucas*, distribution and survey effort in the Gulf of Alaska. Mar. Fish. Rev. 62(3):27-36.

Lammers, M. O., M. Castellote, R. J. Small, S. Atkinson, J. Jenniges, A. Rosinski, J. N. Oswald, and C. Garner. 2013. Passive acoustic monitoring of Cook Inlet beluga whales (*Delphinapterus leucas*). J. Acoust. Soc. Am. 134:2497-2504. DOI: dx.doi.org/10.1121/1.4816575 .

Lowry, L. F. 1985. The belukha whale (*Delphinapterus leucas*), p. 3-13. *In* J. J. Burns, K. J. Frost, and L. F. Lowry (eds.), Marine mammals species accounts. Alaska Department of Fish and Game, Game Tech. Bull. 7.

Lowry, L., G. O'Corry-Crowe, and D. Goodman. 2006. *Delphinapterus leucas* (Cook Inlet population). *In* IUCN 2006. 2006 IUCN Red List of Threatened Species.

Lowry, L. F., J. J. Citta, G. O'Corry-Crowe, L. T. Quakenbush, K. J. Frost, R. Suydam, R. C. Hobbs, and T. Gray. 2019. Distribution, abundance, harvest, and status of western Alaska beluga whale, *Delphinapterus leucas*, stocks. Mar. Fish. Rev. 81(3-4):54-71.

Lucey, W., H. E. Abraham, G. O'Corry-Crowe, K. M. Stafford, and M. Castellote. 2015. Traditional knowledge and historical and opportunistic sightings of beluga whales, *Delphinapterus leucas*, in Yakutat Bay, Alaska. Mar. Fish. Rev. 77(1):41-46. DOI: dx.doi.org/10.7755/MFR.77.1.4 .

Manly, B. F. J. 2006. Incidental catch and interactions of marine mammals and birds in the Cook Inlet salmon driftnet and setnet fisheries, 1999-2000. Final Report to NMFS Alaska Region. 98 p.

McGuire, T., A. Stephens, and J. McClung. 2018. Photo-identification of beluga whales in upper Cook Inlet, Alaska. Summary of field activities and whales identified in 2017. Report prepared by the Cook Inlet Beluga Whale Photo-ID Project for the National Marine Fisheries Service, Alaska Region. 90 p.

Moore, S. E., K. E. W. Shelden, L. K. Litzky, B. A. Mahoney, and D. J. Rugh. 2000. Beluga, *Delphinapterus leucas*, habitat associations in Cook Inlet, Alaska. Mar. Fish. Rev. 62(3):60-80.

National Marine Fisheries Service (NMFS). 2008. Cook Inlet beluga whale subsistence harvest: Final Supplemental Environmental Impact Statement. U.S. Dep. Commer., NOAA, NMFS, Alaska Region, Office of Protected Resources, Juneau, AK. Available online: https://repository.library.noaa.gov/view/noaa/4948 . Accessed December 2020.

National Marine Fisheries Service (NMFS). 2016a. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2020.

National Marine Fisheries Service (NMFS). 2016b. Recovery plan for the Cook Inlet beluga whale (*Delphinapterus leucas*). National Marine Fisheries Service, Alaska Region, Protected Resources Division, Juneau, AK.

Norman, S. A., R. C. Hobbs, C. E. C. Goertz, K. A. Burek-Huntington, K. E. W. Shelden, W. A. Smith, and L. A. Beckett. 2015. Potential natural and anthropogenic impediments to the conservation and recovery of Cook Inlet beluga whales, *Delphinapterus leucas*. Mar. Fish. Rev. 77(2):89-105. DOI: dx.doi.org/10.7755/MFR.77.2.5 .

O'Corry-Crowe, G., W. Lucey, F. I. Archer, and B. Mahoney. 2015. The genetic ecology and population origins of the beluga whales, *Delphinapterus leucas*, of Yakutat Bay. Mar. Fish. Rev. 77(1):47-58. DOI: dx.doi.org/10.7755/MFR.77.1.5 .

O'Corry-Crowe, G., R. Suydam, L. Quakenbush, B. Potgieter, L. Harwood, D. Litovka, T. Ferrer, J. Citta, V. Burkanov, K. Frost, and B. Mahoney 2018. Migratory culture, population structure and stock identity in North Pacific beluga whales (*Delphinapterus leucas*). PLoS ONE 13(3):e0194201. DOI: dx,doi.org/10.1371/journal.pone.0194201 .

Quakenbush, L. 2003. Summer movements of beluga whales captured in the Kvichak River in May 2002 and 2003. Alaska Beluga Whale Committee Report 03-03. 15 p.

Rugh, D. J., K. E. W. Shelden, and B. Mahoney. 2000. Distribution of beluga whales in Cook Inlet, Alaska, during June/July, 1993 to 1999. Mar. Fish. Rev. 62(3):6-21.

Rugh, D. J., B. A. Mahoney, and B. K. Smith. 2004. Aerial surveys of beluga whales in Cook Inlet, Alaska, between June 2001 and June 2002. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-145, 26 p.

Rugh, D. J., K. E. W. Shelden, C. L. Sims, B. A. Mahoney, B. K. Smith, L. K. Litzky, and R. C. Hobbs. 2005. Aerial surveys of belugas in Cook Inlet, Alaska, June 2001, 2002, 2003, and 2004. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-149, 71 p.

Shelden, K. E. W., D. J. Rugh, B. A. Mahoney, and M. E. Dahlheim. 2003. Killer whale predation on belugas in Cook Inlet, Alaska: implications for a depleted population. Mar. Mammal Sci. 19(3):529-544.

Shelden, K. E. W., D. J. Rugh, K. T. Goetz, C. L. Sims, L. Vate Brattström, J. A. Mocklin, B. A. Mahoney, B. K. Smith, and R. C. Hobbs. 2013. Aerial surveys of beluga whales, *Delphinapterus leucas*, in Cook Inlet, Alaska, June 2005 to 2012. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-263, 122 p.

Shelden, K. E. W., K. T. Goetz, D. J. Rugh, D. G. Calkins, B. A. Mahoney, and R. C. Hobbs. 2015. Spatio-temporal changes in beluga whale, *Delphinapterus leucas*, distribution: results from aerial surveys (1977-2014), opportunistic sightings (1975-2014), and satellite tagging (1999-2003) in Cook Inlet, Alaska. Mar. Fish. Rev. 77(2):1-31 + appendices. DOI: dx.doi.org/10.7755/MFR.77.2.1 .

Shelden, K. E. W., K. T. Goetz, R. C. Hobbs, L. K. Hoberecht, K. L. Laidre, B. A. Mahoney, T. L. McGuire, S. A. Norman, G. O'Corry-Crowe, D. J. Vos, G. M. Ylitalo, S. A. Mizroch, S. Atkinson, K. A. Burek-Huntington, and C. Garner. 2018. Beluga whale, *Delphinapterus leucas*, satellite-tagging and health assessments in Cook Inlet, Alaska, 1999 to 2002. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-369, 227 p.

Shelden, K. E. W., C. Boyd, C. L. Sims, V. A. Gill, and B. A. Mahoney. 2019. Chapter 1: Field report for the June 2018 Cook Inlet beluga aerial abundance and distribution survey. *In* K. E. W. Shelden and P. R. Wade (eds.), Aerial surveys, distribution, abundance, and trend of belugas (*Delphinapterus leucas*) in Cook Inlet, Alaska, June 2018. AFSC Processed Rep. 2019-09, 93 p. Alaska Fisheries Science Center, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Suydam, R. S. 2009. Age, growth, reproduction, and movements of beluga whales (*Delphinapterus leucas*) from the eastern Chukchi Sea. Ph.D. Dissertation, University of Washington, School of Aquatic and Fishery Sciences, Seattle, WA.

Vos, D. J., and K. E. W. Shelden. 2005. Unusual mortality in the depleted Cook Inlet beluga population. Northwest. Nat. 86(2):59-65.

Wade, P. R. 1998. Calculating limits to the allowable human-caused mortality of cetaceans and pinnipeds. Mar. Mammal Sci. 14:1-37. DOI: dx.doi.org/10.1111/j.1748-7692.1998.tb00688.x .

Wade, P. R., C. Boyd, K. E. W. Shelden, and C. L. Sims. 2019. Chapter 2: Group size estimates and revised abundance estimates and trend for the Cook Inlet beluga population. *In* K. E. W. Shelden and P. R. Wade (eds.), Aerial surveys, distribution, abundance, and trend of belugas (*Delphinapterus leucas*) in Cook Inlet, Alaska, June 2018. AFSC Processed Rep. 2019-09, 93 p. Alaska Fisheries Science Center, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Young, N. C., B. J. Delean, V. T. Helker, J. C. Freed, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2014-2018. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-413, 142 p.

NARWHAL (*Monodon monoceros*): Unidentified Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Narwhals are found year-round north of 60°N, primarily in the waters of the Canadian Arctic, Hudson Bay, Baffin Bay, Davis Strait, West Greenland, East Greenland, and the waters around Svalbard, Franz Josef Land, and Novaya Zemyla (Gjertz 1991, Jefferson et al. 2012, Higdon and Ferguson 2014) While large aggregations are found in eastern Arctic waters, they rarely occur in the western Arctic, namely the East Siberian, Bering, Chukchi, and Beaufort seas (COSEWIC 2004) (Fig. 1). The three recognized narwhal populations are based on geographic separation: Baffin Bay, Hudson Bay, and East Greenland (DFO 1998a, 1998b; COSEWIC 2004). The Baffin Bay population summers in the waters along West Greenland and the Canadian High Arctic and overwinters in Baffin Bay and Davis Strait (Koski and Davis 1994, Dietz et al. 2001, Heide-Jørgensen et al. 2003). Narwhals from the northwest Hudson Bay population are thought to overwinter in eastern Hudson Strait (Richard 1991). The East Greenland population is believed to winter in the pack ice between eastern Greenland and Svalbard (Dietz et al. 1994). A poorly described population inhabits the waters around Svalbard, Franz Josef Land, and Novaya Zemyla (Gjertz 1991, Lydersen et al. 2007). The amount of interchange between these populations is unknown. Populations are defined for management purposes, and these designated populations may actually consist of several populations (COSEWIC 2004). Population definition based on molecular genetic studies of narwhals remains unresolved at this time due to extremely low genetic variability within and among management stocks (Palsbøll et al. 1997; de March et al. 2001, 2003).

Figure 1. Potential distribution of narwhals in arctic waters based on extralimital sightings and strandings (George and Suydam, unpubl. ms.; Reeves and Tracey 1980; COSEWIC 2004).

Local observations and traditional ecological knowledge are the primary source for any data on narwhals in Alaska waters, dating back to the 1800s (Bee and Hall 1956; Geist et al. 1960; Noongwook et al. 2007; George and Suydam, unpubl. ms.). The earliest record dates back to 1874, with most occasional sightings occurring around the area east of Point Barrow (Scammon 1874, Ray and Murdoch 1885, Turner 1886, Nelson and True 1887, Murdoch 1898, MacFarlane 1905, Dufresne 1946, Anderson 1947, Bee and Hall 1956, Geist et al. 1960). Narwhal occurrences are reported in Bee and Hall (1956) from Point Barrow to the Colville River Delta. Ljungblad et al. (1983) reported a sighting of two male narwhals northwest of King Island in the Bering Sea, during a systematic scientific survey. Sightings have occurred in Russian waters of the northern Chukchi Sea (Yablokov and Bel'kovich 1968, Reeves and Tracey 1980). George and Suydam (unpubl. ms.) summarized observations from Alaska Native hunters during eight sightings of narwhals in the Chukchi and Beaufort seas between 1989 and 2008. Of these records, seven sightings were live animals totaling 11-12 individuals; one record was of a beachcast narwhal tusk at Cape Sabine. Four of the seven live narwhal sightings consisted of mixed groups of belugas and narwhals (George and Suydam, unpubl. ms.).

Several narwhal specimens collected in Alaska have been documented. Murie (1936) reported a single tusk that was found on a sandbar at Cape Chibukak, St. Lawrence Island. Huey (1952) reported on a specimen collected near Cape Halkett, Harrison Bay, at the mouth of the Colville River, in the Beaufort Sea. Three additional specimen records from various locations were documented in Geist et al. (1960): one specimen was found on the beach of Kiwalik Bay (Kotzebue Sound), another was initially sighted alive at the mouth of the Caribou River in Nelson Lagoon (Alaska Peninsula) but later died, and a third specimen was a tusk found on a beach near Wainwright, on the Chukchi Sea.

It is believed that these incidental narwhal records that occurred in the Beaufort, Chukchi, and Bering seas and Bristol Bay are whales from the Baffin Bay population, which are known to move into the Canadian Arctic Archipelago and as far north and west as ice conditions will permit (COSEWIC 2004). However, there is no evidence or method to confirm this. There are insufficient data to apply the phylogeographic approach to stock structure (Dizon et al. 1992) for narwhals.

POPULATION SIZE

Reliable estimates of abundance for narwhals in Alaska are currently unavailable.

Minimum Population Estimate

At this time, it is not possible to produce a reliable minimum population estimate (N_{MIN}) for this stock, as current estimates of abundance are unavailable.

Current Population Trend

At present, reliable data on trends in population abundance are unavailable.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate is currently unavailable for narwhals in Alaska. Hence, until additional data become available, it is recommended that the cetacean maximum theoretical net productivity rate (R_{MAX}) of 4% be employed (Wade and Angliss 1997).

POTENTIAL BIOLOGICAL REMOVAL

Under the 1994 reauthorized Marine Mammal Protection Act (MMPA), the potential biological removal (PBR) is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: $PBR = N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for these stocks is 0.5, the value for cetacean stocks with unknown population status (Wade and Angliss 1997). However, in the absence of a reliable estimate of a minimum abundance, the PBR for this stock is unknown.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Fisheries Information

There are no U.S. commercial fisheries operating within the normal range of narwhals in Alaska. There are no observer program records of narwhal mortality or serious injury incidental to commercial fisheries in Alaska. The estimated mean annual mortality and serious injury rate incidental to U.S. commercial fisheries is zero.

Subsistence/Native Harvest Information

There is no known subsistence harvest of narwhals by Alaska Natives.

STATUS OF STOCK

Narwhals are not designated as depleted under the MMPA or listed as threatened or endangered under the Endangered Species Act. Reliable estimates of the minimum population, population trend, PBR, and status of the stock relative to its Optimum Sustainable Population are currently not available. There are no federal or state commercial fisheries operating in the marine waters of the Arctic, and there are no reports of mortality or serious injury of narwhals in Alaska, therefore, the mean annual mortality and serious injury rate is considered insignificant and approaching zero. The estimated annual rate of human-caused mortality and serious injury is believed to be zero for this stock. Thus, the Unidentified stock of narwhals in Alaska is not classified as strategic.

HABITAT CONCERNS

Narwhals tend to prefer heavy ice cover in the winter and animals studied in Baffin Bay chose areas associated with high concentrations of Greenland halibut, which correspond to the coldest bottom temperatures (Laidre et al. 2004b; Laidre and Heide-Jørgensen 2005b, 2011). Narwhals wintering in Hudson Strait are also found in ice-covered areas of deep water, but the maximum depths are much shallower than the areas used by narwhals in Baffin Bay (Laidre et al. 2003, 2004a). As the Arctic warms through climate change, ice cover will be thinner, form later, melt earlier, and be less predictable. A warming Arctic will also see changes in ocean currents which create conditions that support concentrations of winter narwhal prey species, such as Greenland halibut. This may result in a shift in distribution of narwhals and their prey, requiring changes in migration timing, as well as destinations

(Kovaks and Lydersen 2008; Laidre et al. 2008, 2010, 2015). An increased risk of ice entrapment is associated with the changes in sea-ice formation, because seasonal cues for the timing of freeze up have changed and because later freezing may result in large expanses of open water freezing at one time (Heide-Jørgensen et al. 2002, Heide-Jørgensen and Laidre 2004, Laidre and Heide-Jørgensen 2005a, Laidre et al. 2012).

In addition to changing sea ice, narwhals are threatened by a number of changes associated with warming of the Arctic, including increased shipping and development, which adds noise; risk of pollution and ship strikes; risk of predation by killer whales (*Orcinus orca*) (Laidre et al. 2006); shifts in prey abundance and distribution; and exposure to novel diseases (Laidre et al. 2015).

CITATIONS

Anderson, R. M. 1947. Catalogue of Canadian recent mammals. Bull. Natl. Mus. Canada, Biol. Ser. No. 31: vi + 238 p.

Bee, J. W., and E. R. Hall. 1956. Mammals of northern Alaska on the Arctic slope. Univ. Kansas Mus. Nat. Hist., Misc. Publ. No. 8. 309 p.

Committee on the Status of Endangered Wildlife in Canada (COSEWIC). 2004. COSEWIC assessment and update status report on the narwhal, *Monodon monoceros*, in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa. vii + 50 p. Available online: http://www.sararegistry.gc.ca/virtual_sara/files/cosewic/sr_narwhal_e.pdf. Accessed December 2016.

de March, B. G. E., L. D. Maiers, and D. Tenkula. 2001. A preliminary analysis of the molecular genetics of narwhal (*Monodon monoceros*) samples collected from Canadian and adjacent waters from 1982 to 2000. Canada/Greenland Joint Commission on the Management and Conservation of Narwhal and Beluga (JCNB), Scientific Working Group, Quqetarsuaq, Greenland, May 9-13, 2001. Document No. SWG-2001-10.

de March, B. G. E., D. A. Tenkula, and L. D. Postma. 2003. Molecular genetics of narwhal (*Monodon monoceros*) from Canada and West Greenland (1982-2001). Department of Fisheries and Oceans Canada, Canadian Science Advisory Secretariat Research Document 2003/080. 23 p.

Department of Fisheries and Oceans Canada (DFO). 1998a. Hudson Bay narwhal. Department of Fisheries and Oceans Canada, Central and Arctic Region, DFO Sci. Stock Status Rep. E5-44. 5 p.

Department of Fisheries and Oceans Canada (DFO). 1998b. Baffin Bay narwhal. Department of Fisheries and Oceans Canada, Central and Arctic Region, DFO Sci. Stock Status Rep. E5-43. 5 p.

Dietz, R., M. P. Heide-Jørgensen, E. Born, and C. M. Glahder. 1994. Occurrence of narwhals (*Monodon monoceros*) and white whales (*Delphinapterus leucas*) in East Greenland. Medd. Grønl. Biosci. 39:69-86.

Dietz, R., M. P. Heide-Jørgensen, P. R. Richard, and M. Acquarone. 2001. Summer and fall movements of narwhals (*Monodon monoceros*) from northeastern Baffin Island towards northern Davis Strait. Arctic 54:244-261.

Dizon, A. E., C. Lockyer, W. F. Perrin, D. P. DeMaster, and J. Sisson. 1992. Rethinking the stock concept: a phylogeographic approach. Conserv. Biol. 6:24-36.

Dufresne, F. 1946. Alaska's Animals and Fishes. A. S. Barnes, New York. xviii + 297 p.

Geist, O. W., J. L. Buckley, and R. H. Manville. 1960. Alaskan records of the narwhal. J. Mammal. 41(2):250-253.

George, J. C., and R. Suydam Unpubl. manuscript. Recent observations of narwhal in the Chukchi and Beaufort Seas by local hunters, 13 January 2009. 3 p. Available from Marine Mammal Laboratory, Alaska Fisheries Science Center, 7600 Sand Point Way NE, Seattle, WA 98115.

Gjertz, I. 1991. The narwhal, *Monodon monoceros*, in the Norwegian High Arctic. Mar. Mammal Sci. 7:402-408.

Heide-Jørgensen, M. P., and K. L. Laidre. 2004. Declining open water refugia for top predators in Baffin Bay and adjacent waters. Ambio 33(8):488-495.

Heide-Jørgensen, M. P., P. Richard, M. Ramsay, and S. Akeeagok. 2002. Three recent ice entrapments of arctic cetaceans in West Greenland and the eastern Canadian High Arctic. NAMMCO Scientific Publications 4:143-148. DOI: 10.7557/3.2841.

Heide-Jørgensen, M. P., R. Dietz, K. L. Laidre, P. R. Richard, J. Orr, and H. C. Schmidt. 2003. The migratory behaviour of narwhals (*Monodon monoceros*). Can. J. Zool. 81:1298-1305.

Higdon, J. W., and S. H. Ferguson. 2014. History of narwhal aerial surveys and abundance estimates in the Canadian Arctic. Working Paper NAMMCO/SC/21-JCNB/SWG/2014-JWG/09 presented at NAMMCO/JCNB Joint Working Group on narwhals and belugas, 10-12 March 2014, Copenhagen, Denmark.

Huey, L. M. 1952. An Alaskan record of the narwhal. J. Mammal. 33:496.

Jefferson, T. A., L. Karkzmarski, K. Laidre, G. O'Corry-Crowe, R. Reeves, L. Rojas-Bracho, E. Secchi, E. Slooten, B. D. Smith, J. Y. Wang, and K. Zhou. 2012. *Monodon monoceros*. The IUCN Red List of Threatened Species. Version 2014.2.

Koski, W. R., and R. A. Davis. 1994. Distribution and numbers of narwhals (*Monodon monoceros*) in Baffin Bay and Davis Strait. Medd. Grønl. Biosci. 39:15-40.

Kovacs, K. M., and C. Lydersen. 2008. Climate change impacts on seals and whales in the North Atlantic Arctic and adjacent shelf seas. Sci. Prog. 91(Pt. 2):117-150.

Laidre, K. L., and M. P. Heide-Jørgensen. 2005a. Arctic sea ice trends and narwhal vulnerability. Biol. Conserv. 121:509-517.

Laidre, K. L., and M. P. Heide-Jørgensen. 2005b. Winter feeding intensity of narwhals (*Monodon monoceros*). Mar. Mammal Sci. 21:45-57.

Laidre, K. L., and M. P. Heide-Jørgensen. 2011. Life in the lead: extreme densities of narwhals in the offshore pack ice. Mar. Ecol. Prog. Ser. 423:269-278.

Laidre, K. L., M. P. Heide-Jørgensen, R. Dietz, R. C. Hobbs, and O. A. Jørgensen. 2003. Deep-diving by narwhals, *Monodon monoceros*: differences in foraging behavior between wintering areas? Mar. Ecol. Prog. Ser. 261:269-281.

Laidre, K. L., M. P. Heide-Jørgensen, O. A. Jørgensen, and M. A. Treble. 2004a. Deep ocean predation by a high Arctic cetacean. ICES J. Mar. Sci. 61(3):430-440.

Laidre, K. L., M. P. Heide-Jørgensen, M. L. Logsdon, R. C. Hobbs, P. Heagerty, R. Dietz, O. A. Jørgensen, and M. A. Treble. 2004b. Seasonal habitat associations of narwhals in the high Arctic. Mar. Biol. 145:821-831.

Laidre, K. L., M. P. Heide-Jørgensen, and J. Orr. 2006. Reactions of narwhals, *Monodon monoceros*, to killer whale, *Orcinus orca*, attacks in the eastern Canadian Arctic. Can. Field-Nat. 120:457-465.

Laidre, K. L., I. Stirling, L. Lowry, Ø. Wiig, M. P. Heide-Jørgensen, and S. Ferguson. 2008. Quantifying the sensitivity of arctic marine mammals to climate-induced habitat change. Ecol. Appl. 18(2):S97-S125.

Laidre, K. L., M. P. Heide-Jørgensen, W. Ermold, and M. Steele. 2010. Narwhals document continued warming of southern Baffin Bay. J. Geophys. Res. 115:C10049.

Laidre K. L., M. P. Heide-Jørgensen, H. Stern, and P. Richard. 2012. Unusual sea ice entrapments and delayed autumn ice-up timing reinforce narwhal vulnerability to climate change. Polar Biol. 35(1):149-154.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conserv. Biol. 29(3):724-737.

Ljungblad, D. K., S. E. Moore, and D. R. Van Schoik. 1983. Aerial surveys of endangered whales in the Beaufort, eastern Chukchi and northern Bering seas, 1982. NOSC Technical Document 605. 110 p + appendix.

Lydersen, C., A. R. Martin, I. Gjertz, and K. M. Kovacs. 2007. Satellite tracking and diving behaviour of sub-adult narwhals (*Monodon monoceros*) in Svalbard, Norway. Polar Biol. 30:437-442.

MacFarlane, R. 1905. Notes on mammals collected and observed in the northern Mackenzie River District, Northwest Territories of Canada. Proc. U.S. Nat. Mus. 28:673-764.

Murdoch, J. 1898. The animals known to the Eskimos of Northwestern Alaska. Amer. Nat. 32:719-734.

Murie, O. J. 1936. Notes on the mammals of St. Lawrence Island, Alaska, p. 337-326. *In* Archaeological excavations at Kukulik, St. Lawrence Island, Alaska. Univ. Alaska, Misc. Publ. 2.

Nelson, E. W., and F. W. True. 1887. Mammals of northern Alaska. Pt. 2, p. 227-293. *In* Report upon natural history collections made in Alaska between the years 1877 and 1881 by Edward W. Nelson. Arctic Publ. No. 3, Signal Service, U.S. Army

Noongwook, G., The Native Village of Savoonga, The Native Village of Gambell, H. P. Huntington, and J. C. George. 2007. Traditional knowledge of the bowhead whale (*Balaena mysticetus*) around St. Lawrence Island, Alaska. Arctic 60 (1):47-54.

Palsbøll, P. J., M. P. Heide-Jørgensen, and R. Dietz. 1997. Genetic studies of narwhals, *Monodon monoceros*, from West and East Greenland. Heredity 78:284-292.

Ray, P. H., and J. Murdoch. 1885. Report of the International Polar Expedition to Point Barrow, Alaska. Government Printing Office, Washington. 695 p.

Reeves, R. R., and S. Tracey. 1980. *Monodon monoceros*. Mamm. Species 127:1-7.

Richard, P. 1991. Abundance and distribution of narwhals (*Monodon monoceros*) in northern Hudson Bay. Can. J. Fish. Aquat. Sci. 48:276-283.

Scammon, C. M. 1874. The Marine Mammals of the Northwestern Coast of North America, Described and Illustrated: Together with an Account of the American Whale Fishery. G. P. Putnam's Sons, New York. 319 p.

Turner, L. M. 1886. Contributions to the natural history of Alaska; results of investigations made chiefly in the Yukon District and the Aleutian Islands. Arctic Publ. No. 2, Signal Service, U.S. Army. 226 p.

Wade, P. R., and R. Angliss. 1997. Guidelines for assessing marine mammal stocks: report of the GAMMS Workshop April 3-5, 1996, Seattle, Washington. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-OPR-12, 93 p.

Yablokov A. V., and V. M. Bel'kovich. 1968. Cetaceans of the Arctic; their proper utilization and conservation. Probl. of the North, Nat. Res. Council, Ottawa 11:199-218.

Revised 12/30/2016

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

NOTE – NMFS has preliminary genetic information on killer whales in Alaska which indicates that the current stock structure of killer whales in Alaska needs to be reassessed. NMFS is evaluating the new genetic information. In the interim, new information on killer whale mortality levels is provided within this report. A complete revision of the killer whale stock assessments will be postponed until the stock structure evaluation is completed and any new stocks are identified.

STOCK DEFINITION AND GEOGRAPHIC RANGE

Killer whales have been observed in all oceans and seas of the world (Leatherwood and Dahlheim 1978). Although reported from tropical and offshore waters, killer whales occur at higher densities in colder and more productive waters of both hemispheres, with the greatest densities found at high latitudes (Mitchell 1975, Leatherwood and Dahlheim 1978, Forney and Wade, 2006). Killer whales are found throughout the North Pacific. Along the west coast of North America, killer whales occur along the entire Alaska coast (Braham and Dahlheim 1982), in British Columbia and Washington inland waterways (Bigg et al. 1990), and along the outer coasts of Washington, Oregon, and California (Green et al. 1992; Barlow 1995, 1997; Forney et al. 1995). Seasonal and year-round occurrence has been noted for killer whales throughout Alaska (Braham and Dahlheim 1982) and in the intracoastal waterways of British Columbia and Washington State, where whales have been labeled as "resident," "transient," and "offshore" type killer whales (Bigg et al. 1990, Ford et al. 2000, Dahlheim et al. 2008) 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, 2002; Barrett-Lennard 2000; Dahlheim et al. 2008). Through examination of photographs of recognizable individuals and pods, movements of whales between geographical areas have been documented. For example, whales identified in Prince William Sound have been observed near Kodiak Island (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). Movements of killer whales between the waters of Southeast Alaska and central California have also been documented (Goley and Straley 1994, Black et al. 1997, Dahlheim and White 2010).

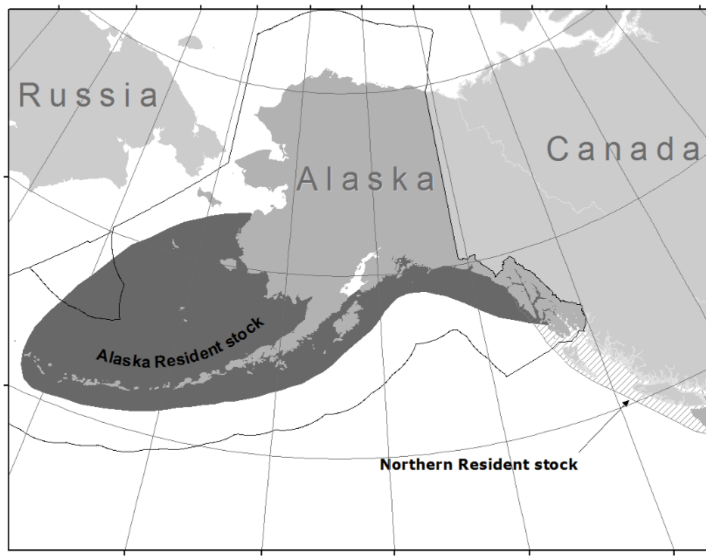

Figure 1. Approximate distribution of resident killer whales in the eastern North Pacific (shaded areas). The distribution of resident and transient killer whale stocks in the eastern North Pacific largely overlap (see text).

Several studies provide evidence that the resident, offshore, and transient ecotypes are genetically distinct in both mtDNA and nuclear DNA (Hoelzel and Dover 1991; Hoelzel et al. 1998, 2002; Barrett-Lennard 2000). A recent global genetic study of killer whales using the entire mitochondrial genome found that some killer whale ecotypes represent deeply divergent evolutionary lineages and warrant elevation to species or subspecies status (Morin et al. 2010). In particular, estimates from mitogenome sequence data indicate that transient killer whales diverged from all other killer whale lineages ~700,000 years ago. In light of these differences, the Society for Marine Mammalogy's Committee on Taxonomy currently recognizes the resident and transient North Pacific ecotypes as un-named *Orcinus orca* subspecies (Committee on Taxonomy 2012). In recognition of its status as an un-named subspecies or species, some researchers now refer to transient-type killer whales as Bigg's killer whales (e.g., Ford 2011, Riesch et al. 2012), in tribute to the late Dr. Michael Bigg.

Genetic differences have also been found between populations within the transient and resident ecotypes (Hoelzel et al. 1998, 2002; Barrett-Lennard 2000). Within the resident ecotype, association data were used to describe three separate populations in the North Pacific: Southern Residents, Northern Residents, and Alaska Residents (Bigg et al. 1990; Ford et al. 1994, 2000; Dahlheim et al. 1997; Matkin et al. 1999). In previous stock assessment reports, the Alaska and Northern Resident populations were considered one stock. Acoustic data (Ford 1989, 1991; Yurk et al. 2002) and genetic data (Hoelzel et al. 1998, 2002; Barrett-Lennard 2000) have now confirmed that these three units represent discrete populations. The Southern Resident population is found in summer primarily in waters of Washington state and southern British Columbia and has never been seen to associate with other resident stocks. The Northern Resident population is found in summer primarily in central and northern British Columbia. Members of the Northern Resident population have been documented in southeastern Alaska; however, they have not been seen to intermix with Alaska Residents (Fig. 1). Alaska Resident whales are found from southeastern Alaska to the Aleutian Islands and Bering Sea. Intermixing of Alaska Residents have been documented among the three areas, at least as far west as the eastern Aleutian Islands.

Based on data regarding association patterns, acoustics, movements, and genetic differences, eight killer whale stocks are now recognized within the Pacific U.S. EEZ: 1) the Alaska Resident stock - occurring from southeastern Alaska to the Aleutian Islands and Bering Sea, 2) the Northern Resident stock - occurring from Washington State through part of southeastern Alaska, 3) the Southern Resident stock - occurring mainly within the inland waters of Washington State and southern British Columbia, but also in coastal waters from southeastern Alaska through California, 4) the Gulf of Alaska, Aleutian Islands, and Bering Sea Transient stock - occurring mainly from Prince William Sound through the Aleutian Islands and Bering Sea, 5) the AT1 Transient stock - occurring in Alaska from Prince William Sound through the Kenai Fjords, 6) the West Coast transient stock - occurring from California through southeastern Alaska, 7) the Offshore stock - occurring from California through Alaska, and 8) the Hawaiian stock. Transient whales in Canadian waters are considered part of the West Coast Transient stock. The Stock Assessment Reports for the Alaska Region contain information concerning all the killer whale stocks except the Hawaiian and Offshore stocks.

Resident killer whales ranging from Southeastern Alaska to Kodiak Island have been observed in regular association during multipod encounters since 1984 (Matkin et al. 2010). Tagging data also indicates the range of killer whales seen in these aggregations extends from Southeastern Alaska to south of Kodiak Island (Matkin et al. 2010). Although recent studies have documented movements of Alaska Resident killer whales from the Bering Sea into the Gulf of Alaska as far north as southern Kodiak Island, none of these whales have been photographed further north and east in the Gulf of Alaska where regular photoidentification studies have been conducted since 1984 (P. Wade, pers. comm., MML-AFSC, Seattle, WA, 10 December 2012; unpublished data; Matkin et al. 2010). The resident-type killer whales encountered in western Alaska possibly belong to groups that are distinct from the groups of resident killer whales in the Gulf of Alaska because no call syllables or call patterns (sequence of syllables) between groups were found to match (Matkin et al. 2007).

POPULATION SIZE

The Alaska Resident stock includes killer whales from southeastern Alaska to the Aleutian Islands and Bering Sea. Preliminary analysis of photographic data resulted in the following minimum counts for resident killer whales belonging to the Alaska Resident stock (Note: individual whales have been matched between geographical regions and missing animals likely to be dead have been subtracted). In southeastern Alaska, 109 resident whales have been identified as of 2009 (MML and North Gulf Oceanic Society (NGOS), 3430 Main Street, Suite B1, Homer, Alaska; unpublished data). In Prince William Sound and Kenai Fjords, another 675 resident whales have been identified as of 2009 (Matkin et al. 2003; C. Matkin, North Gulf Oceanic Society, pers. comm.).

Beginning in 2001, dedicated killer whale studies were initiated by the NMFS Marine Mammal Laboratory (MML) in Alaska waters west of Kodiak Island, including the Aleutian Islands and Bering Sea. Between 2001 and 2009, using field assessments based on morphology, association data, and genetic analyses, additional resident whales were added to the Alaska Resident stock. Internal matches within the MML data set have been subtracted, resulting in a final count of western Alaska residents for 2001-2012 as 1,475 whales. Studies conducted in western Alaska by the NGOS have resulted in the collection of photographs of approximately 600 resident killer whales; however, the NGOS and MML data sets have not yet been matched so it is unknown how many of these 600 animals are included in the MML collection. Another 41 whales were identified off Kodiak between 2000 and 2003 by the NGOS. These whales are added to the total of western Alaska residents although they have not been matched to MML photographs.

MML conducted killer whale line-transect surveys for 3 years in July and August in 2001-2003. These surveys covered an area from approximately Resurrection Bay in the Kenai Fjords to the central Aleutians. The surveys covered an area from shore to 30-45 nautical miles offshore, with randomly located transects in a zigzag pattern. A total of 9,053 km of tracklines were surveyed between the Kenai Peninsula (~150°W) and Amchitka Pass

(~179ºW). A total of 41 on-effort sightings of killer whales were recorded, with an additional 16 sightings off-effort. Estimated abundance of resident killer whale from these surveys was 991 (CV = 0.52), with a 95% confidence interval of 380-2,585 (Zerbini et al. 2007).

The line transect surveys provide an "instantaneous" (across ~40 days) estimate of the number of resident killer whales in the survey area. It should be noted that the photographic catalogue encompasses a larger area, including some data from areas such as Prince William Sound and the Bering Sea that were outside the line-transect survey area. Additionally, the number of whales in the photographic catalogue is a documentation of all whales seen in the area over the time period of the catalogue; movements of some individual whales have been documented between the line-transect survey area and locations outside the survey area. Accordingly, a larger number of resident killer whales may use the line-transect survey area at some point over the 3 years than would necessarily be found at one time in the survey area in July and August in a particular year.

Combining the counts of known resident whales gives a minimum number of 2,347 (Southeast Alaska + Prince William Sound + Western Alaska; 121 + 751 + 1,475) killer whales belonging to the Alaska Resident stock (Table 1).

Table 1. Numbers of animals in each pod of killer whales belonging to the Alaska Resident stock of killer whales. A number followed by a "+" indicates a minimum count for that pod.

Pod ID	**1999/2000 estimate (and source)**	**2001/2004 estimate (and source)**	**2005-2012 estimate (and source)**
Southeast Alaska			33 (Matkin et al. in prep.)
AF22			
AF5	49 (Dahlheim et al. 1997, Matkin et al. 1999)	61 (C. Matkin, NGOS, pers. comm.)	46 (Matkin et al. in prep.)
AG	27 (Dahlheim et al. 1997, Matkin et al. 1999)	33 (C. Matkin, NGOS, pers. comm.)	42 (Matkin et al. in prep.)
AZ	23+ (Dahlheim, AFSC-MML, pers. comm.)	23+ (Dahlheim et al. 1997)	Not seen since prior to 1997
Total, Southeast Alaska	**99+**	**117+**	**121 (excluding AZ)**
Prince William Sound	**Matkin et al. 1999**	**Matkin et al. 2003 and C. Matkin, NGOS, pers. comm.**	**Matkin et al. in prep.**
AA1	---	8	8
AA30	---	---	24
AB	25	19	20
AB25	---	10	19
AD05	---	16	22
AD16	7	4	9
AE	16	19	17
AH01		9	9
AH20		12	12
AI	7	7	8
AJ	38	42	57
AK	12	13	19
AL	---	---	23
AN10	20	27	36
AN20	assume 9	33	30
AS2	assume 20	21	31
AS30		14	19
AW		24	27
AX01	21	20	33
AX27		24	26
AX32		15	18
AX40		14	16
AX48		20	23
AY	assume 11	18	21
Unassigned to pods	138 (C. Matkin, NGOS, pers. comm.)	112	220

Pod ID	1999/2000 estimate (and source)	2001/2004 estimate (and source)	2005-2012 estimate (and source)
Total, Prince William Sound/ Kenai Fjord/ Kodiak	**341**	**501**	**751**
Western Alaska	**Dahlheim et al. 1997 and MML unpublished data[2]**	**2001/2003 MML unpublished data[2]**	**2001-2012 MML/NGOS unpublished catalog[2]**
Unassigned to pods (MML)	68+	464	1,475 (H. Fearnbach, NOAA-SWFSC, pers. comm., April 2013)
Total, Western Alaska	**68+**	**505**	**1,475**
Total, all areas	**507**	**1,123**	**2,347[1]**

[1]Although there is strong evidence (Matkin et al. 2003, 2010) the resident killer whale numbers have been increasing in the Gulf of Alaska, the bulk of the increase from the 2001-2004 counts to the 2005-2009 counts is believed to be due to the discovery of new animals, not recruitment. Animals reported here have been photographed in the 2001-2012 period. [2]Available from M. Dahlheim, Marine Mammal Laboratory, Alaska Fisheries Science Center, 7600 Sand Point Way NE, Seattle, WA 98115.

Minimum Population Estimate

The survey technique utilized for obtaining the abundance estimate of killer whales is a direct count of individually identifiable animals. Thus the minimum population estimate (N_{MIN}) for the Alaska Resident stock of killer whales based on photo-identification studies conducted between 2005-2009 is 2,084 animals (Table 1). Other estimates of the overall population size (i.e., N_{BEST}) and associated CV(N) are not currently available. Given that researchers continue to identify new whales, the estimate of abundance based on the number of uniquely identified individuals known to be alive is likely conservative. However, the rate of discovering new resident whales within southeastern Alaska and Prince William Sound is relatively low (MML, unpublished data). Conversely, the rate of discovery of new whales in western Alaska was initially high (i.e., 2001 and 2002 field seasons). However, recent photographic data collected during 2003 and 2004 indicates that the rate of discovering new individual whales has decreased.

Using the line-transect estimate of 991 (CV = 0.52) results in an estimate of N_{MIN} (20th percentile) of 656. This is lower than the minimum number of individuals identified from photographs in recent years, so the photographic catalogue number is used for PBR calculations.

Some overlap of Northern Resident whales occur with the Alaska Resident stock in southeastern Alaska. However, information on the percentage of time that the Northern Resident stock spends in Alaska waters is unknown. However, as noted above, this minimum population estimate is considered conservative. This approach is consistent with the recommendations of the Alaska Scientific Review Group (DeMaster 1996).

Current Population Trend

Data from Matkin et al. (2003) indicate that the component of the Alaska Resident stock that summers in the Prince William Sound and Kenai Fjords area is increasing. With the exception of AB pod, which declined drastically after the *Exxon Valdez* oil spill and has not yet recovered, the component of the Alaska Resident stock in the Prince William Sound and Kenai Fjords area increased 3.2% (95% CI = 1.94 to 4.36%) per year from 1990 to 2005 (Matkin et al. 2008). Although the current minimum population count of 2,084 is higher than the last population count of 1,123, examination of only count data does not provide a direct indication of the net recruitment into the population. At present, reliable data on trends in population abundance for the entire Alaska Resident stock of killer whales are unavailable.

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 the Pacific Northwest 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), and 3.3% over the period 1984-2002 (Matkin et al. 2003). Until additional stock-specific data become available, it is recommended that the cetacean maximum theoretical net productivity rate (R_{MAX}) of 4% be employed for this stock (Wade and Angliss 1997).

POTENTIAL BIOLOGICAL REMOVAL

Under the 1994 reauthorized Marine Mammal Protection Act (MMPA), the potential biological removal (PBR) is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: PBR = $N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 0.5, the value for cetacean stocks with unknown population status (Wade and Angliss 1997). Thus, for the Eastern North Pacific Alaska Resident killer whale stock, PBR = 24 animals (2,347 × 0.02 × 0.5).

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Fisheries Information

Detailed information on U.S. commercial fisheries in Alaska waters (including observer programs, observer coverage, and observed incidental takes of marine mammals) is presented in Appendices 3-6 of the Alaska Stock Assessment Reports.

Three of the federally-regulated U.S. commercial fisheries, monitored for incidental mortality and serious injury of marine mammals by fishery observers, incurred mortality and serious injury of killer whales (unknown stock) between 2010 and 2014: the Bering Sea/Aleutian Islands flatfish trawl, Bering Sea/Aleutian Islands rockfish trawl, and Bering Sea/Aleutian Islands Pacific cod longline fisheries (Table 1; Breiwick 2013; MML, unpubl. data).

Fishery observers have collected tissue samples from many of the killer whales that were killed incidental to U.S. commercial fisheries. Genetic analyses of samples from seven killer whales collected between 1999 and 2004 have confirmed that Alaska Resident killer whale mortality occurred incidental to the Bering Sea/Aleutian Islands flatfish trawl (n = 3) and Bering Sea/Aleutian Islands Pacific cod longline fisheries (n = 1) and that Gulf of Alaska, Aleutian Islands, and Bering Sea Transient killer whale mortality occurred incidental to the Bering Sea/Aleutian Islands pollock trawl fishery (n = 3) (M. Dahlheim, NMFS-AFSC-MML, pers. comm., 20 February 2013). Given the overlap in the range of transient and resident stocks in Alaska waters, unless genetic samples can be collected from animals injured or killed by gear or the ship's propeller, these events are assigned to both the transient and resident stock occurring in that area. Thus, the estimated mean annual mortality and serious injury rate of one killer whale in 2010-2014 will be assigned to both the Alaska Resident and Gulf of Alaska, Aleutian Islands, and Bering Sea Transient stocks of killer whales (Table 2; Breiwick 2013; MML, unpubl. data).

Typically, if mortality or serious injury occurs incidental to U.S. commercial fishing, it is due to interactions with the fishing gear. However, reports indicate that observed killer whale mortality incidental to the Bering Sea/Aleutian Islands trawl fisheries often occurs due to contact with the ship's propeller (e.g., the 2010 mortality in the Bering Sea/Aleutian Islands rockfish trawl fishery).

Table 2. Summary of incidental mortality and serious injury of Alaska Resident killer whales due to U.S. commercial fisheries in 2010-2014 and calculation of the mean annual mortality and serious injury rate (Breiwick 2013; MML, unpubl. data). Methods for calculating percent observer coverage are described in Appendix 6 of the Alaska Stock Assessment Reports. N/A indicates that data are not available.

Fishery name	Years	Data type	Percent observer coverage	Observed mortality	Estimated mortality	Mean estimated annual mortality
Bering Sea/Aleutian Is. flatfish trawl	2010 2011 2012 2013 2014	obs data	99 100 99 99 99	0 0 0 (+1)[a] 2 0	0 0 0 (+1)[b] 2 0	0.4 (+0.2)[c] (CV = 0)
Bering Sea/Aleutian Is. rockfish trawl	2010 2011 2012 2013 2014	obs data	99 99 100 99 99	1 0 0 0 0	1 0 0 0 0	0.2 (CV = 0)

Fishery name	**Years**	**Data type**	**Percent observer coverage**	**Observed mortality**	**Estimated mortality**	**Mean estimated annual mortality**
Bering Sea/Aleutian Is. Pacific cod longline	2010 2011 2012 2013 2014	obs data	64 57 51 66 64	0 0 0 (+1)[d] 0 0	0 0 0 (+1)[e] 0 0	0 (+0.2)[f] (CV = N/A)
Minimum total estimated annual mortality						1 (CV = 0)

[a]Total mortality and serious injury observed in 2012: 0 whales in sampled hauls + 1 whale in an unsampled haul.
[b]Total estimate of mortality and serious injury in 2012: 0 whales (extrapolated estimate from 0 whales observed in sampled hauls) + 1 whale (1 whale observed in an unsampled haul).
[c]Mean annual mortality and serious injury for fishery: 0.4 whales (mean of extrapolated estimates from sampled hauls) + 0.2 whales (mean of number observed in unsampled hauls).
[d]Total mortality and serious injury observed in 2012: 0 whales in sampled hauls + 1 whale in an unsampled haul.
[e]Total estimate of mortality and serious injury in 2012: 0 whales (extrapolated estimate from 0 whales observed in sampled hauls) + 1 whale (1 whale observed in an unsampled haul).
[f]Mean annual mortality and serious injury for fishery: 0 whales (mean of extrapolated estimates from sampled hauls) + 0.2 whales (mean of number observed in unsampled hauls).

A minimum estimate of the mean annual mortality and serious injury rate incidental to U.S. commercial fisheries in 2010-2014 is one Alaska Resident killer whale, based on observer data (Table 2).

Subsistence/Native Harvest Information

There are no reports of a subsistence harvest of killer whales in Alaska.

Other Mortality

During the 1992 killer whale surveys conducted in the Bering Sea and western Gulf of Alaska, 9 of 182 (4.9%) individual whales in 7 of the 12 (58%) pods encountered had evidence of bullet wounds (Dahlheim and Waite 1993). The relationship between wounding due to shooting and survival is unknown. In Prince William Sound, the pod responsible for most of the fishery interactions experienced a high level of mortality: between 1986 and 1991, 22 whales out of a pod of 37 (59%) disappeared (Matkin et al. 1994). The cause of death for these whales is unknown, but it may be related to gunshot wounds or effects of the *Exxon Valdez* oil spill (Dahlheim and Matkin 1994). It is unknown who was responsible for shooting at killer whales.

There have been no obvious bullet wounds observed on killer whales during surveys in the Bering Sea and western Gulf of Alaska (J. Durban, NMFS-SWFSC, pers. comm.). However, researchers have reported that killer whale pods in certain areas exhibit vessel avoidance behavior, which may indicate that shootings occur in some places.

Other Issues

Killer whales are known to depredate longline catches in the Bering Sea (Dahlheim 1988; Yano and Dahlheim 1995; Perez 2003, 2006; Sigler et al. 2003) and in the Gulf of Alaska (Sigler et al. 2003, Perez 2006). In addition, there have been many reports of killer whales consuming the processing waste of Bering Sea groundfish trawl fishing vessels (Perez 2006). Resident killer whales are most likely to be involved in such fishery interactions since these whales are known to be fish eaters.

Fisheries observers report that large groups of killer whales in the Bering Sea follow vessels for days at a time, actively consuming the processing waste (NMFS-AFSC, Fishery Observer Program, unpubl. data). On some vessels, the waste is discharged in the vicinity of the vessel's propeller (NMFS, unpubl. data); consumption of the processing waste in the vicinity of the propeller may be the cause of the propeller-caused mortalities of killer whales in the trawl fisheries.

STATUS OF STOCK

The Eastern North Pacific Alaska Resident stock of killer whales is not designated as depleted under the MMPA or listed as threatened or endangered under the Endangered Species Act. The minimum abundance estimate for the Alaska Resident stock is likely underestimated because researchers continue to encounter new whales in the

Gulf of Alaska and western Alaska waters. Because the population estimate is likely to be conservative, the PBR is also conservative.

Based on currently available data, a minimum estimate of the mean annual mortality and serious injury rate due to U.S. commercial fisheries (1 whale) is less than 10% of the PBR (10% of PBR = 2.4) and, therefore, is considered to be insignificant and approaching zero mortality and serious injury rate. A minimum estimate of the total annual level of human-caused mortality and serious injury (1 whale) is not known to exceed the PBR (24). Therefore, the Eastern North Pacific Alaska Resident stock of killer whales is not classified as a strategic stock. Population trends and status of this stock relative to its Optimum Sustainable Population are currently unknown.

CITATIONS

Baird, R. W., and P. J. Stacey. 1988. Variation in saddle patch pigmentation in populations of killer whales (*Orcinus orca*) from British Columbia, Alaska, and Washington State. Can. J. Zool. 66(11):2582-2585.

Baird, R. W., P. A. Abrams, and L. M. Dill. 1992. Possible indirect interactions between transient and resident killer whales: implications for the evolution of foraging specializations in the genus *Orcinus*. Oecologia 89:125-132.

Barlow, J. 1995. The abundance of cetaceans in California waters. Part I: Ship surveys in summer and fall of 1991. Fish. Bull., U.S. 93:1-14.

Barlow, J. 1997. Preliminary estimates of cetacean abundance off California, Oregon and Washington based on a 1996 ship survey and comparisons of passing and closing modes. Southwest Fisheries Science Center Administrative Report LJ-97-11, 25 p. Available from SWFSC, NMFS, 8901 La Jolla Shores Drive, La Jolla, CA 92037.

Barrett-Lennard, L. G. 2000. Population structure and mating patterns of killer whales (*Orcinus orca*) as revealed by DNA analysis. Ph.D. Dissertation, University of British Columbia, Vancouver, BC, Canada. 97 p.

Bigg, M. A., P. F. Olesiuk, G. M. Ellis, J. K. B. Ford, and K. C. Balcomb III. 1990. Social organization and genealogy of resident killer whales (*Orcinus orca*) in the coastal waters of British Columbia and Washington State, p. 386-406. *In* P. S. Hammond, S. A. Mizroch, and G. P. Donovan (eds.), Individual Recognition of Cetaceans: Use of Photo-identification and Other Techniques to Estimate Population Parameters. Rep. Int. Whal. Comm. (Special Issue) 12.

Black, N. A., A. Schulman-Janiger, R. L. Ternullo, and M. Guerrero-Ruiz. 1997. Killer whales of California and western Mexico: a catalog of photo-identified individuals. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-SWFSC-247, 174 p.

Braham, H. W., and M. E. Dahlheim. 1982. Killer whales in Alaska documented in the Platforms of Opportunity Program. Rep. Int. Whal. Comm. 32:643-646.

Brault, S., and H. Caswell. 1993. Pod-specific demography of killer whales (*Orcinus orca*). Ecology 74(5):1444-1454.

Breiwick, J. M. 2013. North Pacific marine mammal bycatch estimation methodology and results, 2007-2011. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-260, 40 p.

Committee on Taxonomy. 2012. List of marine mammal species and subspecies. Society for Marine Mammalogy, www.marinemammalscience.org, consulted on 12 December 2012.

Dahlheim, M. E. 1988. Killer whale (*Orcinus orca*) depredation on longline catches of sablefish (*Anoplopoma fimbria*) in Alaskan waters. NWAFC Processed Report 88-14, 31 p. Available online: http://www.afsc.noaa.gov/Publications/ProcRpt/PR%2088-14.pdf. Accessed December 2016.

Dahlheim, M. E., and C. O. Matkin. 1994. Assessment of injuries to Prince William Sound killer whales, p. 163-171. *In* T. R. Loughlin (ed.), Marine Mammals and the *Exxon Valdez*. Academic Press, Inc., San Diego, CA.

Dahlheim, M. E., and J. M. Waite. 1993. Abundance and distribution of killer whales (*Orcinus orca*) in Alaska in 1992. Annual report to the MMPA Assessment Program, Office of Protected Resources, NMFS, NOAA, 1335 East-West Highway, Silver Spring, MD 20910.

Dahlheim, M. E., and P. A. White. 2010. Ecological aspects of transient killer whales (*Orcinus orca*) as predators in southeastern Alaska. Wildl. Biol. 16:308-322.

Dahlheim, M. E., D. Ellifrit, and J. Swenson. 1997. Killer Whales of Southeast Alaska: A Catalogue of Photoidentified Individuals. Day Moon Press, Seattle, WA. 82 p. + appendices.

Dahlheim, M. E., A. Schulman-Janiger, N. Black, R. Ternullo, D. Ellifrit, and K. C. Balcomb. 2008. Eastern temperate North Pacific offshore killer whales (*Orcinus orca*): occurrence, movements, and insights into feeding ecology. Mar. Mammal Sci. 24:719-729.

DeMaster, D. P. 1996. Minutes from the 11-13 September 1996 meeting of the Alaska Scientific Review Group, Anchorage, Alaska. 20 p. + appendices. Available from Alaska Fisheries Science Center, 7600 Sand Point Way NE, Seattle, WA 98115.

Ford, J. K. B. 1989. Acoustic behaviour of resident killer whales (*Orcinus orca*) off Vancouver Island, British Columbia. Can. J. Zool. 67(3):727-745.

Ford, J. K. B. 1991. Vocal traditions among resident killer whales (*Orcinus orca*) in coastal waters of British Columbia. Can. J. Zool. 69(6):1454-1483.

Ford, J. K. B. 2011. Killer whales of the Pacific Northwest coast: from pest to paragon. Whalewatcher 40(1):15-23.

Ford, J. K. B., and H. D. Fisher. 1982. Killer whale (*Orcinus orca*) dialects as an indicator of stocks in British Columbia. Rep. Int. Whal. Comm. 32:671-679.

Ford, J. K. B., G. Ellis, and K. C. Balcomb. 1994. Killer Whales: The Natural History and Genealogy of *Orcinus orca* in British Columbia and Washington State. University of British Columbia Press, Vancouver, BC, and University of Washington Press, Seattle. 102 p.

Ford, J. K. B., G. M. Ellis, and K. C. Balcomb. 2000. Killer Whales: The Natural History and Genealogy of *Orcinus orca* in British Columbia and Washington State. Second edition. University of British Columbia Press, Vancouver, BC, Canada. 104 p.

Forney, K. A., and P. R. Wade. 2006. World-wide abundance and density of killer whales, p. 145-162. *In* J. A. Estes, D. P. DeMaster, D. F. Doak, T. M. Williams, and R. L. Brownell, Jr. (eds.), Whales, Whaling, and Ocean Ecosystems. University of California Press.

Forney, K. A., J. Barlow, and J. V. Carretta. 1995. The abundance of cetaceans in California waters. Part II: Aerial surveys in winter and spring of 1991 and 1992. Fish. Bull., U.S. 93:15-26.

Goley, P. D., and J. M. Straley. 1994. Attack on gray whales (*Eschrichtius robustus*) in Monterey Bay, California, by killer whales (*Orcinus orca*) previously identified in Glacier Bay, Alaska. Can. J. Zool. 72:1528-1530.

Green, G. A., J. J. Brueggeman, R. A. Grotefendt, C. E. Bowlby, M. L. Bonnell, and K. C. Balcomb. 1992. Cetacean distribution and abundance off Oregon and Washington, 1989-1990, p. 1-100. *In* J. J. Brueggeman (ed.), Oregon and Washington marine mammal and seabird surveys. Final Report OCS Study MMS 91-0093.

Hoelzel, A. R., and G. A. Dover. 1991. Genetic differentiation between sympatric killer whale populations. Heredity 66:191-195.

Hoelzel, A. R., M. E. Dahlheim, and S. J. Stern. 1998. Low genetic variation among killer whales (*Orcinus orca*) in the eastern North Pacific, and genetic differentiation between foraging specialists. J. Hered. 89:121-128.

Hoelzel, A. R., A. Natoli, M. Dahlheim, C. Olavarria, R. Baird and N. Black. 2002. Low worldwide genetic diversity in the killer whale (*Orcinus orca*): implications for demographic history. Proc. R. Soc. Lond. 269:1467-1473.

Leatherwood, J. S., and M. E. Dahlheim. 1978. Worldwide distribution of pilot whales and killer whales. Naval Ocean Systems Center, Tech. Rep. 443:1-39.

Leatherwood, S., C. O. Matkin, J. D. Hall, and G. M. Ellis. 1990. Killer whales, *Orcinus orca*, photo-identified in Prince William Sound, Alaska 1976 to 1987. Can. Field Nat. 104:362-371.

Matkin, C. O., G. M. Ellis, M. E. Dahlheim, and J. Zeh. 1994. Status of killer whales in Prince William Sound, 1985-1992, p. 141-162. *In* T. R. Loughlin (ed.), Marine Mammals and the *Exxon Valdez*. Academic Press, Inc., San Diego, CA.

Matkin, C., G. Ellis, E. Saulitis, L. Barrett-Lennard, and D. Matkin. 1999. Killer Whales of Southern Alaska. North Gulf Oceanic Society. 96 p.

Matkin, C. O., G. Ellis, L. Barrett-Lennard, H. Yurk, E. Saulitis, D. Scheel, P. Olesiuk, and G. Ylitalo. 2003. Photographic and acoustic monitoring of killer whales in Prince William Sound and Kenai Fjords. *Exxon Valdez* Oil Spill Restoration Project 030012, Final Report, North Gulf Oceanic Society, 60920 Mary Allen Ave, Homer, AK 99603. 118 p.

Matkin, C. O, L. Barrett-Lennard, H. Yurk, D. Ellifrit, and A. Trites. 2007. Ecotypic variation and predatory behavior of killer whales *Orcinus orca* in the eastern Aleutian Islands, Alaska. Fish. Bull., U.S. 105:74-87.

Matkin, C. O., E. L. Saulitis, G. M. Ellis, P. Olesiuk, and S. D. Rice. 2008. Ongoing population-level impacts on killer whales *Orcinus orca* following the 'Exxon Valdez' oil spill in Prince William Sound, Alaska. Mar. Ecol. Prog. Ser. 356:269-281.

Matkin, C. O., G. Ellis, D. Herman, E. Saulitis, R. Andrews, A. Gaylord, and H. Yurk. 2010. Monitoring, tagging, acoustics, feeding habits and restoration of killer whales in Prince William Sound/Kenai Fjords 2003-2009. EVOS Trustee Council Restoration Project 090742 Final Report, North Gulf Oceanic Society, Homer, AK.

Matkin, C. O., G. Ellis, D. Herman, E. Saulitis, R. Andrews, and A. Gaylord. In prep. Monitoring, tagging, feeding habits and restoration of killer whales in Prince William Sound/Kenai Fjords 2010-2012. EVOS Trustee Council Restoration Project 10100742 Final Report, North Gulf Oceanic Society, Homer, Alaska.

Mitchell, E. D. 1975. Report on the meeting on small cetaceans, Montreal, April 1-11, 1974. J. Fish. Res. Board Can. 32:914-916.

Morin, P. A., F. I. Archer, A. D. Foote, J. Vilstrup, E. E. Allen, P. R. Wade, J. W. Durban, K. M. Parsons, R. Pitman, L. Li, P. Bouffard, S. C. A. Nielsen, M. Rasmussen, E. Willerslev, M. T. P. Gilbert, and T. Harkins. 2010. Complete mitochondrial genome phylogeographic analysis of killer whales (*Orcinus orca*) indicates multiple species. Genome Res. 20:908-916.

Olesiuk, P. F., M. A. Bigg, and G. M. Ellis. 1990. Life history and population dynamics of resident killer whales (*Orcinus orca*) in the coastal waters of British Columbia and Washington State. Rep. Int. Whal. Comm. (Special Issue 12):209-242.

Perez, M. A. 2003. Compilation of marine mammal-fisheries interaction data from the domestic and joint venture groundfish fisheries in the U.S. EEZ of the North Pacific, 1989-2001. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-138, 145 p.

Perez, M. A. 2006. Analysis of marine mammal bycatch data from the trawl, longline, and pot groundfish fisheries of Alaska, 1998-2004, defined by geographic area, gear type, and target groundfish catch species. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-167, 194 p.

Riesch, R., L. G. Barrett-Lennard, G. M. Ellis, J. K. B. Ford, and V. B. Deecke. 2012. Cultural traditions and the evolution of reproductive isolation: ecological speciation in killer whales? Biol. J. Linn. Soc. 106:1-17.

Sigler, M. F., C. R. Lunsford, J. T. Fujioka, and S. A. Lowe. 2003. Alaska sablefish assessment for 2004. *In* Stock assessment and fishery evaluation report for the groundfish fisheries of the Bering Sea/Aleutian Islands regions. North Pacific Fishery Management Council, Anchorage, AK, Section 3:223-292.

Wade, P. R., and R. Angliss. 1997. Guidelines for assessing marine mammal stocks: report of the GAMMS Workshop April 3-5, 1996, Seattle, Washington. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-OPR-12, 93 p.

Yano, K., and M. E. Dahlheim. 1995. Killer whale, *Orcinus orca*, depredation on longline catches of bottomfish in the southeastern Bering Sea and adjacent waters. Fish. Bull., U.S. 93:355-372.

Yurk, H., L. Barrett Lennard, J. K. B. Ford, and C. O. Matkin. 2002. Cultural transmission within maternal lineages: vocal clans in resident killer whales in southern Alaska. Anim. Behav. 63:1103-1119.

Zerbini, A. N., J. M. Waite, J. Durban, R. LeDuc, M. E. Dahlheim and P. R. Wade. 2007. Estimating abundance of killer whales in the nearshore waters of the Gulf of Alaska and Aleutian Islands using line-transect sampling. Mar. Biol. 150(5):1033-1045.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

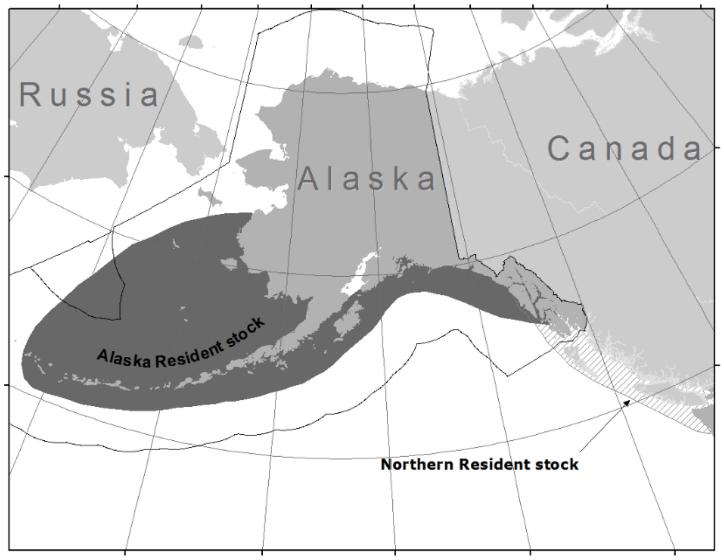

Figure 1. Approximate distribution of killer whales in the eastern North Pacific (shaded area). The distribution of the eastern North Pacific Resident and Transient stocks are largely overlapping (see text). The U.S. Exclusive Economic Zone is delineated by a black line.

Killer whales have been observed in all oceans and seas of the world (Leatherwood and Dahlheim 1978). Although reported from tropical and offshore waters, killer whales occur at higher densities in colder and more productive waters of both hemispheres, with the greatest densities found at high latitudes (Mitchell 1975, Leatherwood and Dahlheim 1978, Forney and Wade 2006). Killer whales are found throughout the North Pacific Ocean. Along the west coast of North America, seasonal and year-round occurrence of killer whales has been noted along the entire Alaska coast (Braham and Dahlheim 1982), in British Columbia and Washington inland waterways (Bigg et al. 1990), and along the outer coasts of Washington, Oregon, and California (Green et al. 1992; Barlow 1995, 1997; Forney et al. 1995). Killer whales from these areas have been labeled as “resident,” “transient,” and “offshore” type killer whales (Bigg et al. 1990, Ford et al. 2000, Dahlheim et al. 2008) 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, 2002; Barrett-Lennard 2000; Dahlheim et al. 2008). Through examination of photographs of recognizable individuals and pods, movements of whales between geographical areas have been documented. For example, whales identified in Prince William Sound have been observed near Kodiak Island (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). Movements of killer whales between the waters of Southeast Alaska and central California have also been documented (Goley and Straley 1994, Black et al. 1997, Dahlheim and White 2010).

Several studies provide evidence that the resident, offshore, and transient ecotypes are genetically distinct in both mtDNA and nuclear DNA (Hoelzel and Dover 1991; Hoelzel et al. 1998, 2002; Barrett-Lennard 2000). Genetic differences have also been found between populations within the transient and resident ecotypes (Hoelzel et al. 1998, 2002; Barrett-Lennard 2000). A global genetic study of killer whales using the entire mitochondrial genome found that some killer whale ecotypes represent deeply divergent evolutionary lineages and warrant elevation to species or subspecies status (Morin et al. 2010). In particular, estimates from mitogenome sequence data indicate that transient killer whales diverged from all other killer whale lineages approximately 700,000 years ago. In light of these differences, the Society for Marine Mammalogy’s Committee on Taxonomy currently recognizes the resident and transient North Pacific ecotypes as un-named *Orcinus orca* subspecies (Committee on Taxonomy 2018). In recognition of its status as an un-named subspecies or species, some researchers now refer to transient-type killer whales as Bigg’s killer whales (e.g., Ford 2011, Riesch et al. 2012), in tribute to the late Dr. Michael Bigg.

Acoustic data (Ford 1989, 1991; Yurk et al. 2002), association data (Bigg et al. 1990; Ford et al. 1994, 2000; Dahlheim et al. 1997; Matkin et al. 1999), and genetic data (Hoelzel et al. 1998, 2002; Barrett-Lennard 2000) confirm that Southern Residents, Northern Residents, and Alaska Residents are discrete populations. The Southern Resident population is found in summer primarily in waters of Washington State and southern British Columbia and has never been seen to associate with other resident stocks. The Eastern North Pacific Northern Resident stock is a transboundary stock and includes killer whales that frequent British Columbia, Canada, and Southeast Alaska (Dahlheim et al. 1997, Ford et al. 2000). They have been seen infrequently in Washington State waters. Members of the Northern Resident population have been documented in Southeast Alaska; however, they have not been seen to intermix with Alaska Residents (Fig. 1).

Based on data regarding association patterns, acoustics, movements, and genetic differences, eight killer whale stocks are now recognized within the Pacific U.S. Exclusive Economic Zone: 1) the Alaska Resident stock - occurring from Southeast Alaska to the Aleutian Islands and Bering Sea, 2) the Northern Resident stock - occurring from Washington State through part of Southeast Alaska (Fig. 1), 3) the Southern Resident stock - occurring mainly within the inland waters of Washington State and southern British Columbia but also in coastal waters from Southeast Alaska through California, 4) the Gulf of Alaska, Aleutian Islands, and Bering Sea Transient stock - occurring mainly from Prince William Sound through the Aleutian Islands and Bering Sea, 5) the AT1 Transient stock - occurring in Alaska from Prince William Sound through the Kenai Fjords, 6) the West Coast Transient stock - occurring from California through Southeast Alaska, 7) the Offshore stock - occurring from California through Alaska, and 8) the Hawaiian stock. Transient killer whales in Canadian waters are considered part of the West Coast Transient stock. The Hawaiian and Offshore stocks are reported in the Stock Assessment Reports for the U.S. Pacific Region.

POPULATION SIZE

Photo-identification studies since 1970 (e.g., Ford et al. 2000) have attempted to catalogue every individual belonging to the Eastern North Pacific Northern Resident population. The Canadian government published a recent summary of abundance and trends for the population (Fisheries and Oceans Canada 2019). The abundance numbers reported in that document are based on the most recent census data. They report the population was approximately 122 when first censused in 1974, and the number known to be alive in a specified year has grown over the years as the photo-identification catalogue has been updated. Note that the number reported from the Northern Resident catalogue is calculated slightly differently than the number reported in the Southern Resident catalogue; for Northern Residents, it represents the number of whales known to be alive at any time during the year, even if known or suspected to have died later in the calendar year (Fisheries and Oceans Canada 2018).

Although the majority of Northern Resident killer whales are photographed each year, it is not always possible to locate every matrilineal group during each field season, and there can remain some uncertainty about the status of missing individuals until their death is confirmed in subsequent years. For this reason, the census reports a minimum and a maximum population size, as well as a "best" number derived from the best estimates of the year of birth and year of death of individuals. For 2018, the total best population size was estimated at 302 individuals (range = 302 to 310).

Minimum Population Estimate

The technique used for estimating abundance of Northern Resident killer whales is a direct count of individually identifiable animals known to be alive in a specified year. Because this population has been studied for such a long time, each individual is well documented and, except for births, no new individuals are expected to be discovered. For populations with a statistical estimate of the overall population size (i.e., N_{BEST}) and its associated precision (i.e., coefficient of variation CV(N)), the minimum population estimate can be substantially lower than the best estimate of abundance. This is not the case here, as the minimum population estimate of 302 whales reported in Fisheries and Oceans Canada (2019) can serve as a minimum count of the population.

Thus, the minimum population estimate (N_{MIN}) for the Northern Resident stock of killer whales is 302 whales, which includes whales found in Canadian waters (see PBR Guidelines (NMFS 2016) regarding the status of migratory transboundary stocks). Information on the percentage of time animals typically encountered in Canadian waters spend in U.S. waters is unquantified.

Current Population Trend

Trends for this population have been recently summarized and contrasted with trends for the Southern Resident population (Fisheries and Oceans Canada 2018). From the mid-1970s to the 1990s, the Northern Resident killer whale population increased at an annual rate of 2.6% (i.e., from 122 whales in 1974 to 218 in 1997). A decline was reported from 1998 to 2001 at a rate of 7% per year. The increased mortality that drove this decline coincided with a period of reduced range-wide Chinook salmon abundance, their primary prey (Ford et al. 2010). Then, after 2001, the growth was positive again with the population increasing at an average rate of 2.9% per year from 2002 to 2014. At the end of the 2015 field season, 290 whales were catalogued alive for the 2014 assessment.

This represents an average annual increase of 2.2% over the 40-year time series (Towers et al. 2015). However, annual Northern Resident killer whale population growth rates have slowed over the past five census years, from 5.1% in 2014 to -0.3% in 2018 (Fisheries and Oceans Canada 2019).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

As summarized in the previous paragraph, studies of Northern Resident killer whale pods in British Columbia and Washington waters resulted in estimated population growth rates of 2.6% from 1974 to 1997 and 2.9% from 2002 to 2014 (Towers et al. 2015), separated by a short period of decline from 1998 to 2001. The period from 2002 to 2014 was a period of maximum growth for this population when it grew at an average rate of 2.9% per year. Therefore, the maximum net productivity rate (R_{MAX}) is estimated to be 2.9% (Towers et al. 2015).

POTENTIAL BIOLOGICAL REMOVAL

Potential biological removal (PBR) is defined as the product of the minimum population estimate, one-half the maximum estimated net productivity rate, and a recovery factor: $PBR = N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 0.5, the value for cetacean stocks with unknown population status (NMFS 2016). Thus, for the Eastern North Pacific Northern Resident killer whale stock, PBR = 2.2 animals (302 × 0.0145 × 0.5).

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2013 and 2017 is listed, by marine mammal stock, in Delean et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports. The minimum estimated mean annual level of human-caused mortality and serious injury for Northern Resident killer whales between 2013 and 2017 is 0.2 killer whales in unknown (commercial, recreational, or subsistence) fisheries. Potential threats most likely to result in direct human-caused mortality or serious injury of this stock include oil spills, vessel strikes, and interactions with fisheries.

Fisheries Information

Information on U.S. commercial fisheries in Alaska waters (including observer programs, observer coverage, and observed incidental takes of marine mammals) is presented in Appendices 3-6 of the Alaska Stock Assessment Reports.

Incidental mortality or serious injury of Northern Resident killer whales has not been observed in federally-managed or state-managed U.S. commercial fisheries which operate within the range of this stock; however, the state-managed fisheries are not observed or have not been observed in a long time.

Reports from the NMFS Alaska Region stranding network of killer whales entangled in fishing gear or with injuries caused by interactions with gear are another source of mortality and serious injury data. There was one report of a killer whale entangled in pot gear in Icy Strait in 2016, resulting in a mean annual mortality and serious injury rate of 0.2 killer whales in unknown (commercial, recreational, or subsistence) Southeast Alaska pot fisheries between 2013 and 2017 (Table 1; Delean et al. 2020). Because the killer whale stock identification is unknown, this mortality and serious injury was assigned to the three killer whale stocks that occur in the area: the Eastern North Pacific Alaska Resident, Eastern North Pacific Northern Resident, and West Coast Transient stocks. This mortality and serious injury estimate results from an actual count of verified human-caused deaths and serious injuries and is a minimum because not all entangled animals strand nor are all stranded animals found or reported.

All Canadian longline fisheries (including halibut, rockfish, dogfish, sablefish, jig for lingcod, and troll for lingcod and Chinook salmon) are monitored by observers or video. However, only groundfish trawl fisheries have observer or electronic monitoring in Canada, whereas, trawl fisheries for krill, scallop, and shrimp have no observer coverage and salmon net fisheries are not observed (T. Doniol-Valcroze, pers. comm., Department of Fisheries and Oceans, BC, Canada, 14 May 2019). The interaction of Alaska resident killer whales with the sablefish longline fishery accounts for a large proportion of the commercial fishing/killer whale interactions in Alaska waters. Such interactions have not been reported in Canadian waters where sablefish are taken via a pot fishery; however, Northern Resident killer whale interactions with Pacific halibut longline and salmon troll fisheries in British Columbia have been reported (Ford 2014). Reports of killer whale interactions with gillnets in Canadian waters include one killer whale that contacted a salmon gillnet in 1994 but did not entangle (Guenther et al. 1995) and one killer whale (Northern Resident I103) that entangled in a gillnet in 2014 but was quickly released (Fisheries and Oceans Canada 2018).

Table 1. Summary of mortality and serious injury of Northern Resident killer whales, by year and type, reported to the NMFS Alaska Region marine mammal stranding network between 2013 and 2017 (Delean et al. 2020).

Cause of injury	2013	2014	2015	2016	2017	Mean annual mortality
Entangled in Southeast Alaska pot gear*	0	0	0	1[a]	0	0.2
*Total in unknown (commercial, recreational, or subsistence) fisheries						0.2

[a]This mortality and serious injury was assigned to the Eastern North Pacific Alaska Resident, Eastern North Pacific Northern Resident, and West Coast Transient stocks of killer whales since the stock is unknown and these three stocks overlap in the area where the event occurred.

Subsistence/Native Harvest Information

Killer whales are not harvested for subsistence in Alaska.

Other Mortality

Collisions of killer whales with vessels occur occasionally. One ship-strike mortality of a Northern Resident killer whale (C21) in Prince Rupert, BC, was reported in 2006 (Williams and O'Hara 2010). The shooting of killer whales in Canadian waters has been a concern in the past. Since 1974, however, fresh bullet wounds are rarely, if ever, seen on whales in British Columbia and Washington (Ford et al. 2000, Fisheries and Oceans Canada 2018).

Other Issues

Killer whales are known to depredate longline catches in the Bering Sea (Dahlheim 1988; Yano and Dahlheim 1995; Perez 2003, 2006; Sigler et al. 2003) and in the Gulf of Alaska (Sigler et al. 2003, Perez 2006). In Canada, Northern Resident killer whales have been reported to depredate fish from both commercial salmon trollers and recreational sportfishermen, as well as Pacific halibut longliners (Ford 2014). Most reports occur in the northern half of the coast, especially Dixon Entrance, and early in the season (April to June), although some are scattered throughout the summer (J. Ford, pers. comm., Department of Fisheries and Oceans, BC, Canada, 3 December 2012).

STATUS OF STOCK

The Northern Resident killer whale stock is not designated as depleted under the MMPA or listed as threatened or endangered under the Endangered Species Act. In 2001, the Committee on the Status of Endangered Wildlife in Canada designated Northern Resident killer whales in British Columbia as threatened and listed in Schedule 1 of the Species at Risk Act (SARA) for Canada. Resident killer whales in British Columbia are considered to be at risk based on their small population size, low reproductive rate, and the existence of a variety of anthropogenic threats that have the potential to prevent recovery or to cause further declines (Fisheries and Oceans Canada 2008). Monitoring of fisheries in BC over the past decade has been quite extensive and likely at the same level as in U.S. waters. One serious injury from an entanglement in unidentified pot gear was reported in Alaska waters in 2016 and a Northern Resident killer whale entangled in a gillnet in British Columbia waters in 2014 but was quickly released. Northern Resident killer whale interactions with longline and troll fisheries in British Columbia waters have also been reported.

Based on currently available data, the minimum estimated mean annual U.S. commercial fishery-related mortality and serious injury rate is zero, which does not exceed 10% of the PBR (10% of PBR = 0.22) and, therefore, is considered to be insignificant and approaching a zero mortality and serious injury rate. The minimum estimated mean annual level of human-caused mortality and serious injury (0.2) is not known to exceed the PBR (2.2). Therefore, the Eastern North Pacific Northern Resident stock of killer whales is not classified as a strategic stock. Status of this stock relative to its Optimum Sustainable Population size has not been quantified.

There are few other uncertainties in the assessment of the Northern Resident stock of killer whales. Individual whales can be counted annually and the stock increased at an average rate of 2.9% per year from 2002 to 2014, although the growth rate has slowed in the last five census years.

HABITAT CONCERNS

Ford et al. (2005) showed that a sharp drop in coast-wide Chinook salmon abundance during the late 1990s was correlated with a significant decline in resident killer whale survival. They noted that the whales' preference for

Chinook salmon is likely due to this species' relatively large size, high lipid content and, unlike other salmonids, its year-round presence in the whales' range. They further note that resident killer whales may be especially dependent on Chinook during winter, when this species is the primary salmonid available in coastal waters, and the whales may be subject to nutritional stress leading to increased mortality if the quantity and/or quality of this prey resource declines.

Environmental contaminants and vessel traffic, particularly increased whale-watching activity, are other potential concerns for this stock (Fisheries and Oceans Canada 2018).

CITATIONS

Baird, R. W., and P. J. Stacey. 1988. Variation in saddle patch pigmentation in populations of killer whales (*Orcinus orca*) from British Columbia, Alaska, and Washington State. Can. J. Zool. 66(11):2582-2585.

Baird, R. W., P. A. Abrams, and L. M. Dill. 1992. Possible indirect interactions between transient and resident killer whales: implications for the evolution of foraging specializations in the genus *Orcinus*. Oecologia 89:125-132.

Barlow, J. 1995. The abundance of cetaceans in California waters. Part I: Ship surveys in summer and fall of 1991. Fish. Bull., U.S. 93:1-14.

Barlow, J. 1997. Preliminary estimates of cetacean abundance off California, Oregon and Washington based on a 1996 ship survey and comparisons of passing and closing modes. Southwest Fisheries Science Center Administrative Report LJ-97-11. 25 p. Available from Southwest Fisheries Science Center, NMFS, 8901 La Jolla Shores Drive, La Jolla, CA 92037.

Barrett-Lennard, L. G. 2000. Population structure and mating patterns of killer whales (*Orcinus orca*) as revealed by DNA analysis. Ph.D. Dissertation, University of British Columbia, Vancouver, BC, Canada. 97 p.

Bigg, M. A., P. F. Olesiuk, G. M. Ellis, J. K. B. Ford, and K. C. Balcomb III. 1990. Social organization and genealogy of resident killer whales (*Orcinus orca*) in the coastal waters of British Columbia and Washington State, p. 386-406. *In* P. S. Hammond, S. A. Mizroch, and G. P. Donovan (eds.), Individual Recognition of Cetaceans: Use of Photo-identification and Other Techniques to Estimate Population Parameters. Rep. Int. Whal. Comm. Special Issue 12.

Black, N. A., A. Schulman-Janiger, R. L. Ternullo, and M. Guerrero-Ruiz. 1997. Killer whales of California and western Mexico: a catalogue of photo-identified individuals. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-SWFSC-247, 174 p.

Braham, H. W., and M. E. Dahlheim. 1982. Killer whales in Alaska documented in the Platforms of Opportunity Program. Rep. Int. Whal. Comm. 32:643-646.

Committee on Taxonomy. 2018. List of marine mammal species and subspecies. Society for Marine Mammalogy. Available online: www.marinemammalscience.org . Accessed December 2019.

Dahlheim, M. E. 1988. Killer whale (*Orcinus orca*) depredation on longline catches of sablefish (*Anoplopoma fimbria*) in Alaskan waters. NWAFC Processed Report 88-14, 31 p. Available online: http://www.afsc.noaa.gov/Publications/ProcRpt/PR%2088-14.pdf . Accessed December 2019.

Dahlheim, M. E., and P. A. White. 2010. Ecological aspects of transient killer whales (*Orcinus orca*) as predators in southeastern Alaska. Wildl. Biol. 16:308-322.

Dahlheim, M. E., D. Ellifrit, and J. Swenson. 1997. Killer Whales of Southeast Alaska: A Catalogue of Photoidentified individuals. Day Moon Press, Seattle, WA. 82 p. + appendices.

Dahlheim, M. E. A. Schulman-Janiger, N. Black, R. Ternullo, D. Ellifrit, and K. C. Balcomb. 2008. Eastern temperate North Pacific offshore killer whales (*Orcinus orca*): occurrence, movements, and insights into feeding ecology. Mar. Mammal Sci. 24(3):719-729.

Delean, B. J., V. T. Helker, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, J. Jannot, and N. C. Young. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2013-2017. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-401, 86 p.

Fisheries and Oceans Canada. 2008. Recovery strategy for the Northern and Southern Resident killer whales (*Orcinus orca*) in Canada. Species at Risk Act Recovery Strategy Series, Fisheries and Oceans Canada, Ottawa. ix + 81 p.

Fisheries and Oceans Canada. 2018. Recovery strategy for the Northern and Southern Resident killer whales (*Orcinus orca*) in Canada. Species at Risk Act Recovery Strategy Series, Fisheries and Oceans Canada, Ottawa. x + 84 p.

Fisheries and Oceans Canada. 2019. Population status update for the Northern Resident killer whale (*Orcinus orca*) in 2018. DFO Canadian Science Advisory Secretariat Science Response 2019/025. 13 p.

Ford, J. K. B. 1989. Acoustic behaviour of resident killer whales (*Orcinus orca*) off Vancouver Island, British Columbia. Can. J. Zool. 67(3):727-745.

Ford, J. K. B. 1991. Vocal traditions among resident killer whales (*Orcinus orca*) in coastal waters of British Columbia. Can. J. Zool. 69(6):1454-1483.

Ford J. K. B. 2011. Killer whales of the Pacific Northwest coast: from pest to paragon. Whale Watcher 40(1):15-23.

Ford, J. K. B. 2014. Marine Mammals of British Columbia. Royal BC Museum Handbook, Mammals of BC, Volume 6. Royal BC Museum, Victoria. 460 p.

Ford, J. K. B., and H. D. Fisher. 1982. Killer whale (*Orcinus orca*) dialects as an indicator of stocks in British Columbia. Rep. Int. Whal. Comm. 32:671-679.

Ford, J. K. B., G. Ellis, and K. C. Balcomb. 1994. Killer Whales: The Natural History and Genealogy of *Orcinus orca* in British Columbia and Washington State. University of British Columbia Press, Vancouver BC, and University of Washington Press, Seattle. 102 p.

Ford, J. K. B., G. M. Ellis, and K. C. Balcomb. 2000. Killer Whales: The Natural History and Geneology of *Orcinus orca* in British Columbia and Washington State. Second edition. University of British Columbia Press, Vancouver, BC, Canada. 104 p.

Ford, J. K. B., G. M. Ellis, and P. F. Olesiuk. 2005. Linking prey and population dynamics: did food limitation cause recent declines of 'resident' killer whales (*Orcinus orca*) in British Columbia? Canadian Science Advisory Secretariat Research Document 2005/042.

Ford, J. K. B., G. M. Ellis, P. F. Olesiuk, and K. C. Balcomb. 2010. Linking killer whale survival and prey abundance: food limitation in the oceans' apex predator? Biol. Lett. 6:139-142. DOI: dx.doi.org/10.1098/rsbl.2009.0468 .

Forney, K. A., and P. R. Wade. 2006. World-wide abundance and density of killer whales, p. 145-162. In J. A. Estes, D. P. DeMaster, D. F. Doak, T. M. Williams, and R. L. Brownell, Jr. (eds.), Whales, Whaling, and Ocean Ecosystems. University of California Press.

Forney, K. A., J. Barlow, and J. V. Carretta. 1995. The abundance of cetaceans in California waters. Part II: Aerial surveys in winter and spring of 1991 and 1992. Fish. Bull., U.S. 93:15-26.

Goley, P. D., and J. M. Straley. 1994. Attack on gray whales (*Eschrichtius robustus*) in Monterey Bay, California, by killer whales (*Orcinus orca*) previously identified in Glacier Bay, Alaska. Can. J. Zool. 72:1528-1530.

Green, G. A., J. J. Brueggeman, R. A. Grotefendt, C. E. Bowlby, M. L. Bonnell, and K. C. Balcomb. 1992. Cetacean distribution and abundance of Oregon and Washington, 1989-1990, p. 1-100. *In* J. J. Brueggeman (ed.), Oregon and Washington marine mammal and seabird surveys. Final Report OCS Study MMS 91-0093.

Guenther, T. J., R. W. Baird, R. L. Bates, P. M. Willis, R. L. Hahn, and S. G. Wischniowski. 1995. Strandings and fishing gear entanglements of cetaceans of the west coast of Canada in 1994. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/47/O6). 7 p.

Hoelzel, A. R., and G. A. Dover. 1991. Genetic differentiation between sympatric killer whale populations. Heredity 66: 191-195.

Hoelzel, A. R., M. E. Dahlheim, and S. J. Stern. 1998. Low genetic variation among killer whales (*Orcinus orca*) in the eastern North Pacific, and genetic differentiation between foraging specialists. J. Hered. 89:121-128.

Hoelzel, A. R., A. Natoli, M. Dahlheim, C. Olavarria, R. Baird, and N. Black. 2002. Low worldwide genetic diversity in the killer whale (*Orcinus orca*): implications for demographic history. Proc. R. Soc. Lond. 269:1467-1473.

Leatherwood, J. S., and M. E. Dahlheim. 1978. Worldwide distribution of pilot whales and killer whales. Naval Ocean Systems Center, Tech. Rep. 443:1-39.

Leatherwood, S., C. O. Matkin, J. D. Hall, and G. M. Ellis. 1990. Killer whales, *Orcinus orca*, photo-identified in Prince William Sound, Alaska 1976 to 1987. Can. Field Nat. 104:362-371.

Matkin, C., G. Ellis, E. Saulitis, L. Barrett-Lennard, and D. Matkin. 1999. Killer Whales of Southern Alaska. North Gulf Oceanic Society, Homer, AK. 96 p.

Mitchell, E. D. 1975. Report on the meeting on small cetaceans, Montreal, April 1-11, 1974. J. Fish. Res. Board Can. 32:914-916.

Morin P. A., F. I. Archer, A. D. Foote, J. Vilstrup, E. E. Allen, P. Wade, J. Durban, K. Parsons, R. Pitman, L. Li, P. Bouffard, S. C. Abel Nielsen, M. Rasmussen, E. Willerslev, M. T. P. Gilbert, and T. Harkins. 2010. Complete mitochondrial genome phylogeographic analysis of killer whales (*Orcinus orca*) indicates multiple species. Genome Res. 20:908-916.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2019.

Perez, M. A. 2003. Compilation of marine mammal-fisheries interaction data from the domestic and joint venture groundfish fisheries in the U.S. EEZ of the North Pacific, 1989-2001. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-138, 145 p.

Perez, M. A. 2006. Analysis of marine mammal bycatch data from the trawl, longline, and pot groundfish fisheries of Alaska, 1998-2004, defined by geographic area, gear type, and target groundfish catch species. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-167, 194 p.

Riesch R., L. G. Barrett-Lennard, G. M. Ellis, J. K. B. Ford, and V. B. Deecke. 2012. Cultural traditions and the evolution of reproductive isolation: ecological speciation in killer whales? Biol. J. Linn. Soc. 106:1-17.

Sigler, M. F., C. R. Lunsford, J. T. Fujioka, and S. A. Lowe. 2003. Alaska sablefish assessment for 2004. *In* Stock assessment and fishery evaluation report for the groundfish fisheries of the Bering Sea/Aleutian Islands regions. North Pacific Fishery Management Council, Anchorage, AK, Section 3:223-292.

Towers, J. R., G. M. Ellis, and J. K. B. Ford. 2015. Photo-identification catalogue and status of the Northern Resident killer whale population in 2014. Can. Tech. Rep. Fish. Aquat. Sci. 3139:iv + 75 p.

Williams, R., and P. O'Hara. 2010. Modelling ship strike risk to fin, humpback, and killer whales in British Columbia, Canada. J. Cetacean Res. Manage. 11(1):1-8.

Yano, K., and M. E. Dahlheim. 1995. Killer whale, *Orcinus orca*, depredation on longline catches of bottomfish in the southeastern Bering Sea and adjacent waters. Fish. Bull., U.S. 93:355-372.

Yurk, H., L. Barrett-Lennard, J. K. B. Ford, and C. O. Matkin. 2002. Cultural transmission within maternal lineages: vocal clans in resident killer whales in southern Alaska. Anim. Behav. 63:1103-1119.

Revised 12/30/2020

KILLER WHALE (*Orcinus orca*): Eastern North Pacific
Gulf of Alaska, Aleutian Islands, and Bering Sea Transient Stock

NOTE – NMFS has preliminary genetic information on killer whales in Alaska which indicates that the current stock structure of killer whales in Alaska needs to be reassessed. NMFS is evaluating the new genetic information. In the interim, new information on killer whale mortality levels is provided within this report. A complete revision of the killer whale stock assessments will be postponed until the stock structure evaluation is completed and any new stocks are identified.

STOCK DEFINITION AND GEOGRAPHIC RANGE

Killer whales have been observed in all oceans and seas of the world (Leatherwood and Dahlheim 1978). Although reported from tropical and offshore waters, killer whales occur at higher densities in colder and more productive waters of both hemispheres, with the greatest densities found at high latitudes (Mitchell 1975, Leatherwood and Dahlheim 1978, Forney and Wade 2006). Killer whales are found throughout the North Pacific Ocean. Along the west coast of North America, seasonal and year-round occurrence of killer whales has been noted along the entire Alaska coast (Braham and Dahlheim 1982), in British Columbia and Washington inland waterways (Bigg et al. 1990), and along the outer coasts of Washington, Oregon, and California (Green et al. 1992; Barlow 1995, 1997; Forney et al. 1995). Killer whales from these areas have been labeled as "resident," "transient," and "offshore" type killer whales (Bigg et al. 1990, Ford et al. 2000, Dahlheim et al. 2008) 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, 2002; Barrett-Lennard 2000; Dahlheim et al. 2008). Through examination of photographs of recognizable individuals and pods, movements of whales between geographical areas have been documented. For example, whales identified in Prince William Sound have been observed near Kodiak Island (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). Movements of killer whales between the waters of Southeast Alaska and central California have also been documented (Goley and Straley 1994, Black et al. 1997, Dahlheim and White 2010).

Figure 1. Approximate distribution of transient killer whales in the eastern North Pacific (shaded areas). The distribution of resident and transient killer whale stocks in the eastern North Pacific largely overlap (see text). The U.S. Exclusive Economic Zone is delineated by a black line.

Several studies provide evidence that the resident, offshore, and transient ecotypes are genetically distinct in both mtDNA and nuclear DNA (Hoelzel and Dover 1991; Hoelzel et al. 1998, 2002; Barrett-Lennard 2000). Genetic differences have also been found between populations within the transient and resident ecotypes (Hoelzel et al. 1998, 2002; Barrett-Lennard 2000). A global genetic study of killer whales using the entire mitochondrial genome found that some killer whale ecotypes represent deeply divergent evolutionary lineages and warrant elevation to species or subspecies status (Morin et al. 2010). In particular, estimates from mitogenome sequence data indicate that transient killer whales diverged from all other killer whale lineages approximately 700,000 years ago. In light of these differences, the Society for Marine Mammalogy's Committee on Taxonomy currently recognizes the resident and transient North Pacific ecotypes as un-named *Orcinus orca* subspecies (Committee on Taxonomy 2019). In recognition of its status as an un-named subspecies or species, some researchers now refer to transient-type killer whales as Bigg's killer whales (e.g., Ford 2011, Riesch et al. 2012), in tribute to the late Dr. Michael Bigg.

The first studies of transient killer whales in Alaska were conducted in Southeast Alaska and in the Gulf of Alaska (from Prince William Sound, through the Kenai Fjords, and around Kodiak Island). In the Gulf of Alaska, Matkin et al. (1999) described two genetically distinct populations of transients which were never found in association with one another, the so-called "Gulf of Alaska" transients and "AT1" transients. In the past, neither of these populations were known to associate with the population of transient killer whales that ranged from California to Southeast Alaska, which are described as the West Coast Transient stock. Gulf of Alaska transients are documented throughout the Gulf of Alaska, including occasional sightings in Prince William Sound. AT1 transients have been seen only in Prince William Sound and in the Kenai Fjords region, and are therefore partially sympatric with Gulf of Alaska transients. In addition, 14 out of 217 transients on the outer coast of Southeast Alaska and British Columbia were identified as Gulf of Alaska transients and in one encounter they were observed mixing with West Coast transients (Matkin et al. 2012, Ford et al. 2013). Transients within the Gulf of Alaska population have been found to have two mtDNA haplotypes, neither of which is found in the West Coast or AT1 populations. Members of the AT1 population share a single mtDNA haplotype. Transient killer whales from the West Coast population have been found to share a single mtDNA haplotype that is not found in the other populations. Additionally, all three populations have been found to have significant differences in nuclear (microsatellite) DNA (Barrett-Lennard 2000). Acoustic differences have been found as well; Saulitis et al. (2005) described acoustic differences between Gulf of Alaska transients and AT1 transients. For these reasons, the Gulf of Alaska transients are considered part of a population that is discrete from the AT1 population, and both of these populations are considered discrete from the West Coast transients.

Transient-type killer whales from the Aleutian Islands and Bering Sea are currently considered to be part of a single population that includes Gulf of Alaska transients; however, recent genetic analyses suggest substructure within the region. Biopsy samples from the eastern Aleutians and the south side of the west end of the Alaska Peninsula have produced the same haplotypes as killer whales in the northern Gulf of Alaska; however, nuclear DNA analysis strongly suggests they belong to a separate population (Parsons et al. 2013). The geographic distribution of mtDNA haplotypes revealed samples from the central Aleutian Islands and Bering Sea with haplotypes not found in Gulf of Alaska transients, suggesting additional population structure in western Alaska. Killer whales observed in the northern Bering Sea and north and east to the western Beaufort Sea have characteristics of transient-type whales, but little is known about these whales (Braham and Dahlheim 1982, George and Suydam 1998). AT1 haplotype whales are also present west of the Aleutian Islands and into the Bering Sea; however, nuclear DNA analysis indicates these animals are not part of the AT1 transient population in the Gulf of Alaska (Parsons et al. 2013).

In summary, within the transient ecotype, association data (Ford et al. 1994, Ford and Ellis 1999, Matkin et al. 1999), acoustic data (Ford and Ellis 1999, Saulitis et al. 2005), and genetic data (Hoelzel et al. 1998, 2002; Barrett-Lennard 2000) confirm that at least three communities of transient whales exist and represent three discrete populations: 1) Gulf of Alaska, Aleutian Islands, and Bering Sea transients, 2) AT1 transients, and 3) West Coast transients.

Based on data regarding association patterns, acoustics, movements, and genetic differences, eight killer whale stocks are now recognized within the Pacific U.S. Exclusive Economic Zone: 1) the Alaska Resident stock - occurring from Southeast Alaska to the Aleutian Islands and Bering Sea, 2) the Northern Resident stock - occurring from Washington State through part of Southeast Alaska, 3) the Southern Resident stock - occurring mainly within the inland waters of Washington State and southern British Columbia, but also in coastal waters from Southeast Alaska through California, 4) the Gulf of Alaska, Aleutian Islands, and Bering Sea Transient stock - occurring mainly from Prince William Sound through the Aleutian Islands and Bering Sea (Fig. 1), 5) the AT1 Transient stock - occurring in Alaska from Prince William Sound through the Kenai Fjords, 6) the West Coast Transient stock - occurring from California through Southeast Alaska, 7) the Offshore stock - occurring from California through Alaska, and 8) the Hawaiian stock. Transient killer whales in Canadian waters are considered part of the West Coast Transient stock. The Hawaiian and Offshore stocks are reported in the Stock Assessment Reports for the U.S. Pacific Region.

POPULATION SIZE

In January 2004, the North Gulf Oceanic Society (NGOS) and the Marine Mammal Laboratory (MML) held a joint workshop to match identification photographs of transient killer whales from this population. That analysis of photographic data resulted in the following minimum counts for transient killer whales belonging to the Gulf of Alaska, Aleutian Islands, and Bering Sea Transient stock. In the Gulf of Alaska (east of the Shumagin Islands), 82 whales were identified by NGOS, including whales from Matkin et al. (1999) as well as whales identified in subsequent years (but not including whales identified as part of the AT1 population). MML identified

43 whales and 11 matches were found between the NGOS and MML catalogues. Since that time an additional 22 whales have been added to the NGOS catalogue (Matkin et al. 2013). Therefore, a total of 136 transients (104 + 43 - 11) have been identified in the Gulf of Alaska. In the Aleutian Islands (west of and including the Shumagin Islands) and Bering Sea, the combined NGOS/MML catalogue (NGOS/MML 2012) now contains 451 individually identifiable whales (not counting unmarked calves and not counting two Gulf of Alaska transient whales that have been photographed in that region). Combining the Aleutian Islands and Bering Sea count (451) with the Gulf of Alaska count (136), a total count of 587 individual whales have been identified in catalogues of this stock.

MML conducted killer whale line-transect surveys for 3 years in July and August in 2001-2003. These surveys covered an area from approximately Resurrection Bay in the Kenai Fjords to the central Aleutians. The surveys covered an area from shore to 30-45 nautical miles offshore, with randomly located transects in a zigzag pattern. Estimated transient killer whale abundance from these surveys, using post-encounter estimates of group size, was 249 (CV = 0.50), with a 95% confidence interval of 99-628 (Zerbini et al. 2007).

Mark-recapture methods were used to estimate the number of transient killer whales using the coastal waters from the central Gulf of Alaska to the central Aleutian Islands, using photographs collected during the three line-transect surveys (Zerbini et al. 2007), along with photographs collected from a variety of additional surveys during the same time period (Durban et al. 2010). A total of 154 individuals were identified from 6,489 photographs collected between July 2001 and August 2003. A Bayesian mixture model estimated seven distinct clusters (95% Probability Interval = 7-10) of individuals that were differentially covered by 14 boat-based surveys exhibiting varying degrees of association in space and time, leading to a total estimate of 345 whales (95% Probability Interval = 255-487). This estimate is higher than the line-transect estimate for at least two reasons. First, the line-transect estimate provides an "instantaneous" (across ~40 days) estimate of the average number of transient killer whales in the survey area, whereas the mark-recapture methods provide an estimate of the total number of whales to use the survey area over the 3 years, which is known to be greater due to the long distance movements documented by satellite tags (J. Durban, Southwest Fisheries Science Center, pers. comm.). Second, the mark-recapture estimate included photographic data from a broader seasonal time period and, therefore, includes transient killer whales documented in the False Pass/Unimak Island area in spring where they aggregate to prey on gray whales on migration (Matkin et al. 2007). Many of these whales have not been seen in that region in the summer. However, mark recapture estimates do not include most of the Bering Sea and Pribilof Islands.

It should be noted that the photographic catalogue encompasses a larger area, including some data from areas such as the Bering Sea and Pribilof Islands that were outside the line-transect survey area. The photo catalogue also encompasses a much longer time period (through 2012). Additionally, the number of whales in the photographic catalogue is a documentation of all whales seen in the area over the time period of the catalogue; movements of some individual whales have been documented between the line-transect survey area and locations outside the survey area. Accordingly, a larger number of transient killer whales may use the line-transect survey area at some point over the 3 years than would necessarily be found at one time in the survey area in July and August in a particular year.

Minimum Population Estimate

A total count of 587 individual whales have been identified in the photograph catalogues from the Gulf of Alaska (Matkin et al. 2013) and from western Alaska (NGOS/MML 2012). The photograph catalogue estimate of transient killer whales is a direct count of individually identifiable animals. However, the number of catalogued whales does not necessarily represent the number of live animals. Some animals may have died, but whales cannot be presumed dead if not resighted because long periods of time between sightings are common for some transient animals. The catalogue for the western area used data only from 2001-2012, decreasing the potential bias from using whales that may have died prior to the end of the time period. However, given that researchers continue to identify new whales and the entire range has not been surveyed, the estimate of abundance based on the number of uniquely identified individuals catalogued is likely conservative.

Thus, the minimum population estimate (N_{MIN}) for the Gulf of Alaska, Aleutian Islands, and Bering Sea Transient stock of killer whales is 587 animals based on the count of individuals using photo-identification.

Current Population Trend

Matkin et al. (2012) analyzed photographic data collected since 1984 and determined Gulf of Alaska transients in the northern Gulf of Alaska have had stable numbers. At present, reliable data on trends in population abundance for the Aleutian Islands and Bering Sea portion of this stock of killer whales are not available.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate (R_{MAX}) is not available for the Gulf of Alaska, Aleutian Islands, and Bering Sea Transient stock of killer whales. Between 2012 and 2018, Towers et al. (2019) observed a mean annual growth rate of 4.1% for a population subset of transient killer whales in Canadian coastal waters, which was higher than the mean annual growth rate of 2.7% documented by Ford et al. (2013) between 2006 and 2011 for a sub-population of inner-coast transient killer whales that contained most of the same individuals. However, until additional data become available for the Gulf of Alaska, Aleutian Islands, and Bering Sea Transient stock of killer whales, the default cetacean maximum theoretical net productivity rate (R_{MAX}) of 4% will be used for this stock (NMFS 2016).

POTENTIAL BIOLOGICAL REMOVAL

Potential biological removal (PBR) is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: PBR is $N_{MIN} \times 0.5R_{Max} \times F_R$. The recovery factor ($F_R$) for this stock is 0.5, the value for cetacean stocks with unknown population status (NMFS 2016). Thus, for the Gulf of Alaska, Aleutian Islands, and Bering Sea Transient killer whale stock, PBR is 5.9 animals (587 × 0.02 × 0.5). Although only a few individuals have been observed in Canadian waters, the proportion of time that this trans-boundary stock spends in Canadian waters cannot be determined (G. Ellis, Pacific Biological Station, Canada, pers. comm.).

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2014 and 2018 is listed, by marine mammal stock, in Young et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports. The minimum estimated mean annual level of human-caused mortality and serious injury for Gulf of Alaska, Aleutian Islands, and Bering Sea Transient killer whales between 2014 and 2018 is 0.8 killer whales in U.S. commercial fisheries. Potential threats most likely to result in direct human-caused mortality or serious injury of this stock include oil spills, vessel strikes, and interactions with fisheries.

Fisheries Information

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2014 and 2018 is listed, by marine mammal stock, in Young et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports.

Information for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is available in Appendix 3 of the Alaska Stock Assessment Reports (observer coverage) and in the NMFS List of Fisheries (LOF) and the fact sheets linked to fishery names in the LOF (observer coverage and reported incidental takes of marine mammals: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-list-fisheries, accessed December 2020).

Two of the federally-regulated U.S. commercial fisheries, monitored for incidental mortality and serious injury of marine mammals by fishery observers, incurred serious injury and mortality of killer whales of unknown stock between 2014 and 2018: the Bering Sea/Aleutian Islands flatfish trawl and Bering Sea/Aleutian Islands Greenland turbot longline fisheries (Table 1; Breiwick 2013; MML, unpubl. data).

Fishery observers have collected tissue samples from many of the killer whales that were killed incidental to U.S. commercial fisheries. Genetic analyses of samples from seven killer whales collected between 1999 and 2004 have confirmed that Alaska Resident killer whale mortality occurred incidental to the Bering Sea/Aleutian Islands flatfish trawl (n = 3) and Bering Sea/Aleutian Islands Pacific cod longline fisheries (n = 1) and that Gulf of Alaska, Aleutian Islands, and Bering Sea Transient killer whale mortality occurred incidental to the Bering Sea/Aleutian Islands pollock trawl fishery (n = 3) (M. Dahlheim, NMFS-AFSC-MML (retired), pers. comm., 20 February 2013). Given the overlap in the range of transient and resident stocks in Alaska waters, unless genetic samples can be collected from animals injured or killed by gear or the ship's propeller, these events are assigned to both the transient and resident stock occurring in that area. Thus, the estimated mean annual mortality and serious injury rate of 0.6 killer whales between 2014 and 2018 will be assigned to both the Gulf of Alaska, Aleutian Islands, and Bering Sea Transient and Alaska Resident stocks of killer whales (Table 1).

Typically, if mortality or serious injury occurs incidental to U.S. commercial fishing, it is due to interactions with the fishing gear. However, reports indicate that observed killer whale mortality incidental to Bering Sea/Aleutian Islands trawl fisheries often occurs due to contact with the ship's propeller (e.g., the 2016 mortality in the Bering Sea/Aleutian Islands flatfish trawl fishery).

Table 1. Summary of incidental mortality and serious injury of Gulf of Alaska, Aleutian Islands, and Bering Sea Transient killer whales due to U.S. commercial fisheries between 2014 and 2018 and calculation of the mean annual mortality and serious injury rate (Breiwick 2013; MML, unpubl. data). Methods for calculating percent observer coverage are described in Appendix 3 of the Alaska Stock Assessment Reports. N/A indicates that data are not available.

Fishery name	Years	Data type	Percent observer coverage	Observed mortality	Estimated mortality (CV)	Mean estimated annual mortality
Bering Sea/Aleutian Is. flatfish trawl[a]	2014 2015 2016 2017 2018	obs data	100 100 99 100 100	0 0 1 0 1	0 0 1 (0.05) 0 1 (0.05)	0.4 (CV = 0.03)
Bering Sea/Aleutian Is. Greenland turbot longline[a]	2014 2015 2016 2017 2018	obs data	56 52 60 56 62	0 0 (+1)[b] 0 0 0	0 0 (+1)[c] 0 0 0	0 (+0.2)[d] (CV = N/A)
Minimum total estimated annual mortality						0.6 (CV = 0.03)

[a]Mortality and serious injury in this fishery was assigned to both the Gulf of Alaska, Aleutian Islands, and Bering Sea Transient and Alaska Resident stocks of killer whales, since stock is unknown and the two stocks occur within the area of operation of the fishery.
[b]Total mortality and serious injury observed in 2015: 0 whales in sampled hauls + 1 whale in an unsampled haul.
[c]Total estimate of mortality and serious injury in 2015: 0 whales (extrapolated estimate from 0 whales observed in sampled hauls) + 1 whale (1 whale observed in an unsampled haul).
[d]Mean annual mortality and serious injury for fishery: 0 whales (mean of extrapolated estimates from sampled hauls) + 0.2 whales (mean of number observed in unsampled hauls).

Reports to NMFS Region marine mammal stranding networks of killer whales entangled in fishing gear or with injuries caused by interactions with gear are another source of mortality and serious injury data. A killer whale mortality in commercial California Dungeness crab pot gear in 2015 reported to the NMFS West Coast Region stranding network was genetically identified as a transient ecotype. Because the whale could not be assigned to a specific stock, the mean annual mortality and serious injury rate of 0.2 killer whales in this fishery between 2014 and 2018 was assigned to the Gulf of Alaska, Aleutian Islands, and Bering Sea Transient and West Coast Transient killer whale stocks; it was not assigned to the AT1 Transient killer whale stock because none of the whales in this population are missing (Table 2; Young et al. 2020).

Table 2. Summary of mortality and serious injury of Gulf of Alaska, Aleutian Islands, and Bering Sea Transient killer whales, by year and type, reported to the NMFS West Coast Region marine mammal stranding network between 2014 and 2018 (Young et al. 2020).

Cause of Injury	2014	2015	2016	2017	2018	Mean annual mortality
Entangled in commercial CA Dungeness crab pot gear	0	1[a]	0	0	0	0.2
Total in commercial fisheries						0.2

[a]This whale was genetically identified as a transient ecotype but could not be assigned to a specific stock; therefore, the mortality was assigned to the Gulf of Alaska, Aleutian Islands, and Bering Sea Transient and West Coast Transient killer whale stocks.

A minimum estimate of the mean annual mortality and serious injury rate incidental to U.S. commercial fisheries between 2014 and 2018 is 0.8 Gulf of Alaska, Aleutian Islands, and Bering Sea Transient killer whales, based on observer data (0.6) and stranding data (0.2) (Tables 1 and 2).

Alaska Subsistence/Native Harvest Information

Killer whales are not harvested for subsistence in Alaska.

Other Mortality

Collisions with vessels are an occasional source of mortality or serious injury of killer whales. For example, a killer whale struck the propeller of a vessel in the Bering Sea/Aleutian Islands flatfish trawl fishery in 2016 (Table 1; Young et al. 2020).

Other Issues

Killer whales are known to depredate longline catches in the Bering Sea (Dahlheim 1988; Yano and Dahlheim 1995; Perez 2003, 2006; Sigler et al. 2003) and in the Gulf of Alaska (Sigler et al. 2003, Perez 2006). In addition, there have been many reports of killer whales consuming the processing waste of Bering Sea groundfish trawl fishing vessels (Perez 2006). More recently, Peterson and Hanselman (2017) estimated that killer whales reduce commercial sablefish fishery catch rates by approximately 45% to 70%. However, resident killer whales are most likely to be involved in such fishery interactions since these whales are known to be fish eaters.

STATUS OF STOCK

The Gulf of Alaska, Aleutian Islands, and Bering Sea Transient stock of killer whales is not designated as depleted under the MMPA or listed as threatened or endangered under the Endangered Species Act. Based on currently available data, a minimum estimate of the mean annual mortality and serious injury rate due to U.S. commercial fisheries (0.8 whales) is greater than 10% of the PBR (10% of PBR = 0.6) and, therefore, cannot be considered to be insignificant and approaching a zero mortality and serious injury rate. A minimum estimate of the total annual level of human-caused mortality and serious injury (0.8 whales) is less than the PBR (5.9). Therefore, the Gulf of Alaska, Aleutian Islands, and Bering Sea Transient stock of killer whales is not classified as a strategic stock. Population trends and status of this stock relative to its Optimum Sustainable Population are currently unknown.

There are key uncertainties in the assessment of the Gulf of Alaska, Aleutian Islands, and Bering Sea Transient stock of killer whales. The estimate of abundance, based on the number of uniquely identified individuals, is likely conservative because researchers continue to identify new whales and there has not been a comprehensive survey in recent years to allow an updated line-transect or mark-recapture estimate.

CITATIONS

Baird, R. W., and P. J. Stacey. 1988. Variation in saddle patch pigmentation in populations of killer whales (*Orcinus orca*) from British Columbia, Alaska, and Washington State. Can. J. Zool. 66 (11):2582-2585.

Baird, R. W., P. A. Abrams, and L. M. Dill. 1992. Possible indirect interactions between transient and resident killer whales: implications for the evolution of foraging specializations in the genus *Orcinus*. Oecologia 89:125-132.

Barlow, J. 1995. The abundance of cetaceans in California waters. Part I: Ship surveys in summer and fall of 1991. Fish. Bull., U.S. 93:1-14.

Barlow, J. 1997. Preliminary estimates of cetacean abundance off California, Oregon and Washington based on a 1996 ship survey and comparisons of passing and closing modes. Southwest Fisheries Science Center Administrative Report LJ-97-11, 25 p. Available from SWFSC, NMFS, 8901 La Jolla Shores Drive, La Jolla, CA 92037. 25 p.

Barrett-Lennard, L. G. 2000. Population structure and mating patterns of killer whales (*Orcinus orca*) as revealed by DNA analysis. Ph.D. Dissertation, University of British Columbia, Vancouver, BC, Canada. 97 p.

Bigg, M. A., P. F. Olesiuk, G. M. Ellis, J. K. B. Ford, and K. C. Balcomb III. 1990. Social organization and genealogy of resident killer whales (*Orcinus orca*) in the coastal waters of British Columbia and Washington State, p. 386-406. *In* P. S. Hammond, S. A. Mizroch, and G. P. Donovan (eds.), Individual Recognition of Cetaceans: Use of Photo-identification and Other Techniques to Estimate Population Parameters. Rep. Int. Whal. Comm. Special Issue 12.

Black, N. A., A. Schulman-Janiger, R. L. Ternullo, and M. Guerrero-Ruiz. 1997. Killer whales of California and western Mexico: a catalog of photo-identified individuals. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-SWFSC-247, 174 p.

Braham, H. W., and M. E. Dahlheim. 1982. Killer whales in Alaska documented in the Platforms of Opportunity Program. Rep. Int. Whal. Comm. 32:643-646.

Breiwick, J. M. 2013. North Pacific marine mammal bycatch estimation methodology and results, 2007-2011. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-260, 40 p.

Committee on Taxonomy. 2019. List of marine mammal species and subspecies. Society for Marine Mammalogy. Available online: www.marinemammalscience.org . Accessed December 2020.

Dahlheim, M. E. 1988. Killer whale (*Orcinus orca*) depredation on longline catches of sablefish (*Anoplopoma fimbria*) in Alaskan waters. NWAFC Processed Rep. 88-14, 31 p. Available online: http://www.afsc.noaa.gov/Publications/ProcRpt/PR%2088-14.pdf . Accessed December 2020.

Dahlheim, M. E., and P. A. White. 2010. Ecological aspects of transient killer whales (*Orcinus orca*) as predators in southeastern Alaska. Wildl. Biol. 16: 308-322.

Dahlheim, M. E., D. Ellifrit, and J. Swenson. 1997. Killer Whales of Southeast Alaska: A Catalogue of Photoidentified Individuals. Day Moon Press, Seattle, WA. 82 p. + appendices.

Dahlheim, M. E., A. Schulman-Janiger, N. Black, R. Ternullo, D. Ellifrit, and K. C. Balcomb. 2008. Eastern temperate North Pacific offshore killer whales (*Orcinus orca*): occurrence, movements, and insights into feeding ecology. Mar. Mammal Sci. 24:719-729.

Durban, J., D. Ellifrit, M. Dahlheim, J. Waite, C. Matkin, L. Barrett-Lennard, G. Ellis, R. Pitman, R. LeDuc, and P. R. Wade. 2010. Photographic mark-recapture analysis of clustered mammal-eating killer whales around the Aleutian Islands and Gulf of Alaska. Mar. Biol. 157:1591-1604.

Ford, J. K. B. 2011. Killer whales of the Pacific Northwest coast: from pest to paragon. Whalewatcher 40(1):15-23.

Ford, J. K. B., and G. M. Ellis. 1999. Transients: Mammal-Hunting Killer Whales of British Columbia, Washington, and Southeastern Alaska. University of British Columbia Press, Vancouver, BC. 96 p.

Ford, J. K. B., and H. D. Fisher. 1982. Killer whale (*Orcinus orca*) dialects as an indicator of stocks in British Columbia. Rep. Int. Whal. Comm. 32:671-679.

Ford, J. K. B., G. M. Ellis, and K. C. Balcomb. 1994. Killer Whales: The Natural History and Genealogy of *Orcinus orca* in British Columbia and Washington State. University of British Columbia Press, Vancouver, BC, and University of Washington Press, Seattle. 102 p.

Ford, J. K. B., G. M. Ellis, and K. C. Balcomb. 2000. Killer Whales: The Natural History and Genealogy of *Orcinus orca* in British Columbia and Washington State. Second edition. University of British Columbia Press, Vancouver, BC, Canada. 104 p.

Ford, J. K. B, E. H. Stredulinsky, J. R. Towers, and G. M. Ellis. 2013. Information in support of the identification of critical habitat for transient killer whales (*Orcinus orca*) off the west coast of Canada. DFO Canadian Science Advisory Secretariat Research Document 2012/nnn.

Forney, K. A., and P. R. Wade. 2006. World-wide abundance and density of killer whales, p. 145-162. *In* J. A. Estes, D. P. DeMaster, D. F. Doak, T. M. Williams, and R. L. Brownell, Jr. (eds.), Whales, Whaling, and Ocean Ecosystems. University of California Press.

Forney, K. A., J. Barlow, and J. V. Carretta. 1995. The abundance of cetaceans in California waters. Part II: Aerial surveys in winter and spring of 1991 and 1992. Fish. Bull., U.S. 93:15-26.

George, J. C., and R. Suydam. 1998. Observations of killer whale (*Orcinus orca*) predation in the northeastern Chukchi and western Beaufort seas. Mar. Mammal Sci. 14:330-332. DOI: dx.doi.org/10.1111/j.1748-7692.1998.tb00722.x .

Goley, P. D., and J. M. Straley. 1994. Attack on gray whales (*Eschrichtius robustus*) in Monterey Bay, California, by killer whales (*Orcinus orca*) previously identified in Glacier Bay, Alaska. Can. J. Zool. 72:1528-1530.

Green, G. A., J. J. Brueggeman, R. A. Grotefendt, C. E. Bowlby, M. L. Bonnell, and K. C. Balcomb. 1992. Cetacean distribution and abundance off Oregon and Washington, 1989-1990, p. 1-100. *In* J. J. Brueggeman (ed.), Oregon and Washington marine mammal and seabird surveys. Final Report OCS Study MMS 91-0093.

Hoelzel, A. R., and G. A. Dover. 1991. Genetic differentiation between sympatric killer whale populations. Heredity 66:191-195.

Hoelzel, A. R., M. E. Dahlheim, and S. J. Stern. 1998. Low genetic variation among killer whales (*Orcinus orca*) in the eastern North Pacific, and genetic differentiation between foraging specialists. J. Hered. 89:121-128.

Hoelzel, A. R., A. Natoli, M. Dahlheim, C. Olavarria, R. Baird, and N. Black. 2002. Low worldwide genetic diversity in the killer whale (*Orcinus orca*): implications for demographic history. Proc. R. Soc. Lond. 269:1467-1473.

Leatherwood, J. S., and M. E. Dahlheim. 1978. Worldwide distribution of pilot whales and killer whales. Naval Ocean Systems Center, Tech. Rep. 443:1-39.

Leatherwood, S., C. O. Matkin, J. D. Hall, and G. M. Ellis. 1990. Killer whales, *Orcinus orca*, photo-identified in Prince William Sound, Alaska 1976 to 1987. Can. Field Nat. 104:362-371.

Matkin, C., G. Ellis, E. Saulitis, L. Barrett-Lennard, and D. Matkin. 1999. Killer Whales of Southern Alaska. North Gulf Oceanic Society. 96 p.

Matkin, C. O., L. G. Barrett-Lennard, H. Yurk, D. Ellifrit, and A. W. Trites. 2007. Ecotypic variation and predatory behavior among killer whales (*Orcinus orca*) off the eastern Aleutian Islands, Alaska. Fish. Bull., U.S. 105:74-87.

Matkin, C. O., J. W. Durban, E. L. Saulitis, R. D. Andrews, J. M. Straley, D. R. Matkin, and G. M. Ellis. 2012. Contrasting abundance and residency patterns of two sympatric populations of transient killer whales (*Orcinus orca*) in the northern Gulf of Alaska. Fish. Bull., U.S. 110:143-155.

Matkin, C. O., G. Ellis, E. Saulitis, D. Herman, R. Andrews, and A. Gaylord. 2013. Monitoring, tagging, feeding habits, and restoration of killer whales in Prince William Sound/Kenai Fjords 2010-2012. *Exxon Valdez* Oil Spill Restoration Project Final Report, EVOS Project #10100742, North Gulf Oceanic Society, 3430 Main Street, Suite B1, Homer, Alaska 99603. 62 p.

Mitchell, E. D. 1975. Report on the meeting on small cetaceans, Montreal, April 1-11, 1974. J. Fish. Res. Board Can. 32:914-916.

Morin, P. A., F. I. Archer, A. D. Foote, J. Vilstrup, E. E. Allen, P. Wade, J. Durban, K. M. Parsons, R. Pitman, L. Li, P. Bouffard, S. C. Abel Nielsen, M. Rasmussen, E. Willerslev, M. T. P. Gilbert, and T. Harkins. 2010. Complete mitochondrial genome phylogeographic analysis of killer whales (*Orcinus orca*) indicates multiple species. Genome Res. 20:908-916.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2020.

North Gulf Oceanic Society/Marine Mammal Laboratory (NGOS/MML). 2012. A working catalogue of western transients in Alaska. North Gulf Oceanic Society, Homer, Alaska, and Marine Mammal Laboratory/NOAA, Seattle, WA. Available electronically by request.

Parsons, K. M, J. W. Durban, A. M. Burdin, V. N. Burkanov, R. L. Pitman, J. Barlow, L. G. Barrett-Lennard, R. G. LeDuc, K. M. Robertson, C. O. Matkin, and P. R. Wade. 2013. Geographic patterns of genetic differentiation among killer whales in the northern North Pacific. J. Hered. 104:737-754.

Perez, M. A. 2003. Compilation of marine mammal-fisheries interaction data from the domestic and joint venture groundfish fisheries in the U.S. EEZ of the North Pacific, 1989-2001. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-138, 145 p.

Perez, M. A. 2006. Analysis of marine mammal bycatch data from the trawl, longline, and pot groundfish fisheries of Alaska, 1998-2004, defined by geographic area, gear type, and target groundfish catch species. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-167, 194 p.

Peterson, M. J., and D. Hanselman. 2017. Sablefish mortality associated with whale depredation in Alaska. ICES J. Mar. Sci. 74(5):1382-1394. DOI: dx.doi.org/10.1093/icesjms/fsw239 .

Riesch, R., L. G. Barrett-Lennard, G. M. Ellis, J. K. B. Ford, and V. B. Deecke. 2012. Cultural traditions and the evolution of reproductive isolation: ecological speciation in killer whales? Biol. J. Linn. Soc. 106:1-17.

Saulitis, E., C. O. Matkin, and F. H. Fay. 2005. Vocal repertoire and acoustic behavior of the isolated AT1 killer whale subpopulation in southern Alaska. Can. J. Zool. 83:1015-1029.

Sigler, M. F., C. R. Lunsford, J. T. Fujioka, and S. A. Lowe. 2003. Alaska sablefish assessment for 2004. *In* Stock assessment and fishery evaluation report for the groundfish fisheries of the Bering Sea/Aleutian Islands regions. North Pacific Fishery Management Council, Anchorage, AK, Section 3:223-292.

Towers, J. R., G. J. Sutton, T. J. H. Shaw, M. Malleson, D. Matkin, B. Gisborne, J. Forde, D. Ellifrit, G. M. Ellis, J. K. B. Ford, and T. Doniol-Valcroze. 2019. Photo-identification catalogue, population status, and distribution of Bigg's killer whales known from coastal waters of British Columbia, Canada. Can. Tech. Rep. Fish. Aquat. Sci. 3311:vi + 299 p.

Yano, K., and M. E. Dahlheim. 1995. Killer whale, *Orcinus orca*, depredation on longline catches of bottomfish in the southeastern Bering Sea and adjacent waters. Fish. Bull., U.S. 93:355-372.

Young, N. C., B. J. Delean, V. T. Helker, J. C. Freed, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2014-2018. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-413, 142 p.

Zerbini, A. N., J. M. Waite, J. Durban, R. LeDuc, M. E. Dahlheim, and P. R. Wade. 2007. Estimating abundance of killer whales in the nearshore waters of the Gulf of Alaska and Aleutian Islands using line-transect sampling. Mar. Biol. 150(5):1033-1045.

Revised 12/30/2020

KILLER WHALE (*Orcinus orca*): AT1 Transient Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Killer whales have been observed in all oceans and seas of the world (Leatherwood and Dahlheim 1978). Although reported from tropical and offshore waters, killer whales occur at higher densities in colder and more productive waters of both hemispheres, with the greatest densities found at high latitudes (Mitchell 1975, Leatherwood and Dahlheim 1978, Forney and Wade 2006). Killer whales are found throughout the North Pacific Ocean. Along the west coast of North America, seasonal and year-round occurrence of killer whales has been noted along the entire Alaska coast (Braham and Dahlheim 1982), in British Columbia and Washington inland waterways (Bigg et al. 1990), and along the outer coasts of Washington, Oregon, and California (Green et al. 1992; Barlow 1995, 1997; Forney et al. 1995). Killer whales from these areas have been labeled as "resident," "transient," and "offshore" type killer whales (Bigg et al. 1990, Ford et al. 2000, Dahlheim et al. 2008) 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, 2002; Barrett-Lennard 2000; Dahlheim et al. 2008). Through examination of photographs of recognizable individuals and pods, movements of whales between geographical areas have been documented. For example, whales identified in Prince William Sound have been observed near Kodiak Island (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). Movements of killer whales between the waters of Southeast Alaska and central California have also been documented (Goley and Straley 1994, Black et al. 1997, Dahlheim and White 2010).

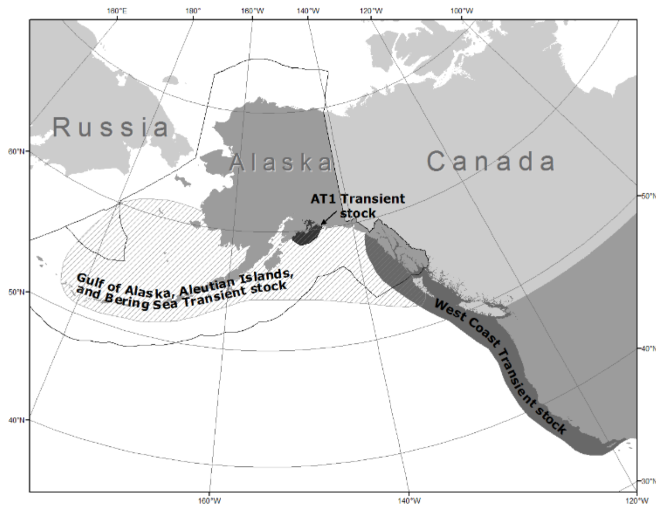

Figure 1. Approximate distribution of transient killer whales in the eastern North Pacific (shaded areas). The distribution of resident and transient killer whale stocks in the eastern North Pacific largely overlap (see text). The U.S. Exclusive Economic Zone is delineated by a black line.

Several studies provide evidence that the resident, offshore, and transient ecotypes are genetically distinct in both mtDNA and nuclear DNA (Hoelzel and Dover 1991; Hoelzel et al. 1998, 2002; Barrett-Lennard 2000). Genetic differences have also been found between populations within the transient and resident ecotypes (Hoelzel et al. 1998, 2002; Barrett-Lennard 2000). A global genetic study of killer whales using the entire mitochondrial genome found that some killer whale ecotypes represent deeply divergent evolutionary lineages and warrant elevation to species or subspecies status (Morin et al. 2010). In particular, estimates from mitogenome sequence data indicate that transient killer whales diverged from all other killer whale lineages approximately 700,000 years ago. In light of these differences, the Society for Marine Mammalogy's Committee on Taxonomy currently recognizes the resident and transient North Pacific ecotypes as un-named *Orcinus orca* subspecies (Committee on Taxonomy 2019). In recognition of its status as an un-named subspecies or species, some researchers now refer to transient-type killer whales as Bigg's killer whales (e.g., Ford 2011, Riesch et al. 2012), in tribute to the late Dr. Michael Bigg.

The first studies of transient killer whales in Alaska were conducted in Southeast Alaska and in the Gulf of Alaska (from Prince William Sound, through the Kenai Fjords, and around Kodiak Island). In the Gulf of Alaska, Matkin et al. (1999) described two genetically distinct populations of transients which were never found in association with one another, the so-called "Gulf of Alaska" transients and "AT1" transients. In the past, neither of these populations were known to associate with the population of transient killer whales that ranged from California to Southeast Alaska, which are described as the West Coast Transient stock. Gulf of Alaska transients are documented throughout the Gulf of Alaska, including occasional sightings in Prince William Sound. AT1 transients have been seen only in Prince William Sound and in the Kenai Fjords region, and are therefore partially sympatric with Gulf of Alaska transients. In addition, 14 out of 217 transients on the outer coast of Southeast Alaska and British Columbia were identified as Gulf of Alaska transients and in one encounter they were observed mixing with West Coast transients (Matkin et al. 2012, Ford et al. 2013). Transients within the Gulf of Alaska population have been found to have two mtDNA haplotypes, neither of which is found in the West Coast or AT1 populations. Members of the AT1 population share a single mtDNA haplotype. Transient killer whales from the West Coast population have been found to share a single mtDNA haplotype that is not found in the other populations. Additionally, all three populations have been found to have significant differences in nuclear (microsatellite) DNA (Barrett-Lennard 2000). Acoustic differences have been found as well; Saulitis et al. (2005) described acoustic differences between Gulf of Alaska transients and AT1 transients. For these reasons, the Gulf of Alaska transients are considered part of a population that is discrete from the AT1 population, and both of these populations are considered discrete from the West Coast transients.

Transient-type killer whales from the Aleutian Islands and Bering Sea are currently considered to be part of a single population that includes Gulf of Alaska transients; however, recent genetic analyses suggest substructure within the region. Biopsy samples from the eastern Aleutians and the south side of the west end of the Alaska Peninsula have produced the same haplotypes as killer whales in the northern Gulf of Alaska; however, nuclear DNA analysis strongly suggests they belong to a separate population (Parsons et al. 2013). The geographic distribution of mtDNA haplotypes revealed samples from the central Aleutian Islands and Bering Sea with haplotypes not found in Gulf of Alaska transients, suggesting additional population structure in western Alaska. Killer whales observed in the northern Bering Sea and north and east to the western Beaufort Sea have characteristics of transient-type whales, but little is known about these whales (Braham and Dahlheim 1982, George and Suydam 1998). AT1 haplotype whales are also present west of the Aleutian Islands and into the Bering Sea; however, nuclear DNA analysis indicates these animals are not part of the AT1 transient population in the Gulf of Alaska (Parsons et al. 2013).

In summary, within the transient ecotype, association data (Ford et al. 1994, Ford and Ellis 1999, Matkin et al. 1999), acoustic data (Ford and Ellis 1999, Saulitis et al. 2005), and genetic data (Hoelzel et al. 1998, 2002; Barrett-Lennard 2000) confirm that at least three communities of transient whales exist and represent three discrete populations: 1) Gulf of Alaska, Aleutian Islands, and Bering Sea transients, 2) AT1 transients, and 3) West Coast transients.

Based on data regarding association patterns, acoustics, movements, and genetic differences, eight killer whale stocks are now recognized within the Pacific U.S. Exclusive Economic Zone: 1) the Alaska Resident stock - occurring from Southeast Alaska to the Aleutian Islands and Bering Sea, 2) the Northern Resident stock - occurring from Washington State through part of Southeast Alaska, 3) the Southern Resident stock - occurring mainly within the inland waters of Washington State and southern British Columbia, but also in coastal waters from Southeast Alaska through California, 4) the Gulf of Alaska, Aleutian Islands, and Bering Sea Transient stock - occurring mainly from Prince William Sound through the Aleutian Islands and Bering Sea, 5) the AT1 Transient stock - occurring in Alaska from Prince William Sound through the Kenai Fjords (Fig. 1), 6) the West Coast Transient stock - occurring from California through Southeast Alaska, 7) the Offshore stock - occurring from California through Alaska, and 8) the Hawaiian stock. Transient killer whales in Canadian waters are considered part of the West Coast Transient stock. The Hawaiian and Offshore stocks are reported in the Stock Assessment Reports for the U.S. Pacific Region.

AT1 killer whales were first identified as a separate, cohesive group in 1984, when 22 transient-type whales were documented in Prince William Sound (Leatherwood et al. 1984, Heise et al. 1991), although individual whales from the group had been photographed as early as 1978 (von Ziegesar et al. 1986). Once the North Gulf Oceanic Society (NGOS) began consistent annual research effort in Prince William Sound, AT1 killer whales were resighted frequently. In fact, AT1 killer whales were found to be some of the most frequently sighted killer whales in Prince William Sound (Matkin et al. 1993, 1994, 1999). Gulf of Alaska transients are seen less frequently in Prince William Sound, with periods of several years or more between resightings.

AT1 killer whales have never been seen in association with sympatric resident killer whale pods or with Gulf of Alaska transients (Matkin et al. 1999, 2012) and appear to have a more limited range than other transients. Their approximately 200-mile known range includes only Prince William Sound and Kenai Fjords and adjacent offshore waters (Matkin et al. 1999, 2012).

POPULATION SIZE

Using photographic-identification, all 22 individuals in the AT1 Transient population were censused for the first time in 1984 (Leatherwood et al. 1984). All 22 AT1 killer whales were seen annually or biannually from 1984 to 1988 (Matkin et al. 1999, 2003). The *Exxon Valdez* oil spill occurred in spring of 1989. Nine individuals from the AT1 group have been missing since 1990 (last seen in 1989), and two have been missing since 1992 (last seen in 1990 and 1991). Three of the missing AT1 killer whales (AT5, AT7, and AT8) were seen near the leaking *Exxon Valdez* shortly after the spill (Matkin et al. 1993, 1994, 2008). Two whales were found dead, stranded in 1989 and 1990, both genetically assigned to the AT1 population and one visually recognized as AT19, one of the missing nine whales (Matkin et al. 1994, 2008; Heise et al. 2003). The second unidentified whale was most likely one of the other missing AT1 whales. Additional mortalities of four older males include whales AT1 found stranded in 2000, AT13 and AT17 missing in 2002 (one of which was thought to be the carcass from the AT1 population that was found in 2002), and AT14 missing in 2003. A stranded whale found in 2003, genetically assigned to the AT1 population, was probably AT14 but could also have been AT13 (Matkin et al. 2008). No births have occurred in this population since 1984 and none of the missing whales have been seen since 2003 and are presumed dead. There is an extremely small probability (0.4%) that AT1 killer whales that are missing for 3 years or more are still alive (Matkin et al. 2008). No AT1 killer whale missing for at least 4 years has ever been resighted, and all 15 missing whales are presumed dead (Matkin et al. 2008). In 2019, photographs of the seven remaining AT1 killer whales were confirmed by researchers from the NGOS (http://www.whalesalaska.org, accessed December 2020); birth year is estimated for whales born before 1983, as described in Matkin et al. (1999): AT2 (female, born $\leq$1969), AT3 (male, born 1984), AT4 (female, born $\leq$1974), AT6 (male, born 1976), AT9 (female, born $\leq$1965), AT10 (male, born 1980), and AT18 (female, born $\leq$1974). Therefore, the population estimate as of the summer of 2019 remains at seven whales (NGOS; C. Matkin, NGOS, pers. comm., 17 October 2019). There has been no recruitment in this population since 1984 (Matkin et al. 2012).

Minimum Population Estimate

The abundance estimate of killer whales is a direct count of individually identifiable animals. Only 11 whales were seen between 1990 and 1999. Since then, four of those whales have not been seen for four or more consecutive years, so the minimum population estimate (N_{MIN}) is seven whales (Matkin et al. 2008; NGOS; C. Matkin, NGOS, pers. comm., 17 October 2019). Fourteen years of annual effort have failed to discover any whales that had not been seen previously, so there is no reason to believe there are additional whales in the population. Therefore, this N_{MIN} is the total population size.

Current Population Trend

The population counts have declined from a level of 22 whales in 1989 to the 7 whales that have been resighted since 2003, a decline of 68%. Most of the mortality apparently occurred in 1989 and 1990.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate (R_{MAX}) is not available for the AT1 Transient stock of killer whales. Between 2012 and 2018, Towers et al. (2019) observed a mean annual growth rate of 4.1% for a population subset of transient killer whales in Canadian coastal waters, which was higher than the mean annual growth rate of 2.7% documented by Ford et al. (2013) between 2006 and 2011 for a subpopulation of inner-coast transient killer whales that contained most of the same individuals. The current net productivity rate for the AT1 Transient stock of killer whales is 0, given that there has been no recruitment into the stock since 1984. Until additional stock-specific data become available, the default cetacean maximum theoretical net productivity rate of 4% will be used for this stock (NMFS 2016).

POTENTIAL BIOLOGICAL REMOVAL

Potential biological removal (PBR) is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: PBR = $N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 0.1, as the stock is considered depleted under the Marine Mammal Protection Act (MMPA) and there has been no recruitment into the stock since 1984. Thus, for the AT1 Transient killer whale stock, PBR is 0.01 whales ($7 \times 0.02 \times 0.1$).

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2014 and 2018 is listed, by marine mammal stock, in Young et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports. No human-caused mortality or serious injury of AT1 Transient killer whales was reported between 2014 and 2018. Potential threats most likely to result in direct human-caused mortality or serious injury of this stock include ship strikes and oil spills (most of the mortality in this stock occurred in 1989 and 1990, following the *Exxon Valdez* oil spill).

Fisheries Information

Information for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is available in Appendix 3 of the Alaska Stock Assessment Reports (observer coverage) and in the NMFS List of Fisheries (LOF) and the fact sheets linked to fishery names in the LOF (observer coverage and reported incidental takes of marine mammals: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-list-fisheries, accessed December 2020).

The known range of the AT1 Transient stock is limited to waters of Prince William Sound and Kenai Fjords. There are no federally-managed commercial fisheries in this area. Incidental mortality or serious injury of AT1 killer whales has not been reported in state-managed commercial fisheries which operate within the range of this stock, such as the Prince William Sound salmon set and drift gillnet fisheries and various herring fisheries, or in several subsistence fisheries (salmon, halibut, non-salmon finfish, and shellfish) which also occur within this area; however, the state-managed fisheries are not observed or have not been observed in a long time. Transient killer whales have entangled in pot fishery gear in other areas (Young et al. 2020) and entanglement in this type of gear may be a risk for the AT1 Transient stock of killer whales.

Alaska Native Subsistence/Harvest Information

Killer whales are not harvested for subsistence in Alaska.

Other Mortality

Collisions with vessels are an occasional source of mortality or serious injury of killer whales. For example, a killer whale struck the propeller of a vessel in the Bering Sea/Aleutian Islands flatfish trawl fishery in 2016 (Young et al. 2020); however, this mortality did not involve a whale from the AT1 Transient stock. There has been no known mortality or serious injury of AT1 killer whales due to vessel collisions. Most of the mortality occurred from 1989 to 1990 following the *Exxon Valdez* oil spill.

STATUS OF STOCK

The AT1 Transient stock of killer whales is below its Optimum Sustainable Population (OSP) and designated as depleted under the MMPA (69 FR 31321, 3 June 2004); therefore, it is classified as a strategic stock. The AT1 Transient stock is not listed as threatened or endangered under the Endangered Species Act. Based on currently available data, the minimum estimated mean annual mortality and serious injury rate due to U.S. commercial fisheries (0) does not exceed 10% of the PBR (10% of PBR = 0.001) and, therefore, can be considered insignificant and approaching a zero mortality and serious injury rate. At least 11 animals were alive in 1998, but it appears that only 7 individuals remain alive. The AT1 killer whale group has been reduced to 32% (7/22) of its 1984 level. Since no births have occurred in the past 30 years, it is unlikely that this stock will recover.

There are few uncertainties in the assessment of the AT1 Transient stock of killer whales. Individual whales can be counted annually and the stock has been declining slowly since a dramatic reduction in the stock occurred immediately after the *Exxon Valdez* oil spill. 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 and that some surplus growth could be removed while still allowing recovery. However, the AT1 Transient killer whale population is at a very small population size, and small populations can have different dynamics than larger populations from Allee effects and stochastic dynamics. Although there is currently no known direct human-caused mortality or serious injury, given the small number of animals in the population, any human-caused mortality or serious injury is likely to have a serious population-level impact.

CITATIONS

Baird, R. W., and P. J. Stacey. 1988. Variation in saddle patch pigmentation in populations of killer whales (*Orcinus orca*) from British Columbia, Alaska, and Washington State. Can. J. Zool. 66(11):2582-2585.

Baird, R. W., P. A. Abrams, and L. M. Dill. 1992. Possible indirect interactions between transient and resident killer whales: implications for the evolution of foraging specializations in the genus *Orcinus*. Oecologia 89:125-132.

Barlow, J. 1995. The abundance of cetaceans in California waters. Part I: Ship surveys in summer and fall of 1991. Fish. Bull., U.S. 93:1-14.

Barlow, J. 1997. Preliminary estimates of cetacean abundance off California, Oregon and Washington based on a 1996 ship survey and comparisons of passing and closing modes. Southwest Fisheries Science Center Administrative Report LJ-97-11, 25 p. Available from SWFSC, NMFS, 8901 La Jolla Shores Drive, La Jolla, CA 92037.

Barrett-Lennard, L. G. 2000. Population structure and mating patterns of killer whales as revealed by DNA analysis. Ph.D. Dissertation, University of British Columbia, Vancouver, BC, Canada. 97 p.

Bigg, M. A., P. F. Olesiuk, G. M. Ellis, J. K. B. Ford, and K. C. Balcomb III. 1990. Social organization and genealogy of resident killer whales (*Orcinus orca*) in the coastal waters of British Columbia and Washington State, p. 386-406. *In* P. S. Hammond, S. A. Mizroch, and G. P. Donovan (eds.), Individual Recognition of Cetaceans: Use of Photo-identification and Other Techniques to Estimate Population Parameters. Rep. Int. Whal. Comm. Special Issue 12.

Black, N. A., A. Schulman-Janiger, R. L. Ternullo, and M. Guerrero-Ruiz. 1997. Killer whales of California and western Mexico: a catalog of photo-identified individuals. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-SWFSC-247, 174 p.

Braham, H. W., and M. E. Dahlheim. 1982. Killer whales in Alaska documented in the Platforms of Opportunity Program. Rep. Int. Whal. Comm. 32:643-646.

Committee on Taxonomy. 2019. List of marine mammal species and subspecies. Society for Marine Mammalogy. Available online: www.marinemammalscience.org . Accessed December 2020.

Dahlheim, M. E., and P. A. White. 2010. Ecological aspects of transient killer whales (*Orcinus orca*) as predators in southeastern Alaska. Wildl. Biol. 16:308-322.

Dahlheim, M. E., D. Ellifrit, and J. Swenson. 1997. Killer Whales of Southeast Alaska: A Catalogue of Photoidentified Individuals. Day Moon Press, Seattle, WA. 82 p. + appendices.

Dahlheim, M. E., A. Schulman-Janiger, N. Black, R. Ternullo, D. Ellifrit, and K. C. Balcomb. 2008. Eastern temperate North Pacific offshore killer whales (*Orcinus orca*): occurrence, movements, and insights into feeding ecology. Mar. Mammal Sci. 24(3):719-729.

Ford, J. K. B. 2011. Killer whales of the Pacific Northwest coast: from pest to paragon. Whale Watcher 40(1):15-23.

Ford, J. K. B., and G. M. Ellis. 1999. Transients: Mammal-Hunting Killer Whales of British Columbia, Washington, and Southeastern Alaska. University of British Columbia Press, Vancouver, BC. 96 p.

Ford, J. K. B., and H. D. Fisher. 1982. Killer whale (*Orcinus orca*) dialects as an indicator of stocks in British Columbia. Rep. Int. Whal. Comm. 32:671-679.

Ford, J. K. B., G. M. Ellis, and K. C. Balcomb. 1994. Killer Whales: The Natural History and Genealogy of *Orcinus orca* in British Columbia and Washington State. University of British Columbia Press, Vancouver, BC, and University of Washington Press, Seattle. 102 p.

Ford, J. K. B., G. M. Ellis, and K. C. Balcomb. 2000. Killer Whales: The Natural History and Genealogy of *Orcinus orca* in British Columbia and Washington State. Second edition. University of British Columbia Press, Vancouver, BC, Canada. 104 p.

Ford, J. K. B., E. H. Stredulinsky, J. R. Towers, and G. M. Ellis. 2013. Information in support of the identification of critical habitat for transient killer whales (*Orcinus orca*) off the west coast of Canada. DFO Canadian Science Advisory Secretariat Research Document 2012/nnn.

Forney, K. A., and P. R. Wade. 2006. World-wide abundance and density of killer whales, p. 145-162. *In* J. A. Estes, D. P. DeMaster, D. F. Doak, T. M. Williams, and R. L. Brownell, Jr. (eds.), Whales, Whaling, and Ocean Ecosystems. University of California Press.

Forney, K. A., J. Barlow, and J. V. Carretta. 1995. The abundance of cetaceans in California waters. Part II: Aerial surveys in winter and spring of 1991 and 1992. Fish. Bull., U.S. 93:15-26.

George, J. C., and R. Suydam. 1998. Observations of killer whale (*Orcinus orca*) predation in the northeastern Chukchi and western Beaufort seas. Mar. Mammal Sci. 14:330-332. DOI: dx.doi.org/10.1111/j.1748-7692.1998.tb00722.x .

Goley, P. D., and J. M. Straley. 1994. Attack on gray whales (*Eschrichtius robustus*) in Monterey Bay, California, by killer whales (*Orcinus orca*) previously identified in Glacier Bay, Alaska. Can. J. Zool. 72:1528-1530.

Green, G. A., J. J. Brueggeman, R. A. Grotefendt, C. E. Bowlby, M. L. Bonnell, and K. C. Balcomb. 1992. Cetacean distribution and abundance off Oregon and Washington, 1989-1990, p. 1-100. *In* J. J. Brueggeman (ed.), Oregon and Washington marine mammal and seabird surveys. Final Report OCS Study MMS 91-0093.

Heise, K., G. Ellis, and C. Matkin. 1991. A Catalogue of Prince William Sound Killer Whales. North Gulf Oceanic Society, Homer, AK. 51 p.

Heise, K., L. G. Barrett-Lennard, E. Saulitis, C. Matkin, and D. Bain. 2003. Examining the evidence for killer whale predation on Steller sea lions in British Columbia and Alaska. Aquat. Mamm. 29(3):325-334.

Hoelzel, A. R., and G. A. Dover. 1991. Genetic differentiation between sympatric killer whale populations. Heredity 66:191-195.

Hoelzel, A. R., M. E. Dahlheim, and S. J. Stern. 1998. Low genetic variation among killer whales (*Orcinus orca*) in the eastern North Pacific, and genetic differentiation between foraging specialists. J. Hered. 89:121-128.

Hoelzel, A. R., A. Natoli, M. Dahlheim, C. Olavarria, R. Baird, and N. Black. 2002. Low worldwide genetic diversity in the killer whale (*Orcinus orca*): implications for demographic history. Proc. R. Soc. Lond. 269:1467-1473.

Leatherwood, J. S., and M. E. Dahlheim. 1978. Worldwide distribution of pilot whales and killer whales. Naval Ocean Systems Center, Tech. Rep. 443:1-39.

Leatherwood, S., A. E. Bowles, E. Krygier, J. D. Hall, and S. Ingell. 1984. Killer whales (*Orcinus orca*) in Southeast Alaska, Prince William Sound, and Shelikof Strait: a review of available information. Rep. Int. Whaling Comm. 34:521-530.

Leatherwood, S., C. O. Matkin, J. D. Hall, and G. M. Ellis. 1990. Killer whales, *Orcinus orca*, photo-identified in Prince William Sound, Alaska 1976 to 1987. Can. Field Nat. 104:362-371.

Matkin, C. O., M. E. Dahlheim, G. Ellis, and E. Saulitis. 1993. Vital rates and pod structure of resident killer whales following the *Exxon Valdez* oil spill, p. 303-307. *In Exxon Valdez* Oil Spill Trustee Council, *Exxon Valdez* oil spill symposium abstract book, February 2-5, 1993, Anchorage, Alaska.

Matkin, C. O., G. M. Ellis, M. E. Dahlheim, and J. Zeh. 1994. Status of killer whales in Prince William Sound, 1985-1992, p. 141-162. *In* T. R. Loughlin (ed.), Marine Mammals and the *Exxon Valdez*. Academic Press, San Diego, CA.

Matkin, C., G. Ellis, E. Saulitis, L. Barrett-Lennard, and D. Matkin. 1999. Killer Whales of Southern Alaska. North Gulf Oceanic Society, Homer, AK. 96 p.

Matkin, C. O., G. M. Ellis, L. G. Barrett-Lennard, H. Yurk, E. L. Saulitis, D. Scheel, P. Olesiuk, and G. Ylitalo. 2003. Photographic and acoustic monitoring of killer whales in Prince William and Kenai Fjords, *Exxon Valdez* Oil Spill Restoration Project Final Report, Restoration Project 03012, North Gulf Oceanic Society, Homer, AK. 118 p.

Matkin, C. O., E. L. Saulitis, G. M. Ellis, P. Olesiuk, and S. D. Rice. 2008. Ongoing population-level impacts on killer whales *Orcinus orca* following the 'Exxon Valdez' oil spill in Prince William Sound, Alaska. Mar. Ecol. Prog. Ser. 356:269-281.

Matkin, C. O., J. W. Durban, E. L. Saulitis, R. D. Andrews, J. M. Straley, D. R. Matkin, and G. M. Ellis. 2012. Contrasting abundance and residency patterns of two sympatric populations of transient killer whales (*Orcinus orca*) in the northern Gulf of Alaska. Fish. Bull., U.S. 110:143-155.

Mitchell, E. D. 1975. Report on the meeting on small cetaceans, Montreal, April 1-11, 1974. J. Fish. Res. Board Can. 32:914-916.

Morin, P. A., F. I. Archer, A. D. Foote, J. Vilstrup, E. E. Allen, P. Wade, J. Durban, K. Parsons, R. Pitman, L. Li, P. Bouffard, S. C. Abel Nielsen, M. Rasmussen, E. Willerslev, M. T. P. Gilbert, and T. Harkins. 2010. Complete mitochondrial genome phylogeographic analysis of killer whales (*Orcinus orca*) indicates multiple species. Genome Res. 20:908-916.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2020.

Parsons, K. M., J. W. Durban, A. M. Burdin, V. N. Burkanov, R. L. Pitman, J. Barlow, L. G. Barrett-Lennard, R. G. LeDuc, K. M. Robertson, C. O. Matkin, and P. R. Wade. 2013. Geographic patterns of genetic differentiation among killer whales in the northern North Pacific. J. Hered. 104:737-754.

Riesch, R., L. G. Barrett-Lennard, G. M. Ellis, J. K. B. Ford, and V. B. Deecke. 2012. Cultural traditions and the evolution of reproductive isolation: ecological speciation in killer whales? Biol. J. Linn. Soc. 106:1-17.

Saulitis, E., C. O. Matkin, and F. H. Fay. 2005. Vocal repertoire and acoustic behavior of the isolated AT1 killer whale subpopulation in southern Alaska. Can. J. Zool. 83:1015-1029.

Towers, J. R., G. J. Sutton, T. J. H. Shaw, M. Malleson, D. Matkin, B. Gisborne, J. Forde, D. Ellifrit, G. M. Ellis, J. K. B. Ford, and T. Doniol-Valcroze. 2019. Photo-identification catalogue, population status, and distribution of Bigg's killer whales known from coastal waters of British Columbia, Canada. Can. Tech. Rep. Fish. Aquat. Sci. 3311:vi + 299 p.

von Ziegesar, O., G. M. Ellis, C. O. Matkin, and B. Goodwin. 1986. Repeated sightings of identifiable killer whales (*Orcinus orca*) in Prince William Sound, Alaska, 1977-1983. Cetus 6(2):9-13.

Wade, P. R. 1998. Calculating limits to the allowable human-caused mortality of cetaceans and pinnipeds. Mar. Mammal Sci. 14:1-37. DOI: dx.doi.org/10.1111/j.1748-7692.1998.tb00688.x .

Young, N. C., B. J. Delean, V. T. Helker, J. C. Freed, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2014-2018. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-413, 142 p.

KILLER WHALE (*Orcinus orca*): West Coast Transient Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Killer whales have been observed in all oceans and seas of the world (Leatherwood and Dahlheim 1978). Although reported from tropical and offshore waters, killer whales occur at higher densities in colder and more productive waters of both hemispheres, with the greatest densities found at high latitudes (Mitchell 1975, Leatherwood and Dahlheim 1978, Forney and Wade 2006). Killer whales are found throughout the North Pacific Ocean. Along the west coast of North America, seasonal and year-round occurrence of killer whales has been noted along the entire Alaska coast (Braham and Dahlheim 1982), in British Columbia and Washington inland waterways (Bigg et al. 1990), and along the outer coasts of Washington, Oregon, and California (Green et al. 1992; Barlow 1995, 1997; Forney et al. 1995). Killer whales from these areas have been labeled as "resident," "transient," and "offshore" type killer whales (Bigg et al. 1990, Ford et al. 2000, Dahlheim et al. 2008) 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, 2002; Barrett-Lennard 2000; Dahlheim et al. 2008). Through examination of photographs of recognizable individuals and pods, movements of whales between geographical areas have been documented. For example, whales identified in Prince William Sound have been observed near Kodiak Island (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). Movements of killer whales between the waters of Southeast Alaska and central California have also been documented (Goley and Straley 1994, Black et al. 1997, Dahlheim and White 2010).

Figure 1. Approximate distribution of transient killer whales in the eastern North Pacific (shaded areas). The distribution of resident and transient killer whale stocks in the eastern North Pacific largely overlap (see text). The U.S. Exclusive Economic Zone is delineated by a black line.

Several studies provide evidence that the resident, offshore, and transient ecotypes are genetically distinct in both mtDNA and nuclear DNA (Hoelzel and Dover 1991; Hoelzel et al. 1998, 2002; Barrett-Lennard 2000). Genetic differences have also been found between populations within the transient and resident ecotypes (Hoelzel et al. 1998, 2002; Barrett-Lennard 2000). A global genetic study of killer whales using the entire mitochondrial genome found that some killer whale ecotypes represent deeply divergent evolutionary lineages and warrant elevation to species or subspecies status (Morin et al. 2010). In particular, estimates from mitogenome sequence data indicate that transient killer whales diverged from all other killer whale lineages approximately 700,000 years ago. In light of these differences, the Society for Marine Mammalogy's Committee on Taxonomy currently recognizes the resident and transient North Pacific ecotypes as un-named *Orcinus orca* subspecies (Committee on Taxonomy 2019). In recognition of its status as an un-named subspecies or species, some researchers now refer to transient-type killer whales as Bigg's killer whales (e.g., Ford 2011, Riesch et al. 2012), in tribute to the late Dr. Michael Bigg.

The first studies of transient killer whales in Alaska were conducted in Southeast Alaska and in the Gulf of Alaska (from Prince William Sound, through the Kenai Fjords, and around Kodiak Island). In the Gulf of Alaska, Matkin et al. (1999) described two genetically distinct populations of transients which were never found in association with one another, the so-called "Gulf of Alaska" transients and "AT1" transients. In the past, neither of these populations were known to associate with the population of transient killer whales that ranged from California to Southeast Alaska, which are described as the West Coast Transient stock. Gulf of Alaska transients are documented throughout the Gulf of Alaska, including occasional sightings in Prince William Sound. AT1 transients have been seen only in Prince William Sound and in the Kenai Fjords region, and are therefore partially sympatric with Gulf of Alaska transients. In addition, 14 out of 217 transients on the outer coast of Southeast Alaska and British Columbia were identified as Gulf of Alaska transients and in one encounter they were observed mixing with West Coast transients (Matkin et al. 2012, Ford et al. 2013). Transients within the Gulf of Alaska population have been found to have two mtDNA haplotypes, neither of which is found in the West Coast or AT1 populations. Members of the AT1 population share a single mtDNA haplotype. Transient killer whales from the West Coast population have been found to share a single mtDNA haplotype that is not found in the other populations. Additionally, all three populations have been found to have significant differences in nuclear (microsatellite) DNA (Barrett-Lennard 2000). Acoustic differences have been found as well; Saulitis et al. (2005) described acoustic differences between Gulf of Alaska transients and AT1 transients. For these reasons, the Gulf of Alaska transients are considered part of a population that is discrete from the AT1 population, and both of these populations are considered discrete from the West Coast transients.

Transient-type killer whales from the Aleutian Islands and Bering Sea are currently considered to be part of a single population that includes Gulf of Alaska transients; however, recent genetic analyses suggest substructure within the region. Biopsy samples from the eastern Aleutians and the south side of the west end of the Alaska Peninsula have produced the same haplotypes as killer whales in the northern Gulf of Alaska; however, nuclear DNA analysis strongly suggests they belong to a separate population (Parsons et al. 2013). The geographic distribution of mtDNA haplotypes revealed samples from the central Aleutian Islands and Bering Sea with haplotypes not found in Gulf of Alaska transients, suggesting additional population structure in western Alaska. Killer whales observed in the northern Bering Sea and north and east to the western Beaufort Sea have characteristics of transient-type whales, but little is known about these whales (Braham and Dahlheim 1982, George and Suydam 1998). AT1 haplotype whales are also present west of the Aleutian Islands and into the Bering Sea; however, nuclear DNA analysis indicates these animals are not part of the AT1 transient population in the Gulf of Alaska (Parsons et al. 2013).

In summary, within the transient ecotype, association data (Ford et al. 1994, Ford and Ellis 1999, Matkin et al. 1999), acoustic data (Ford and Ellis 1999, Saulitis et al. 2005), and genetic data (Hoelzel et al. 1998, 2002; Barrett-Lennard 2000) confirm that at least three communities of transient whales exist and represent three discrete populations: 1) Gulf of Alaska, Aleutian Islands, and Bering Sea transients, 2) AT1 transients, and 3) West Coast transients.

Most of the transient killer whales photographed in the inland waters of Southeast Alaska share the West Coast Transient haplotype and have been seen in association with British Columbia/Washington State transients. Transients most often seen off California also share the West Coast Transient (WCT) haplotype and have been observed in association with transients in Washington and British Columbia. The West Coast Transient stock is therefore considered to include transient killer whales from California through Southeast Alaska. However, it should be noted that Fisheries and Oceans Canada no longer includes whales from California in their assessment of the “West Coast Transient (WCT) Population” (Fisheries and Oceans Canada 2007). They noted that 100 or so transient killer whales identified off the central coast of California (Black et al. 1997) were in the past considered to be an extension of this population because of acoustical similarities and occasional mixing with WCT individuals in BC waters (Ford and Ellis 1999), but that a recent reassessment indicated that the available evidence was insufficient to warrant inclusion of those whales in the WCT population (Fisheries and Oceans Canada 2010). Canadian researchers have now identified 46 individual whales in British Columbia that are known from California (J. Ford, pers. comm., Department of Fisheries and Oceans, British Columbia, Canada, 30 January 2013). They also noted that the Gulf of Alaska transients are seen occasionally within the range of WCTs (in Southeast Alaska and off British Columbia) but have only been observed to travel in association with WCTs on one occasion (Fisheries and Oceans Canada 2007, Matkin et al. 2012). For the purposes of this stock assessment report, the West Coast Transient stock continues to include animals that occur in California, Oregon, Washington, British Columbia, and Southeast Alaska. Based on data regarding association patterns, acoustics, movements, and genetic differences, eight killer whale stocks are now recognized within the Pacific U.S. Exclusive Economic Zone: 1) the Alaska Resident stock - occurring from Southeast Alaska to the Aleutian Islands and Bering Sea, 2) the Northern Resident stock - occurring from Washington State through part of Southeast Alaska, 3) the Southern Resident stock - occurring mainly within the inland waters of Washington State and southern British Columbia, but also in coastal waters from Southeast Alaska through California, 4) the Gulf of Alaska, Aleutian Islands, and Bering Sea Transient stock - occurring mainly from Prince William Sound through the Aleutian Islands and Bering Sea, 5) the AT1

Transient stock - occurring in Alaska from Prince William Sound through the Kenai Fjords, 6) the West Coast Transient stock - occurring from California through Southeast Alaska (Fig. 1), 7) the Offshore stock - occurring from California through Alaska, and 8) the Hawaiian stock. Transient killer whales in Canadian waters are considered part of the West Coast Transient stock. The Hawaiian and Offshore stocks are reported in the Stock Assessment Reports for the U.S. Pacific Region.

POPULATION SIZE

The West Coast Transient stock is a trans-boundary stock, including killer whales from British Columbia. Preliminary analysis of photographic data resulted in the following minimum counts for transient killer whales belonging to the West Coast Transient stock. Towers et al. (2019) used a 61-year archive of photo-identification data (1958-2018) to assess the portion of the West Coast Transient stock that inhabits Canadian coastal waters and, therefore, was most likely to be impacted by human activity in Canada. Because there is evidence that this population may be composed of discrete population clusters (Parsons et al. 2013, Sharpe et al. 2017), they used a set of criteria to ensure that their analysis represented the animals that were the most regularly and recently documented in Canadian waters. Using only mature individuals, the criteria included the number of encounters, the cumulative number of years documented, and the time since the last encounter. Examination of these data produced a population subset of 349 individuals, including 206 mature individuals plus 143 individuals who were offspring and other inferred maternally related kin. Given that this number was limited to the population likely to be impacted by human activity in British Columbia, and that the California transient numbers have not been updated since the publication of the catalogue in 1997 (Black et al. 1997), the total number of transient killer whales reported above should be considered a minimum count for the West Coast Transient stock.

Minimum Population Estimate

The abundance estimate of killer whales is an analysis of individually identifiable animals. However, the number of catalogued whales does not necessarily represent the number of live animals. Some whales may have died, but they cannot be presumed dead if not resighted because long periods of time between sightings are common for some transient whales. The connection of the "outer coast" whales with the West Coast transient population of inshore waters is not well established, and the photographic catalogue from California has not been updated in 23 years. Estimates of the overall population size (i.e., N_{BEST}) and associated CV(N) that include the outer coast whales are not currently available. Thus, the minimum population estimate (N_{MIN}) of 349 whales for the West Coast Transient stock of killer whales is derived from the recent catalogue for West Coast transient population whales from the inside waters of British Columbia (Towers et al. 2019), which focuses on whales found in Canadian waters (see PBR Guidelines regarding the status of migratory trans-boundary stocks, NMFS 2016). Information on the percentage of time whales typically encountered in Canadian waters spend in U.S. waters is unknown. However, as noted above, this minimum population estimate is considered conservative. This approach is consistent with previous recommendations of the Alaska Scientific Review Group (DeMaster 1996).

Current Population Trend

Recent analyses of the inshore West Coast Transient population indicate that this segment grew rapidly from the mid-1970s to mid-1990s as a result of a combination of high birth rate and survival, as well as greater immigration of animals into the nearshore study area (Fisheries and Oceans Canada 2009). The rapid growth of the West Coast Transient population in the mid-1970s to mid-1990s coincided with a dramatic increase in the abundance of the whales' primary prey, harbor seals, in nearshore waters. Population growth began slowing in the mid-1990s but has increased in recent years (Fisheries and Oceans Canada 2009, Towers et al. 2019). Given that population estimates are based on photo identification of individuals and considered minimum estimates, no reliable estimate of trend is available.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate (R_{MAX}) is not available for the West Coast Transient stock of killer whales. Analyses by Fisheries and Oceans Canada (2009) estimated a rate of increase of about 6% per year in this population from 1975 to 2006; however, this included recruitment of non-calf whales into the population, at least in the first half of the time period, interpreted as either a movement of some whales into nearshore waters from elsewhere or a result of better spatial sampling coverage. The population increased at a rate of approximately 2% for the second half of the time period, when recruitment of new individuals was nearly exclusively from new-born individuals (Fisheries and Oceans Canada 2009). Between 2012 and 2018, Towers et al.

(2019) observed a mean annual growth rate of 4.1% for a population subset in Canadian coastal waters, which was higher than the mean annual growth rate of 2.7% documented by Ford et al. (2013) between 2006 and 2011 for a sub-population of inner-coast transient killer whales that contained most of the same individuals. This rate was also higher than Ford et al.'s (2007) mean annual growth rate of 2% estimated for the same population between 1991 and 2006. However, until additional data become available for the West Coast Transient stock of killer whales, the default cetacean maximum theoretical net productivity rate of 4% will be used for this stock (NMFS 2016).

POTENTIAL BIOLOGICAL REMOVAL

Potential biological removal (PBR) is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: PBR = $N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 0.5, the value for cetacean stocks with unknown population status (NMFS 2016). Thus, for the West Coast Transient killer whale stock, PBR is 3.5 whales (349 × 0.02 × 0.5). The proportion of time that this trans-boundary stock spends in Canadian waters cannot be determined (G. Ellis, Pacific Biological Station, Canada, pers. comm.).

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2014 and 2018 is listed, by marine mammal stock, in Young et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports. The minimum estimated mean annual level of human-caused mortality and serious injury for the West Coast Transient stock of killer whales between 2014 and 2018 is 0.4 killer whales: 0.2 in U.S. commercial fisheries and 0.2 in unknown (commercial, recreational, or subsistence) fisheries. Potential threats most likely to result in direct human-caused mortality or serious injury of this stock include oil spills, vessel strikes, and interactions with fisheries.

Fisheries Information

Information for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is available in Appendix 3 of the Alaska Stock Assessment Reports (observer coverage) and in the NMFS List of Fisheries (LOF) and the fact sheets linked to fishery names in the LOF (observer coverage and reported incidental takes of marine mammals: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-list-fisheries, accessed December 2020).

NMFS observers monitored the California swordfish drift gillnet fishery from 1990 to 2017 (Carretta et al. 2019). The one killer whale mortality observed in this fishery, in 1995, was genetically identified as a transient ecotype. Bycatch estimates for 2013-2017, based on a bycatch model, result in a minimum mean annual mortality and serious injury rate of zero killer whales for this stock (Carretta et al. 2019).

Reports to NMFS Region marine mammal stranding networks of killer whales entangled in fishing gear or with injuries caused by interactions with gear are another source of mortality and serious injury data. A killer whale mortality in commercial California Dungeness crab pot gear in 2015 reported to the NMFS West Coast Region marine mammal stranding network was genetically identified as a transient ecotype. Because the whale could not be assigned to a specific stock, the mean annual mortality and serious injury rate of 0.2 killer whales between 2014 and 2018 was assigned to the Gulf of Alaska, Aleutian Islands, and Bering Sea Transient and the West Coast Transient killer whale stocks; it was not assigned to the AT1 Transient killer whale stock because none of the whales in this population are missing (Table 1; Young et al. 2020). An additional whale, photographically identified as a member of the West Coast Transient stock of killer whales, entangled in and self-released from commercial California Dungeness crab pot gear in 2016; however, this was considered to be a non-serious injury (Young et al. 2020). There was also a report to the NMFS Alaska Region marine mammal stranding network of a killer whale entangled in pot gear in Icy Strait in 2016, resulting in a mean annual mortality and serious injury rate of 0.2 killer whales in unknown (commercial, recreational, or subsistence) Southeast Alaska pot fishery gear between 2014 and 2018 (Table 1; Young et al. 2020). Because the stock identification is unknown, this mortality and serious injury was assigned to the three killer whale stocks that occur in the area: the Alaska Resident, Northern Resident, and West Coast Transient stocks. These mortality and serious injury estimates result from an actual count of verified human-caused deaths and serious injuries and are minimums because not all entangled animals strand nor are all stranded animals found or reported.

The minimum estimated mean annual mortality and serious injury rate incidental to fisheries between 2014 and 2018 is 0.4 killer whales: 0.2 in U.S. commercial fisheries and 0.2 in unknown (commercial, recreational, or subsistence) fisheries.

Table 1. Summary of mortality and serious injury of West Coast Transient killer whales, by year and type, reported to the NMFS Alaska Region and NMFS West Coast Region marine mammal stranding networks between 2014 and 2018 (Young et al. 2020).

Cause of Injury	2014	2015	2016	2017	2018	Mean annual mortality
Entangled in commercial CA Dungeness crab pot gear	0	1[a]	0	0	0	0.2
Entangled in Southeast Alaska pot gear*	0	0	1[b]	0	0	0.2
Total in commercial fisheries						0.2
*Total in unknown (commercial, recreational, or subsistence) fisheries						0.2

[a]This whale was genetically identified as a transient ecotype but could not be assigned to a specific stock; therefore, the mortality was assigned to the Gulf of Alaska, Aleutian Islands, and Bering Sea Transient and the West Coast Transient killer whale stocks.
[b]The stock identification of this whale is unknown; therefore, this mortality was assigned to the three killer whale stocks in the area: the Alaska Resident, Northern Resident, and West Coast Transient killer whale stocks.

All Canadian longline fisheries (including halibut, rockfish, dogfish, sablefish, jig for lingcod, and troll for lingcod and Chinook salmon) are monitored by observers or video. However, only groundfish trawl fisheries have observer or electronic monitoring in Canada, whereas, trawl fisheries for krill, scallop, and shrimp have no observer coverage and salmon net fisheries are not observed (T. Doniol-Valcroze, pers. comm., Department of Fisheries and Oceans, BC, Canada, 14 May 2019). The interaction of Alaska Resident killer whales with the sablefish longline fishery accounts for a large proportion of the commercial fishing/killer whale interactions in Alaska waters. However, transient killer whales typically are not involved in these interactions. Such interactions have not been reported in Canadian waters where sablefish are taken via a pot fishery; however, Northern Resident killer whale interactions with Pacific halibut longline and salmon troll fisheries in British Columbia have been reported (Ford 2014). Reports of killer whale interactions with gillnets in Canadian waters include one killer whale that contacted a salmon gillnet in 1994 but did not entangle (Guenther et al. 1995) and one killer whale (Northern Resident I103) that entangled in a gillnet in 2014 but was quickly released (Fisheries and Oceans Canada 2018).

Alaska Native Subsistence/Harvest Information

Killer whales are not harvested for subsistence in Alaska.

Other Mortality

The shooting of killer whales in Canadian waters has been a concern in the past. Since 1974, however, fresh bullet wounds are rarely, if ever, seen on whales in British Columbia and Washington (Ford et al. 2000, Fisheries and Oceans Canada 2018). In fact, the likelihood of shooting incidents involving transient killer whales is thought to be minimal since commercial fishermen are most likely to observe transients feeding on seals or sea lions instead of interacting with their fishing gear (G. Ellis, Pacific Biological Station, Canada, pers. comm.).

Collisions with vessels are an occasional source of mortality or serious injury of killer whales. For example, a killer whale struck the propeller of a vessel in the Bering Sea/Aleutian Islands flatfish trawl fishery in 2016. Stock identification of this whale is unknown; however, this fishery is outside of the known range of the West Coast Transient stock. There has been no known mortality or serious injury of West Coast Transient killer whales due to vessel collisions.

STATUS OF STOCK

The West Coast Transient killer whale stock is not designated as depleted under the MMPA or listed as threatened or endangered under the Endangered Species Act. In 2001, the Committee on the Status of Endangered Wildlife in Canada designated West Coast Transient killer whales in British Columbia as threatened under the Species at Risk Act (SARA) for Canada. Human-caused mortality may have been underestimated, primarily due to a lack of information on Canadian fisheries, and the minimum abundance estimate is considered conservative (because researchers continue to encounter new whales and provisionally classified whales from Southeast Alaska and off the coast of California were not included), resulting in a conservative PBR estimate. Based on currently available data, the minimum estimated mean annual U.S. commercial fishery-related mortality and serious injury rate (0.2) does not exceed 10% of the PBR (10% of PBR = 0.3) and, therefore, is considered to be insignificant and approaching a zero mortality and serious injury rate. The minimum estimated mean annual level of human-caused mortality and serious injury (0.4) is not known to exceed the PBR (3.5). Therefore, the West Coast Transient stock of killer whales is not classified as a strategic stock. Population trends and status of this stock relative to its Optimum Sustainable Population size are currently unknown.

There are key uncertainties in the assessment of the West Coast Transient stock of killer whales. The current population estimate is for a subset of whales that inhabits Canadian coastal waters and this subset has increased at an average rate of 4.1% per year from 2012 to 2018. However, an updated abundance estimate and growth rate is not available for the entire stock.

HABITAT CONCERNS

Analyses of blubber biopsies collected from mammal-eating transient killer whales and fish-eating resident killer whales in Canadian waters between 1993 and 1996 revealed that transient killer whales and Southern Resident killer whales had surprisingly high levels of persistent PCB contamination; the particularly high levels of contamination found in transient killer whales most likely reflected their higher trophic level (Ross et al. 2000). Due to these high levels of contamination, transient and Southern Resident killer whales in Canadian waters were considered to be at risk for toxic effects (Ross et al. 2000).

CITATIONS

Baird, R. W., and P. J. Stacey. 1988. Variation in saddle patch pigmentation in populations of killer whales (*Orcinus orca*) from British Columbia, Alaska, and Washington State. Can. J. Zool. 66 (11):2582-2585.

Baird, R. W., P. A. Abrams, and L. M. Dill. 1992. Possible indirect interactions between transient and resident killer whales: implications for the evolution of foraging specializations in the genus *Orcinus*. Oecologia 89:125-132.

Barlow, J. 1995. The abundance of cetaceans in California waters. Part I: Ship surveys in summer and fall of 1991. Fish. Bull., U.S. 93:1-14.

Barlow, J. 1997. Preliminary estimates of cetacean abundance off California, Oregon and Washington based on a 1996 ship survey and comparisons of passing and closing modes. Southwest Fisheries Science Center Administrative Report LJ-97-11. 25 p. Available from SWFSC, NMFS, 8901 La Jolla Shores Drive, La Jolla, CA 92037. 25 p.

Barrett-Lennard, L. G. 2000. Population structure and mating patterns of killer whales (*Orcinus orca*) as revealed by DNA analysis. Ph.D. Dissertation, University of British Columbia, Vancouver, BC, Canada. 97 p.

Bigg, M. A., P. F. Olesiuk, G. M. Ellis, J. K. B. Ford, and K. C. Balcomb III. 1990. Social organization and genealogy of resident killer whales (*Orcinus orca*) in the coastal waters of British Columbia and Washington State, p. 386-406. *In* P. S. Hammond, S. A. Mizroch, and G. P. Donovan (eds.), Individual Recognition of Cetaceans: Use of Photo-identification and Other Techniques to Estimate Population Parameters. Rep. Int. Whal. Comm. Special Issue 12.

Black, N. A., A. Schulman-Janiger, R. L. Ternullo, and M. Guerrero-Ruiz. 1997. Killer whales of California and western Mexico: a catalog of photo-identified individuals. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-SWFSC-247, 174 p.

Braham, H. W., and M. E. Dahlheim. 1982. Killer whales in Alaska documented in the Platforms of Opportunity Program. Rep. Int. Whal. Comm. 32:643-646.

Carretta, J. V., J. E. Moore, and K. A. Forney. 2019. Estimates of marine mammal, sea turtle, and seabird bycatch from the California large-mesh drift gillnet fishery: 1990-2017. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-SWFSC-619, 76 p.

Committee on Taxonomy. 2019. List of marine mammal species and subspecies. Society for Marine Mammalogy. Available online: www.marinemammalscience.org . Accessed December 2020.

Dahlheim, M. E., D. Ellifrit, and J. Swenson. 1997. Killer Whales of Southeast Alaska: A Catalogue of Photoidentified Individuals. Day Moon Press, Seattle, WA. 82 p. + appendices.

Dahlheim, M. E., A. Schulman-Janiger, N. Black, R. Ternullo, D. Ellifrit, and K. C. Balcomb. 2008. Eastern temperate North Pacific offshore killer whales (*Orcinus orca*): occurrence, movements, and insights into feeding ecology. Mar. Mammal Sci. 24(3):719-729.

Dahlheim, M. E., and P. A. White. 2010. Ecological aspects of transient killer whales (*Orcinus orca*) as predators in southeastern Alaska. Wildlife Biology 16:308-322.

DeMaster, D. P. 1996. Minutes from the 11-13 September 1996 meeting of the Alaska Scientific Review Group, Anchorage, AK. 20 p + appendices. Available upon request from Marine Mammal Laboratory, AFSC, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Fisheries and Oceans Canada. 2007. Recovery strategy for the transient killer whale (*Orcinus orca*) in Canada. Species at Risk Act Recovery Strategy Series, Fisheries and Oceans Canada, Vancouver. 47 p.

Fisheries and Oceans Canada. 2009. Recovery potential assessment for West Coast Transient killer Whales. DFO Canadian Science Advisory Secretariat Science Advisory Report 2009/039.

Fisheries and Oceans Canada. 2010. Population assessment Pacific harbour seal (*Phoca vitulina richardsi*). DFO Canadian Science Advisory Secretariat Science Advisory Report 2009/011.

Fisheries and Oceans Canada. 2018. Recovery strategy for the Northern and Southern Resident killer whales in Canada. Species at Risk Act Recovery Strategy Series, Fisheries and Oceans Canada, Ottawa. x + 84 p.

Ford, J. K. B. 2011. Killer whales of the Pacific Northwest coast: from pest to paragon. Whale Watcher 40(1):15-23.

Ford, J. K. B. 2014. Marine Mammals of British Columbia. Royal BC Museum Handbook, Mammals of BC, Volume 6. Royal BC Museum, Victoria. 460 p.

Ford, J. K. B., and G. M. Ellis. 1999. Transients: Mammal-Hunting Killer Whales of British Columbia, Washington, and Southeastern Alaska. University of British Columbia Press, Vancouver, BC. 96 p.

Ford, J. K. B., and H. D. Fisher. 1982. Killer whale (*Orcinus orca*) dialects as an indicator of stocks in British Columbia. Rep. Int. Whal. Comm. 32:671-679.

Ford, J. K. B., G. M. Ellis, and K. C. Balcomb. 1994. Killer Whales: The Natural History and Genealogy of *Orcinus orca* in British Columbia and Washington State. University of British Columbia Press, Vancouver, BC, and University of Washington Press, Seattle. 102 p.

Ford, J. K. B., G. M. Ellis, and K. C. Balcomb. 2000. Killer Whales: The Natural History and Genealogy of *Orcinus orca* in British Columbia and Washington State. Second edition. University of British Columbia Press, Vancouver, BC, Canada. 104 p.

Ford, J. K. B., G. M. Ellis, and J. W. Durban. 2007. An assessment of the potential for recovery of West Coast Transient killer whales using coastal waters of British Columbia. DFO Canadian Science Advisory Secretariat Research Document 2007/088.

Ford, J. K. B, E. H. Stredulinsky, J. R. Towers, and G. M. Ellis. 2013. Information in support of the identification of critical habitat for transient killer whales (*Orcinus orca*) off the west coast of Canada. DFO Canadian Science Advisory Secretariat Research Document 2012/nnn.

Forney, K. A., J. Barlow, and J. V. Carretta. 1995. The abundance of cetaceans in California waters. Part II: Aerial surveys in winter and spring of 1991 and 1992. Fish. Bull., U.S. 93:15-26.

Forney, K. A., and P. R. Wade. 2006. World-wide abundance and density of killer whales, p. 145-162. *In* J. A. Estes, D. P. DeMaster, D. F. Doak, T. M. Williams, and R. L. Brownell, Jr. (eds.), Whales, Whaling, and Ocean Ecosystems. University of California Press.

George, J. C., and R. Suydam. 1998. Observations of killer whale (*Orcinus orca*) predation in the northeastern Chukchi and western Beaufort seas. Mar. Mammal Sci. 14:330-332. DOI: dx.doi.org/10.1111/j.1748-7692.1998.tb00722.x .

Goley, P. D., and J. M. Straley. 1994. Attack on gray whales (*Eschrichtius robustus*) in Monterey Bay, California, by killer whales (*Orcinus orca*) previously identified in Glacier Bay, Alaska. Can. J. Zool. 72:1528-1530.

Green, G. A., J. J. Brueggeman, R. A. Grotefendt, C. E. Bowlby, M. L. Bonnell, and K. C. Balcomb. 1992. Cetacean distribution and abundance off Oregon and Washington, 1989-1990, p. 1-100. *In* J. J. Brueggeman (ed.), Oregon and Washington marine mammal and seabird surveys. Final Report OCS Study MMS 91-0093.

Guenther, T. J., R. W. Baird, R. L. Bates, P. M. Willis, R. L. Hahn, and S. G. Wischniowski. 1995. Strandings and fishing gear entanglements of cetaceans on the west coast of Canada in 1994. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/47/O6). 7 p.

Hoelzel, A. R., and G. A. Dover. 1991. Genetic differentiation between sympatric killer whale populations. Heredity 66:191-195.

Hoelzel, A. R., M. E. Dahlheim, and S. J. Stern. 1998. Low genetic variation among killer whales (*Orcinus orca*) in the eastern North Pacific, and genetic differentiation between foraging specialists. J. Hered. 89:121-128.

Hoelzel, A. R., A. Natoli, M. Dahlheim, C. Olavarria, R. Baird, and N. Black. 2002. Low worldwide genetic diversity in the killer whale (*Orcinus orca*): implications for demographic history. Proc. R. Soc. Lond. 269:1467-1473.

Leatherwood, J. S., and M. E. Dahlheim. 1978. Worldwide distribution of pilot whales and killer whales. Naval Ocean Systems Center, Tech. Rep. 443:1-39.

Leatherwood, S., C. O. Matkin, J. D. Hall, and G. M. Ellis. 1990. Killer whales, *Orcinus orca*, photo-identified in Prince William Sound, Alaska 1976 to 1987. Can. Field Nat. 104:362-371.

Matkin, C., G. Ellis, E. Saulitis, L. Barrett-Lennard, and D. Matkin. 1999. Killer Whales of Southern Alaska. North Gulf Oceanic Society, Homer, AK. 96 p.

Matkin, C. O., J. W. Durban, E. L. Saulitis, R. D. Andrews, J. M. Straley, D. R. Matkin, and G. M. Ellis. 2012. Contrasting abundance and residency patterns of two sympatric populations of transient killer whales (*Orcinus orca*) in the northern Gulf of Alaska. Fish. Bull., U.S. 110:143-155.

Mitchell, E. D. 1975. Report on the meeting on small cetaceans, Montreal, April 1-11, 1974. J. Fish. Res. Board Can. 32:914-916.

Morin, P. A., F. I. Archer, A. D. Foote, J. Vilstrup, E. E. Allen, P. Wade, J. Durban, K. Parsons, R. Pitman, L. Li, P. Bouffard, S. C. Abel Nielsen, M. Rasmussen, E. Willerslev, M. T. P. Gilbert, and T. Harkins. 2010. Complete mitochondrial genome phylogeographic analysis of killer whales (*Orcinus orca*) indicates multiple species. Genome Res. 20:908-916.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2020.

Parsons, K. M., J. W. Durban, A. M. Burdin, V. N. Burkanov R. L. Pitman, J. Barlow, L. G. Barrett-Lennard, R. G. LeDuc, K. M. Robertson, C. O. Matkin, and P. R. Wade. 2013. Geographic patterns of genetic differentiation among killer whales in the northern North Pacific. J. Hered. 104:737-754.

Riesch, R., L. G. Barrett-Lennard, G. M. Ellis, J. K. B. Ford, and V. B. Deecke. 2012. Cultural traditions and the evolution of reproductive isolation: ecological speciation in killer whales? Biol. J. Linn. Soc. 106:1-17.

Ross, P. S., G. M. Ellis, M. G. Ikonomou, L. G. Barrett-Lennard, and R. F. Addison. 2000. High PCB concentrations in free-ranging Pacific killer whales, *Orcinus orca*: effects of age, sex and dietary preference. Marine Pollution Bulletin 40(6):504-515.

Saulitis, E., C. O. Matkin, and F. H. Fay. 2005. Vocal repertoire and acoustic behavior of the isolated AT1 killer whale subpopulation in southern Alaska. Can. J. Zool. 83:1015-1029

Sharpe, D. L., M. Castellote, P. R. Wade, and L. A. Cornick. 2017. Call types of Bigg's killer whales (*Orcinus orca*) in western Alaska: using vocal dialects to assess population structure. Bioacoustics 28(1):74-99. DOI: dx.doi.org/10.1080/09524622.2017.1396562 .

Towers, J. R., G. J. Sutton, T. J. H. Shaw, M. Malleson, D. Matkin, B. Gisborne, J. Forde, D. Ellifrit, G. M. Ellis, J. K. B. Ford, and T. Doniol-Valcroze. 2019. Photo-identification catalogue, population status, and distribution of Bigg's killer whales known from coastal waters of British Columbia, Canada. Can. Tech. Rep. Fish. Aquat. Sci. 3311: vi + 299 p.

Young, N. C., B. J. Delean, V. T. Helker, J. C. Freed, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2014-2018. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-413, 142 p.

PACIFIC WHITE-SIDED DOLPHIN (*Lagenorhynchus obliquidens*): North Pacific Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The Pacific white-sided dolphin is found throughout the temperate North Pacific Ocean, north of the coasts of Japan and Baja California, Mexico. In the eastern North Pacific, the species occurs from the southern Gulf of California, north to the Gulf of Alaska, west to Amchitka in the Aleutian Islands, and is sometimes encountered in the southern Bering Sea. The species is common both on the high seas and along the continental margins, and animals are known to enter the inshore passes of Alaska, British Columbia, and Washington (Ferrero and Walker 1996).

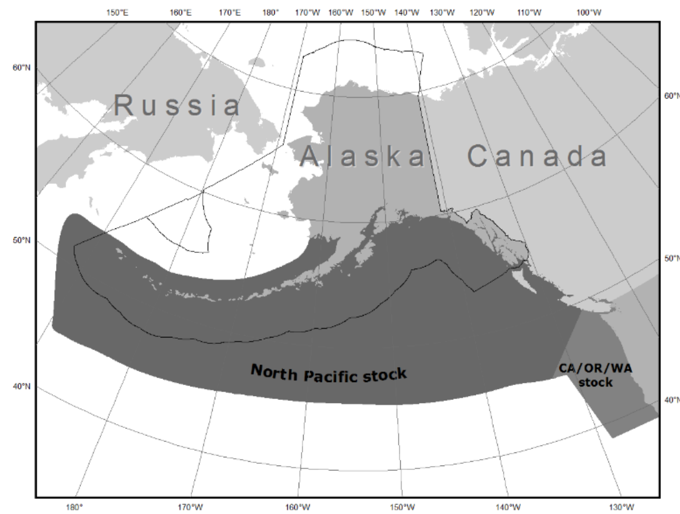

Figure 1. Approximate distribution of Pacific white-sided dolphins in the eastern North Pacific (dark shaded areas). The U.S. Exclusive Economic Zone is delineated by the solid black line.

The following information was considered in classifying Pacific white-sided dolphin stock structure based on the Dizon et al. (1992) phylogeographic approach: 1) Distributional data: geographic distribution is continuous; 2) Population response data: unknown; 3) Phenotypic data: two morphological forms are recognized (Walker et al. 1986, Chivers et al. 1993); and 4) Genotypic data: preliminary genetic analyses on 116 Pacific white-sided dolphins collected in four areas (Baja California, the U.S. west coast, British Columbia/Southeast Alaska, and offshore) do not support phylogeographic partitioning, although they are sufficiently differentiated to be treated as separate management units (Lux et al. 1997). This limited information is not sufficient to define stock structure throughout the North Pacific beyond the generalization that a northern form occurs north of about 33°N from southern California along the coast to Alaska and a southern form ranges from about 36°N southward along the coasts of California and Baja California, while the core of the population ranges across the North Pacific to Japan at latitudes south of 45°N. Data are lacking to determine whether this latter group might include animals from one or both of the coastal forms. Although the genetic data are unclear, management issues support the designation of two stocks; because the California and Oregon thresher shark/swordfish drift gillnet fishery (operating between 33°N and approximately 47°N) and, to a lesser extent, the groundfish and salmon fisheries in Alaska are known to interact with Pacific white-sided dolphins, two management stocks are recognized: 1) the California/Oregon/Washington stock, and 2) the North Pacific stock (Fig. 1). The California/Oregon/Washington stock is reported in the Stock Assessment Reports for the U.S. Pacific Region.

POPULATION SIZE

The most complete population abundance estimate for Pacific white-sided dolphins was calculated from line-transect analyses applied to the 1987-1990 marine mammal sighting survey data across the North Pacific from 25°N and into the Bering Sea (Buckland et al. 1993). The Buckland et al. (1993) abundance estimate, 931,000 dolphins (CV = 0.90), more closely reflects a range-wide estimate rather than one that can be applied to either of the two management stocks off the west coast of North America. Furthermore, Buckland et al. (1993) suggested that Pacific white-sided dolphins show strong vessel attraction but that a correction factor was not available to apply to the estimate. While the Buckland et al. (1993) abundance estimate is not considered appropriate to apply to the management stock in Alaska waters, the portion of the estimate derived from sightings north of 45°N in the Gulf of Alaska can be used as the population estimate for this area (26,880). For comparison, Hobbs and Lerczak (1993) estimated 15,200 Pacific white-sided dolphins (95% CI: 868-265,000) in the Gulf of Alaska. This estimate is based on a single sighting of 20 animals and so should not be used as an abundance estimate. Small cetacean aerial surveys in the Gulf of Alaska during 1997 sighted one group of 164 Pacific white-sided dolphins off Dixon entrance, while similar surveys in Bristol Bay in 1999 made 18 sightings (188 individuals with possible repeat sightings) off Port Moller (MML, unpubl. data).

Minimum Population Estimate

Historically, the minimum population estimate (N_{MIN}) for this stock was 26,880 dolphins, based on the sum of abundance estimates for four separate 5° × 5° blocks north of 45°N (1,970 + 6,427 + 6,101 + 12,382 = 26,880) from surveys conducted during 1987-1990, reported in Buckland et al. (1993). This was considered a minimum estimate because the abundance of animals in a fifth 5° × 5° block (53,885), which straddled the boundary of the two coastal management stocks, was not included in the estimate for the North Pacific stock and because much of the potential habitat for this stock was not surveyed between 1987 and 1990. However, because the abundance estimate is more than 8 years old, N_{MIN} is considered unknown.

Current Population Trend

There is no reliable information on trends in abundance for this stock of Pacific white-sided dolphins.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate (R_{MAX}) is not available for the North Pacific stock of Pacific white-sided dolphins. Life-history analyses by Ferrero and Walker (1996) suggest a reproductive strategy consistent with the delphinid pattern on which the 4% cetacean maximum theoretical net productivity rate was based. Thus, the cetacean maximum theoretical net productivity rate of 4% will be used for this stock (Wade and Angliss 1997).

POTENTIAL BIOLOGICAL REMOVAL

Potential biological removal (PBR) is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: PBR = $N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 0.5, the value for cetacean stocks of unknown status (Wade and Angliss 1997). However, the 2016 guidelines for preparing Stock Assessment Reports (NMFS 2016) state that abundance estimates older than 8 years should not be used to calculate PBR due to a decline in confidence in the reliability of an aged abundance estimate. In addition, there is no corroborating evidence from recent surveys in Alaska that provide abundance estimates for a portion of the stock's range or any indication of the current status of this stock. Therefore, the PBR for this stock is considered undetermined.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals in 2012-2016 is listed, by marine mammal stock, in Helker et al. (in press); however, only the mortality and serious injury data are included in the Stock Assessment Reports. The total estimated annual level of human-caused mortality and serious injury for the North Pacific stock of Pacific white-sided dolphins in 2012-2016 is zero; however, this estimate is considered a minimum because not all of the salmon and herring fisheries operating within the range of this stock have been observed. Potential threats most likely to result in direct human-caused mortality or serious injury of this stock include entanglement in fishing gear.

Fisheries Information

Between 1978 and 1991, mortality and serious injury of thousands of Pacific white-sided dolphins occurred annually incidental to high-seas fisheries for salmon and squid. However, these fisheries were closed in 1991 and no other large-scale fisheries have operated in the central North Pacific since 1991.

Information (including observer programs, observer coverage, and observed incidental takes of marine mammals) for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is presented in Appendices 3-6 of the Alaska Stock Assessment Reports.

No mortality or serious injury of Pacific white-sided dolphins was observed incidental to U.S. federal commercial fisheries in Alaska in 2012-2016 (Breiwick 2013; MML, unpubl. data). However, a complete estimate of the total mortality and serious injury incidental to U.S. commercial fisheries is unavailable for this stock because not all of the salmon and herring fisheries operating within the range of this stock have been observed.

Alaska Native Subsistence/Harvest Information

There are no reports of subsistence takes of Pacific white-sided dolphins in Alaska.

Other Mortality

From 2012 to 2016, no human-caused mortality or serious injury of Pacific white-sided dolphins was reported to the NMFS Alaska Region stranding network (Helker et al. in press).

STATUS OF STOCK

Pacific white-sided dolphins are not designated as depleted under the Marine Mammal Protection Act or listed as threatened or endangered under the Endangered Species Act. The North Pacific stock of Pacific white-sided dolphins is not classified as a strategic stock. The abundance estimate for this stock is unknown because the existing estimate is more than 8 years old and so the PBR level is considered undetermined. Because the PBR is undetermined and fisheries observer coverage is limited, it is unknown if the minimum estimate of the mean annual mortality and serious injury rate (zero) in U.S. commercial fisheries can be considered insignificant and approaching zero mortality and serious injury rate. Population trends and status of this stock relative to its Optimum Sustainable Population are unknown.

There are key uncertainties in the assessment of the North Pacific stock of Pacific white-sided dolphins. The most recent surveys were more than 8 years ago and, given the lack of information on population trend, the abundance estimates are not used to calculate an N_{MIN} and the PBR level is undetermined. Several commercial fisheries overlap with the range of this stock and are not observed or have not been observed in a long time; thus, the estimate of commercial fishery mortality and serious injury is expected to be a minimum estimate.

HABITAT CONCERNS

While the majority of Pacific white-sided dolphins are found throughout the North Pacific, there are also significant numbers found in shelf break and deeper nearshore areas. Thus, they are subject to a variety of habitat impacts. Of particular concern are nearshore areas, bays, channels, and inlets where some Pacific white-sided dolphins are vulnerable to physical modifications of nearshore habitats, resulting from urban and industrial development (including waste management and nonpoint source runoff), and noise (Linnenschmidt et al. 2013, Waite and Shelden 2018).

CITATIONS

Breiwick, J. M. 2013. North Pacific marine mammal bycatch estimation methodology and results, 2007-2011. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-260, 40 p.

Buckland, S. T., K. L. Cattanach, and R. C. Hobbs. 1993. Abundance estimates of Pacific white-sided dolphin, northern right whale dolphin, Dall's porpoise and northern fur seal in the North Pacific, 1987/90, p. 387-407. *In* W. Shaw, R. L. Burgner, and J. Ito (eds.), Biology, distribution and stock assessment of species caught in the high seas driftnet fisheries in the North Pacific Ocean. International North Pacific Fisheries Commission Symposium; 4-6 November 1991, Tokyo, Japan.

Chivers, S. J., K. M. Peltier, W. T. Norman, P. A. Akin, and J. Heyning. 1993. Population structure of cetaceans in California coastal waters. Paper SOCCS9 presented at the Status of California Cetacean Stocks Workshop, held in La Jolla, California, March 31-April 2, 1993. 49 p.

Dizon, A. E., C. Lockyer, W. F. Perrin, D. P. DeMaster, and J. Sisson. 1992. Rethinking the stock concept: a phylogeographic approach. Conserv. Biol. 6:24-36.

Ferrero, R. C., and W. A. Walker. 1996. Age, growth, and reproductive patterns of the Pacific white-sided dolphin (*Lagenorhynchus obliquidens*) taken in high seas drift nets in the central North Pacific Ocean. Can. J. Zool. 74(9):1673-1687.

Helker, V. T., M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. In press. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2012-2016. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-XXX, XXX p.

Hobbs, R. C., and J. A. Lerczak. 1993. Abundance of Pacific white-sided dolphin and Dall's porpoise in Alaska estimated from sightings in the North Pacific Ocean and the Bering Sea during 1987-1991. 13 p. Available from Marine Mammal Laboratory, AFSC, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Linnenschmidt, M., J. Teilmann, T. Akamatsu, R. Dietz, and L. A. Miller. 2013. Biosonar, dive, and foraging activity of satellite tracked harbor porpoises (*Phocoena phocoena*). Mar. Mammal Sci. 29(2):77-97.

Lux, C. A., A. S. Costa, and A. E. Dizon. 1997. Mitochondrial DNA population structure of the Pacific white-sided dolphin. Rep. Int. Whal. Comm. 47:645-652.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2018.

Wade, P. R., and R. Angliss. 1997. Guidelines for assessing marine mammal stocks: report of the GAMMS Workshop April 3-5, 1996, Seattle, Washington. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-OPR-12, 93 p.

Waite, J. M., and K. E. W. Shelden. 2018. The northern extent of Pacific white-sided dolphin (*Lagenorhynchus obliquidens*) distribution in the eastern North Pacific. Northwest. Naturalist 99(2):77-92. DOI: dx.doi.org/10.1898/NWN17-15.1 .

Walker, W. A., S. Leatherwood, K. R. Goodrich, W. F. Perrin, and R. K. Stroud. 1986. Geographical variation and biology of the Pacific white-sided dolphin, *Lagenorhynchus obliquidens*, in the north-eastern Pacific, p. 441-465. *In* M. M. Bryden and R. Harrison (eds.), Research on Dolphins. Clarendon Press, Oxford.

Revised 12/30/2020

HARBOR PORPOISE (*Phocoena phocoena*): Southeast Alaska Stock

NOTE – December 2015: In areas outside of Alaska, studies of harbor porpoise distribution have indicated that stock structure is likely more fine-scaled than is reflected in the Alaska Stock Assessment Reports. No data are available to define stock structure for harbor porpoise on a finer scale in Alaska. However, based on comparisons with other regions, it is likely that several regional and sub-regional populations exist. Should new information on harbor porpoise stocks become available, the harbor porpoise Stock Assessment Reports will be updated.

STOCK DEFINITION AND GEOGRAPHIC RANGE

In the eastern North Pacific Ocean, harbor porpoise range from Point Barrow and offshore areas of the Chukchi Sea, along the Alaska coast, and down the west coast of North America to Point Conception, California (Gaskin 1984, Christman and Aerts 2015). Harbor porpoise primarily frequent the coastal waters of the Gulf of Alaska and Southeast Alaska (Dahlheim et al. 2000, 2009), typically occurring in waters less than 100 m deep; however, occasionally they occur in deeper waters (Hobbs and Waite 2010). Within the inland waters of Southeast Alaska, harbor porpoise distribution is clumped with greatest densities observed in the Glacier Bay/Icy Strait region and near Zarembo and Wrangell Islands and the adjacent waters of Sumner Strait (Dahlheim et al. 2009, 2015). The average density of harbor porpoise in Alaska appears to be less than that reported off the west coast of the continental U.S., although areas of high densities do occur in Glacier Bay and the adjacent waters of Icy Strait, Yakutat Bay, the Copper River Delta, Sitkalidak Strait (Dahlheim et al. 2000, 2009, 2015; Hobbs and Waite 2010), and lower Cook Inlet (Shelden et al. 2014).

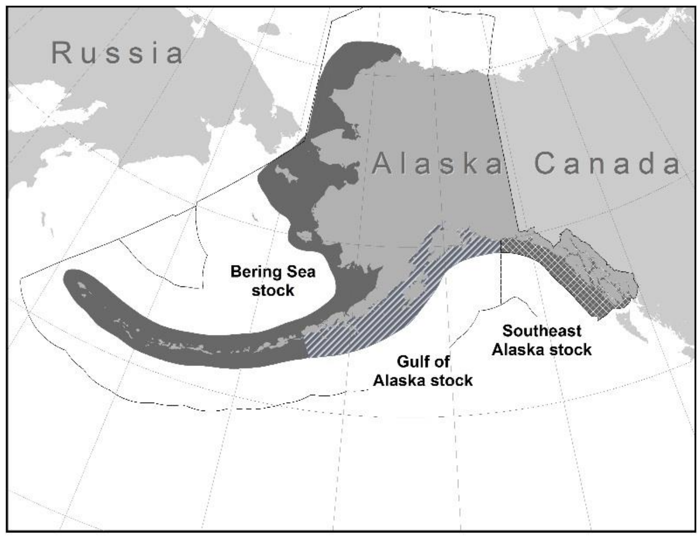

Figure 1. Approximate distribution of harbor porpoise in Alaska waters. The U.S. Exclusive Economic Zone is delineated by a black line.

Stock discreteness in the eastern North Pacific was analyzed using mitochondrial DNA from samples collected along the west coast (Rosel 1992), including one sample from Alaska. Two distinct mitochondrial DNA groupings or clades were found. One clade is present in California, Washington, British Columbia, and the single sample from Alaska (no samples were available from Oregon), while the other is found only in California and Washington. Although these two clades are not geographically distinct by latitude, the results may indicate a low mixing rate for harbor porpoise along the west coast of North America. Investigation of pollutant loads in harbor porpoise ranging from California to the Canadian border also suggests restricted harbor porpoise movements (Calambokidis and Barlow 1991); these results are reinforced by a similar study in the northwest Atlantic (Westgate and Tolley 1999). Further genetic testing of the same samples mentioned above, along with a few additional samples including eight more from Alaska, found differences between some of the four areas investigated, California, Washington, British Columbia, and Alaska, but inference was limited by small sample size (Rosel et al. 1995). Those results demonstrate that harbor porpoise along the west coast of North America are not panmictic and that movement is sufficiently restricted to result in genetic differences (Walton 1997). This is consistent with low movement suggested by genetic analysis of harbor porpoise specimens from the North Atlantic (Rosel et al. 1999). In a genetic analysis of small-scale population structure of eastern North Pacific harbor porpoise, Chivers et al. (2002) included 30 samples from Alaska, 16 of which were from the Copper River Delta, 5 from Barrow, 5 from Southeast Alaska, and 1 sample each from St. Paul, Adak, Kodiak, and Kenai. Unfortunately, no conclusions could be drawn about the genetic structure of harbor porpoise within Alaska because of the insufficient number of samples from each region. Accordingly, harbor porpoise stock structure in Alaska is defined by geographic areas.

Although it is difficult to determine the true stock structure of harbor porpoise populations in the northeast Pacific, from a management standpoint it is prudent to assume that regional populations exist and that they should be managed independently (Rosel et al. 1995, Taylor et al. 1996). Based on the above information, three harbor porpoise stocks in Alaska are currently specified, recognizing that the boundaries of these three stocks are identified primarily based upon geography or perceived areas of porpoise low density: 1) the Southeast Alaska stock - occurring from Dixon Entrance to Cape Suckling, including inland waters (Fig. 1), 2) the Gulf of Alaska stock - occurring from Cape Suckling to Unimak Pass, and 3) the Bering Sea stock - occurring throughout the Aleutian Islands and all waters north of Unimak Pass. There have been no analyses to assess the validity of these stock designations and research to assess substructure is ongoing only within a portion of the Southeast Alaska stock. Preliminary results from the analysis of environmental DNA (eDNA) samples suggested significant genetic differentiation between porpoise concentrations found in Glacier Bay/Icy Strait and around Zarembo/Wrangell Islands (Parsons et al. 2018). Dahlheim et al. (2015) proposed that harbor porpoise in these regions potentially represent different subpopulations based on analogy with other west coast harbor porpoise populations, differences in trends in abundance of the two concentrations, and a possible hiatus in distribution between the two areas. Because eDNA samples were obtained in only one area in the northern region and one area in the southern region, further sampling is needed to better understand substructure within Southeast Alaska, including the connectivity of subpopulations in inland waters and those in adjacent coastal waters. NMFS will consider whether concentrations of harbor porpoise in Glacier Bay/Icy Strait and around Zarembo/Wrangell Islands should be considered "prospective stocks" in a future Stock Assessment Report. Incidental takes from commercial fisheries within a small region (e.g., Wrangell and Zarembo Islands area) are of concern because of the potential impact on undefined localized stocks of harbor porpoise.

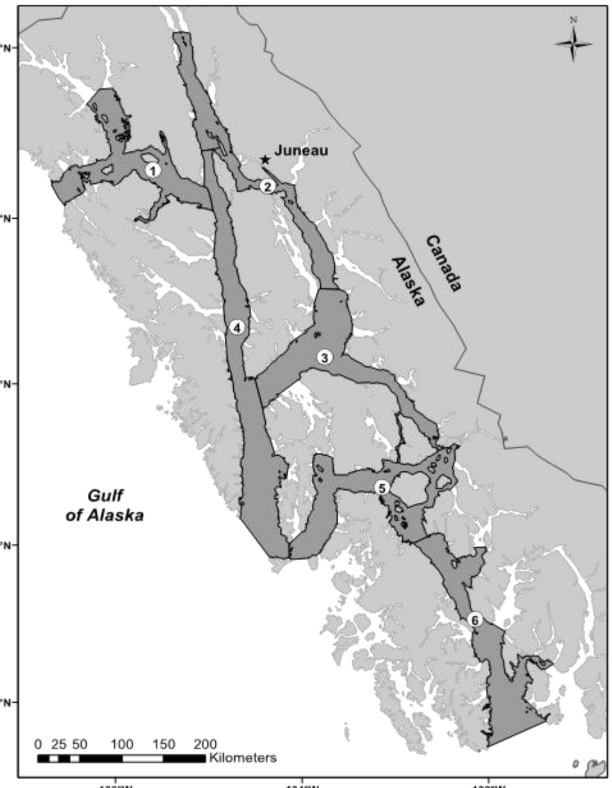

Figure 2. Survey strata defined for line-transect survey effort allocation in Southeast Alaska (as illustrated in Fig. 1 of Dahlheim et al. 2015). The northern region (Areas 1, 2, and 4) includes Cross Sound, Icy Strait, Glacier Bay, Lynn Canal, Stephens Passage, and Chatham Strait; the southern region (Areas 3, 5, and 6) includes Frederick Sound, Sumner Strait, Wrangell and Zarembo Islands, and Clarence Strait as far south as Ketchikan.

POPULATION SIZE

Information on harbor porpoise abundance and relative abundance has been collected for coastal and inside waters of Southeast Alaska by the Alaska Fisheries Science Center's Marine Mammal Laboratory (MML) using both aerial and shipboard surveys. Aerial surveys of this stock were conducted in June and July 1997 and resulted in an observed abundance estimate of 3,766 harbor porpoise (CV = 0.16) (Hobbs and Waite 2010); the surveys included a subset of smaller bays and inlets. Correction factors for observer perception bias and porpoise availability at the surface were used to develop an estimated corrected abundance of 11,146 harbor porpoise (3,766 × 2.96; CV = 0.24) in the coastal and inside waters of Southeast Alaska (Hobbs and Waite 2010).

Relative abundance of harbor porpoise was computed from line-transect surveys carried out in the inland waters of Southeast Alaska in the summers of 1991-1993, 2005-2006, and 2010-2012 (Dahlheim et al 2015). Because these surveys only covered a portion of the inland waters and not the entire range of this stock, they were not used to compute stock-specific estimates of abundance. Relative abundance was found to vary across the 22-year survey period. Abundance estimated in 1991-1993 (N = 1,076; 95% CI = 910-1,272) was higher than the estimate obtained for 2006-2007 (N = 604; 95% CI = 468-780) but comparable to the estimate for 2010-2012 (N =

975; 95% CI = 857-1,109; Dahlheim et al. 2015). There is insufficient information to estimate the probability of detection (g(0)) from the ship surveys in Southeast Alaska; therefore, the abundance estimates above assume the probability of detection directly on the trackline to be unity (g(0) = 1). This assumption is typically violated in harbor porpoise surveys because observers tend to miss animals on the survey trackline. Therefore, the abundances provided by Dahlheim et al. (2015) were corrected using an estimate of g(0) from ship surveys for harbor porpoise off the U.S. east coast (g[0] = 0.72, CV = 0.083: Palka 1995) because the methods used in these surveys (e.g., size of vessels, number of observers) more closely resembled the methods employed in the Southeast Alaska surveys. Estimates corrected for g(0) are N(1991-1993) = 1,494 (95% CI = 1,130-1,974), N(2006-2007) = 839 (95% CI = 494-1,184), and N(2010-2012) = 1,354 (95% CI = 753-1,197).

Using the 2010 to 2012 survey data for the inland waters of Southeast Alaska, Dahlheim et al. (2015) calculated abundance estimates for the concentrations of harbor porpoise in the northern (Areas 1, 2, and 4) and southern (Areas 3, 5, and 6) regions of the inland waters (Fig. 2). The resulting g(0)-corrected abundance estimates are 553 harbor porpoise (CV = 0.13) in the northern inland waters (including Cross Sound, Icy Strait, Glacier Bay, Lynn Canal, Stephens Passage, and Chatham Strait) and 801 harbor porpoise (CV = 0.15) in the southern inland waters (including Frederick Sound, Sumner Strait, Wrangell and Zarembo Islands, and Clarence Strait as far south as Ketchikan).

A line-transect vessel survey was carried out in the inland waters of Southeast Alaska in July/August 2019 and data analysis is underway to compute new estimates of harbor porpoise abundance in this area.

Minimum Population Estimate

For the Southeast Alaska stock of harbor porpoise, the minimum population estimate (N_{MIN}) for the 2010-2012 shipboard surveys is 1,224 porpoise calculated using Equation 1 from the potential biological removal (PBR) guidelines (NMFS 2016): $N_{MIN} = N/\exp(0.842\times[\ln(1+[CV(N)]^2)]^{½})$, where N = 1,354 (assumes g(0) = 0.72) and CV = 0.12. Since this abundance estimate represents some portion of the total number of animals in the stock, using this estimate to calculate N_{MIN} results in a negatively-biased N_{MIN} for the stock. Although harbor porpoise in the northern and southern regions of the inland waters of Southeast Alaska have not been determined to be subpopulations or stocks, PBR calculations for these areas may provide a frame of reference for comparison to harbor porpoise mortality and serious injury in the portion of the Southeast Alaska salmon drift gillnet fishery that was monitored in 2012 and 2013. The pooled 2010 to 2012 abundance estimates of 553 (CV = 0.13; assumes g(0) = 0.72) for the northern region and 801 (CV = 0.15; assumes g(0) = 0.72) for the southern region results in N_{MINs} of 496 and 707, respectively. Alaska Department of Fish and Game (ADF&G) Districts 6, 7, and 8, where the Southeast Alaska salmon drift gillnet fishery was observed in 2012 and 2013 (Manly 2015), partially overlap porpoise survey areas (Areas 5 and 6: Dahlheim et al. 2015) in the southern region of the inland waters.

Current Population Trend

An analysis of the line-transect vessel survey data collected throughout the inland waters of Southeast Alaska between 1991 and 2010 suggested high probabilities of a population decline ranging from 2 to 4% per year for the whole study area and highlighted a potentially important conservation issue (Zerbini et al. 2011). However, when data from 2011 and 2012 were added to this analysis, the population decline was no longer significant (Dahlheim et al. 2015). It is unclear why a negative trend in harbor porpoise numbers was detected in inland waters of Southeast Alaska between 1991 and 2010 and reversed thereafter (Dahlheim et al. 2015). Regionally, abundance was relatively constant in the northern region of the inland waters of Southeast Alaska throughout the survey period, while declines and subsequent increases were documented in the southern region (Dahlheim et al. 2015).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate (R_{MAX}) is not available for the Southeast Alaska stock of harbor porpoise. Until additional data become available, the cetacean maximum theoretical net productivity rate of 4% will be used (NMFS 2016).

POTENTIAL BIOLOGICAL REMOVAL

PBR is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: PBR = $N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 0.5, the value for cetacean stocks with unknown population status (NMFS 2016). Using the N_{MIN} of 1,224 (based on the 2010 to 2012 abundance estimate for harbor porpoise in the inland waters of Southeast Alaska), PBR is 12 harbor porpoise (1,224 × 0.02 × 0.5).

Computing N_{MINs} and PBRs for harbor porpoise in the northern and southern regions of the inland waters of Southeast Alaska may provide a frame of reference for the observed mortality and serious injury of harbor porpoise in the portion of the Southeast Alaska salmon drift gillnet fishery that was monitored in 2012 and 2013. Based on the pooled 2010 to 2012 abundance estimates and corresponding N_{MINs}, the PBR calculations for the northern and southern regions of the inland waters of Southeast Alaska are 5.0 (N = 553; CV = 0.13; N_{MIN} = 496) and 7.1 (N = 801; CV = 0.15; N_{MIN} = 707) harbor porpoise, respectively.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2014 and 2018 is listed, by marine mammal stock, in Young et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports. The minimum estimated mean annual level of human-caused mortality and serious injury for Southeast Alaska harbor porpoise between 2014 and 2018 is 34 porpoise in U.S. commercial fisheries; however, this estimate is considered a minimum because not all of the salmon and herring fisheries operating within the range of this stock have been observed. Potential threats most likely to result in direct human-caused mortality or serious injury of this stock include entanglement in fishing gear.

Fisheries Information

Information for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is available in Appendix 3 of the Alaska Stock Assessment Reports (observer coverage) and in the NMFS List of Fisheries (LOF) and the fact sheets linked to fishery names in the LOF (observer coverage and reported incidental takes of marine mammals: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-list-fisheries, accessed December 2020).

No mortality or serious injury of harbor porpoise from the Southeast Alaska stock was observed incidental to federally-managed U.S. commercial fisheries in Alaska between 2014 and 2018.

In 2007 and 2008, the Alaska Marine Mammal Observer Program (AMMOP) placed observers in four regions where the state-managed Yakutat salmon set gillnet fishery operates (Manly 2009). These regions included the Alsek River area, the Situk area, the Yakutat Bay area, and the Kaliakh River and Tsiu River areas. Based on four mortalities and serious injuries observed during these 2 years, the estimated mean annual mortality and serious injury rate in the Yakutat salmon set gillnet fishery was 22 harbor porpoise (Table 1). Although these observer data are dated, they are considered the best available data on mortality and serious injury levels in these fisheries.

In 2012 and 2013, the AMMOP placed observers on independent vessels in the state-managed Southeast Alaska salmon drift gillnet fishery in ADF&G Management Districts 6, 7, and 8 to assess mortality and serious injury of marine mammals (Manly 2015). These Management Districts cover areas of Frederick Sound, Sumner Strait, Clarence Strait, and Anita Bay which include, but are not limited to, areas around and adjacent to Petersburg and Wrangell and Zarembo Islands. In 2013, four harbor porpoise were observed entangled and released: two were determined to be seriously injured and two were determined to be not seriously injured. Based on the two observed serious injuries, 23 serious injuries were estimated for Districts 6, 7, and 8 in 2013, resulting in an estimated mean annual mortality and serious injury rate of 12 harbor porpoise in 2012 and 2013 (Table 1). Since these three districts represent only a portion of the overall fishing effort in this fishery, this is a minimum estimate of mortality and serious injury for the fishery.

Table 1. Summary of incidental mortality and serious injury of Southeast Alaska harbor porpoise due to U.S. commercial fisheries between 2014 and 2018 (or the most recent data available) and calculation of the mean annual mortality and serious injury rate (Manly 2009, 2015). Methods for calculating percent observer coverage are described in Appendix 3 of the Alaska Stock Assessment Reports.

Fishery name	Years	Data type	Percent observer coverage	Observed mortality	Estimated mortality	Mean estimated annual mortality
Yakutat salmon set gillnet	2007 2008	obs data	5.3 7.6	1 3	16.1 27.5	22 (CV = 0.54)
Southeast Alaska salmon drift gillnet (Districts 6, 7, and 8)	2012 2013	obs data	6.4 6.6	0 2	0 23	12 (CV = 1.0)
Minimum total estimated annual mortality						34 (CV = 0.77)

A complete estimate of the total mortality and serious injury incidental to U.S. commercial fisheries is not available for this stock because not all of the salmon and herring fisheries operating within the range of this stock have been observed. Based on observed mortality and serious injury in two commercial fisheries (Table 1), the minimum estimated mean annual mortality and serious injury rate incidental to U.S. commercial fisheries between 2014 and 2018 is 34 harbor porpoise.

Alaska Native Subsistence/Harvest Information

Subsistence hunters in Alaska have not been reported to take from this stock of harbor porpoise.

STATUS OF STOCK

Southeast Alaska harbor porpoise are not designated as depleted under the Marine Mammal Protection Act or listed as threatened or endangered under the Endangered Species Act. The minimum estimated mean annual level of human-caused mortality and serious injury for Southeast Alaska harbor porpoise (34 porpoise) exceeds the calculated PBR (12 porpoise), which means this stock is strategic. The minimum estimated mean annual U.S. commercial fishery-related mortality and serious injury rate (34 porpoise) is more than 10% of the calculated PBR (10% of PBR = 1.2 porpoise), so it is not considered insignificant and approaching a zero mortality and serious injury rate. However, the calculated PBR is likely biased low for the entire stock because it is based on estimates from the 2010 to 2012 surveys of only a portion (the inside waters of Southeast Alaska) of the range of this stock as currently designated. Population trends and status of this stock relative to its Optimum Sustainable Population are currently unknown.

There are key uncertainties in the assessment of the Southeast Alaska stock of harbor porpoise. This stock likely comprises multiple, smaller stocks based on analogy with harbor porpoise populations that have been the focus of specific studies on stock structure. Concentrations of harbor porpoise in the northern and southern regions of the inland waters of Southeast Alaska are identified, and N_{MINs} and PBR levels are calculated for these areas. The trend in abundance of harbor porpoise in these regions is unclear; an early decline appears to have reversed in recent years. Several commercial fisheries overlap with the range of this stock and are not observed or have not been observed in a long time; thus, the estimate of commercial fishery mortality and serious injury is expected to be a minimum estimate.

HABITAT CONCERNS

Harbor porpoise are mostly found in nearshore areas and inland waters, including bays, tidal areas, and river mouths (Dahlheim et al. 2000, 2009, 2015; Hobbs and Waite 2010). As a result, harbor porpoise are vulnerable to physical modifications of nearshore habitats resulting from urban and industrial development (including waste management and nonpoint source runoff) and activities such as construction of docks and other over-water structures, filling of shallow areas, dredging, and noise (Linnenschmidt et al. 2013).

Algal toxins are a growing concern in Alaska marine food webs, in particular the neurotoxins domoic acid and saxitoxin. While saxitoxin was not detected in harbor porpoise samples collected in Alaska, domoic acid was found in 40% (2 of 5) of the samples and, notably, in maternal transfer to a fetus (Lefebvre et al. 2016).

CITATIONS

Calambokidis, J., and J. Barlow. 1991. Chlorinated hydrocarbon concentrations and their use for describing population discreteness in harbor porpoises from Washington, Oregon, and California, p. 101-110. *In* J. E. Reynolds III and D. K. Odell (eds.), Proceedings of the Second Marine Mammal Stranding Workshop: 3-5 December 1987, Miami, Florida. U.S. Dep. Commer., NOAA Tech. Rep. NMFS-98.

Chivers, S. J., A. E. Dizon, P. J. Gearin, and K. M. Robertson. 2002. Small-scale population structure of eastern North Pacific harbor porpoise (*Phocoena phocoena*) indicated by molecular genetic analyses. J. Cetacean Res. Manage. 4(2):111-122.

Christman, C. L., and L. M. Aerts. 2015. Harbor porpoise (*Phocoena phocoena*) sightings from shipboard surveys in the Chukchi Sea during summer and fall, 2008-2014, p. 197. *In* Book of Abstracts, 2015 Alaska Marine Science Symposium, Anchorage, Alaska, January 19-23, 2015.

Dahlheim, M., A. York, R. Towell, J. Waite, and J. Breiwick. 2000. Harbor porpoise (*Phocoena phocoena*) abundance in Alaska: Bristol Bay to Southeast Alaska, 1991-1993. Mar. Mammal Sci. 16:28-45.

Dahlheim, M., P. A. White, and J. Waite. 2009. Cetaceans of Southeast Alaska: distribution and seasonal occurrence. J. Biogeogr. 36(3):410-426.

Dahlheim, M. E., A. N. Zerbini, J. M. Waite, and A. S. Kennedy. 2015. Temporal changes in abundance of harbor porpoise (*Phocoena phocoena*) inhabiting the inland waters of Southeast Alaska. Fish. Bull., U.S. 113(3):242-255.

Gaskin, D. E. 1984. The harbor porpoise *Phocoena phocoena* (L.): regional populations, status, and information on direct and indirect catches. Rep. Int. Whal. Comm. 34:569-586.

Hobbs, R. C., and J. M. Waite. 2010. Abundance of harbor porpoise (*Phocoena phocoena*) in three Alaskan regions, corrected for observer errors due to perception bias and species misidentification, and corrected for animals submerged from view. Fish. Bull., U.S. 108(3):251-267.

Lefebvre, K. A., L. Quakenbush, E. Frame, K. Burek Huntington, G. Sheffield, R. Stimmelmayr, A. Bryan, P. Kendrick, H. Ziel, T. Goldstein, J. A. Snyder, T. Gelatt, F. Gulland, B. Dickerson, and V. Gill. 2016. Prevalence of algal toxins in Alaskan marine mammals foraging in a changing arctic and subarctic environment. Harmful Algae 55:13-24. DOI: dx.doi.org/10.1016/j.hal.2016.01.007 .

Linnenschmidt, M., J. Teilmann, T. Akamatsu, R. Dietz, and L. A. Miller. 2013. Biosonar, dive, and foraging activity of satellite tracked harbor porpoises (*Phocoena phocoena*). Mar. Mammal Sci. 29(2):77-97.

Manly, B. F. J. 2009. Incidental catch of marine mammals and birds in the Yakutat salmon set gillnet fishery, 2007 and 2008. Final Report to NMFS Alaska Region. 96 p.

Manly, B. F. J. 2015. Incidental takes and interactions of marine mammals and birds in districts 6, 7, and 8 of the Southeast Alaska salmon drift gillnet fishery, 2012 and 2013. Final Report to NMFS Alaska Region. 52 p.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2020.

Palka, D. 1995. Abundance estimate of the Gulf of Maine harbor porpoise. Rep. Int. Whal. Comm. (Special Issue 16):27-50.

Parsons, K. M., M. Everett, M. Dahlheim, and L. Park. 2018. Water, water everywhere: environmental DNA can unlock population structure in elusive marine species. Royal Society Open Science 5:180537. DOI: dx.doi.org/10.1098/rsos.180537 .

Rosel, P. E. 1992. Genetic population structure and systematic relationships of some small cetaceans inferred from mitochondrial DNA sequence variation. Ph.D. Dissertation, University of California San Diego. 191 p.

Rosel, P. E., A. E. Dizon, and M. G. Haygood. 1995. Variability of the mitochondrial control region in populations of the harbour porpoise, *Phocoena phocoena*, on inter-oceanic and regional scales. Can. J. Fish. Aquat. Sci. 52:1210-1219.

Rosel, P. E., R. Tiedemann, and M. Walton. 1999. Genetic evidence for limited trans-Atlantic movements of the harbor porpoise *Phocoena phocoena*. Mar. Biol. 133: 583-591.

Shelden, K. E. W., B. A. Agler, J. J. Brueggeman, L. A. Cornick, S. G. Speckman, and A. Prevel-Ramos. 2014. Harbor porpoise, *Phocoena phocoena vomerina*, in Cook Inlet, Alaska. Mar. Fish. Rev. 76(1-2):22-50.

Taylor, B. L., P. R. Wade, D. P. DeMaster, and J. Barlow. 1996. Models for management of marine mammals. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/48/SM50). 12 p.

Walton, M. J. 1997. Population structure of harbour porpoises *Phocoena phocoena* in the seas around the UK and adjacent waters. Proc. R. Soc. Lond. B 264:89-94.

Westgate, A. J., and K. A. Tolley. 1999. Geographical differences in organochlorine contaminants in harbour porpoises *Phocoena phocoena* from the western North Atlantic. Mar. Ecol. Prog. Ser. 177:255-268.

Young, N. C., B. J. Delean, V. T. Helker, J. C. Freed, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2014-2018. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-413, 142 p.

Zerbini, A. N., M. E. Dahlheim, J. M. Waite, A. S. Kennedy, P. R. Wade, and P. J. Clapham. 2011. Evaluation of population declines of harbor porpoise (*Phocoena phocoena*) in Southeastern Alaska inland waters, p. 23. *In* Book of Abstracts, 19th Biennial Conference on the Biology of Marine Mammals, Tampa, Florida, USA, 28 November-2 December 2011.

Revised 12/30/2020

HARBOR PORPOISE (*Phocoena phocoena*): Gulf of Alaska Stock

NOTE – December 2015: In areas outside of Alaska, studies of harbor porpoise distribution have indicated that stock structure is likely more fine-scaled than is reflected in the Alaska Stock Assessment Reports. No data are available to define stock structure for harbor porpoise on a finer scale in Alaska. However, based on comparisons with other regions, it is likely that several regional and sub-regional populations exist. Should new information on harbor porpoise stocks become available, the harbor porpoise Stock Assessment Reports will be updated.

STOCK DEFINITION AND GEOGRAPHIC RANGE

In the eastern North Pacific Ocean, the harbor porpoise ranges from Point Barrow and offshore areas of the Chukchi Sea, along the Alaska coast, and down the west coast of North America to Point Conception, California (Gaskin 1984, Christman and Aerts 2015). Harbor porpoise primarily frequent the coastal waters of the Gulf of Alaska and Southeast Alaska (Dahlheim et al. 2000, 2009), typically occurring in waters less than 100 m deep; however, occasionally they occur in deeper waters (Hobbs and Waite 2010). The average density of harbor porpoise in Alaska appears to be less than that reported off the west coast of the continental U.S., although areas of high densities do occur in Glacier Bay and the adjacent waters of Icy Strait, Yakutat Bay, the Copper River Delta, Sitkalidak Strait (Dahlheim et al. 2000, 2009, 2015; Hobbs and Waite 2010; Castellote et al. 2015), and lower Cook Inlet (Shelden et al. 2014).

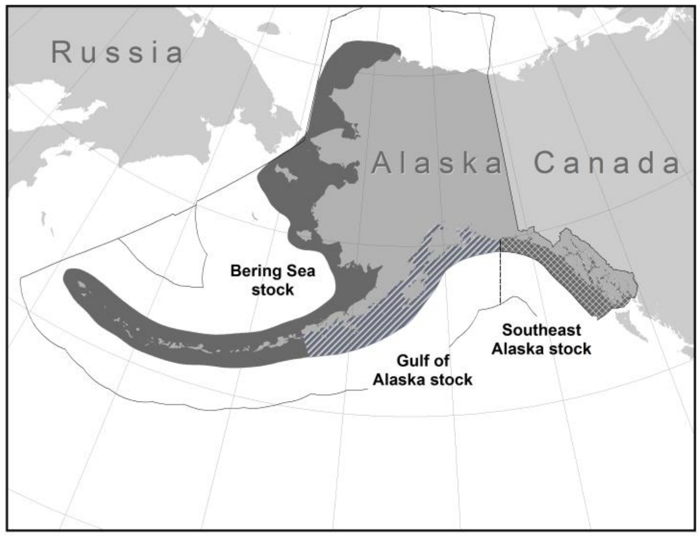

Figure 1. Approximate distribution of harbor porpoise in Alaska waters. The U.S. Exclusive Economic Zone is delineated by a black line.

Stock discreteness in the eastern North Pacific was analyzed using mitochondrial DNA from samples collected along the west coast (Rosel 1992), including one sample from Alaska. Two distinct mitochondrial DNA groupings or clades were found. One clade is present in California, Washington, British Columbia, and the single sample from Alaska (no samples were available from Oregon), while the other is found only in California and Washington. Although these two clades are not geographically distinct by latitude, the results may indicate a low mixing rate for harbor porpoise along the west coast of North America. Investigation of pollutant loads in harbor porpoise ranging from California to the Canadian border also suggests restricted harbor porpoise movements (Calambokidis and Barlow 1991); these results are reinforced by a similar study in the northwest Atlantic (Westgate and Tolley 1999). Further genetic testing of the same samples mentioned above, along with a few additional samples including eight more from Alaska, found differences between some of the four areas investigated, California, Washington, British Columbia, and Alaska, but inference was limited by small sample size (Rosel et al. 1995). Those results demonstrate that harbor porpoise along the west coast of North America are not panmictic and that movement is sufficiently restricted to result in genetic differences (Walton 1997). This is consistent with low movement suggested by genetic analysis of harbor porpoise specimens from the North Atlantic (Rosel et al. 1999). In a genetic analysis of small-scale population structure of eastern North Pacific harbor porpoise, Chivers et al. (2002) included 30 samples from Alaska, 16 of which were from the Copper River Delta, 5 from Barrow, 5 from Southeast Alaska, and 1 sample each from St. Paul, Adak, Kodiak, and Kenai. Unfortunately, no conclusions could be drawn about the genetic structure of harbor porpoise within Alaska because of the insufficient number of samples from each region. Accordingly, harbor porpoise stock structure in Alaska is defined by geographic areas.

Although it is difficult to determine the true stock structure of harbor porpoise populations in the northeast Pacific, from a management standpoint it is prudent to assume that regional populations exist and that they should be managed independently (Rosel et al. 1995, Taylor et al. 1996). Based on the above information, three harbor porpoise stocks in Alaska are currently specified, recognizing that the boundaries of these three stocks are inferred primarily based upon geography or perceived areas of low porpoise density: 1) the Southeast Alaska stock - occurring from Dixon Entrance to Cape Suckling, including inland waters, 2) the Gulf of Alaska stock - occurring from Cape Suckling to Unimak Pass (Fig. 1), and 3) the Bering Sea stock - occurring throughout the Aleutian Islands and all waters north of Unimak Pass. There have been no analyses to assess the validity of these stock designations and research to assess substructure is ongoing only within the Southeast Alaska stock (see the Southeast Alaska harbor porpoise Stock Assessment Report and Parsons et al. 2018).

POPULATION SIZE

In June and July of 1998 and 1999, an aerial survey covered the waters of the western Gulf of Alaska from Cape Suckling to Unimak Island, offshore to the 1,000 fathom depth contour. Two types of corrections were needed for these aerial surveys: one to correct for animals available but not counted because they were not detected by the observers (perception bias) and another to correct for porpoise that were submerged and not available at the surface (availability bias). The 1998 survey resulted in an abundance estimate for the Gulf of Alaska harbor porpoise stock of 10,489 porpoise (coefficient of variation (CV) = 0.12) (Hobbs and Waite 2010), which includes a correction factor (1.372; CV = 0.07) for perception bias. Laake et al. (1997) estimated the availability bias correction factor for aerial surveys of harbor porpoise in Puget Sound to be 2.96 (CV = 0.18); the use of this correction factor is preferred to other published correction factors (e.g., Barlow et al. 1988, Calambokidis et al. 1993) because it is an empirical estimate of availability bias. Hobbs and Waite (2010) applied the Laake et al. (1997) correction factor to the 1998 estimate, resulting in a corrected abundance of 31,046 porpoise (10,489 × 2.96 = 31,046; CV = 0.21) for the Gulf of Alaska stock.

This latest estimate of abundance (31,046) is considerably higher than the estimate reported in the 1999 stock assessment (8,271; CV = 0.31), which was based on surveys conducted in 1991-1993. This disparity largely stems from changes in the area covered by the two surveys and differences in harbor porpoise density encountered in areas added to, or dropped from, the 1998 survey relative to the 1991 to 1993 surveys. The survey area in 1998 (119,183 km^2) was greater than the area covered in the combined portions of the 1991, 1992, and 1993 surveys (106,600 km^2). The 1998 survey included selected bays, channels, and inlets in Prince William Sound, the outer Kenai Peninsula, the south side of the Alaska Peninsula, and the Kodiak Archipelago, whereas, the earlier survey included only open water areas. Several of the bays and inlets covered by the 1998 survey had higher harbor porpoise densities than were observed in the open waters. In addition, the 1998 estimate provided by Hobbs and Waite (2010) empirically estimates the perception bias and uses this in addition to the correction factor for availability bias. Finally, the 1998 estimate extrapolates available densities to estimate the number of porpoise which would likely be found in unsurveyed inlets within the study area. For these reasons, the 1998 survey result is probably more representative of the size of the Gulf of Alaska harbor porpoise stock.

Minimum Population Estimate

The minimum population estimate (N_{MIN}) for this stock is calculated using Equation 1 from the potential biological removal (PBR) guidelines (NMFS 2016): $N_{MIN} = N/\exp(0.842\times[\ln(1+[CV(N)]^2)]^{½})$. Using the population estimate (N) of 31,046 in 1998 and its associated CV of 0.21, N_{MIN} for the Gulf of Alaska stock of harbor porpoise is 26,064. However, because the survey data are now more than 8 years old, N_{MIN} is considered unknown for this stock.

Current Population Trend

There is no reliable information on trends in abundance for the Gulf of Alaska stock of harbor porpoise because survey methods and results are not comparable.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate (R_{MAX}) is not available for the Gulf of Alaska stock of harbor porpoise. Until additional data become available, the cetacean maximum theoretical net productivity rate of 4% will be used (NMFS 2016).

POTENTIAL BIOLOGICAL REMOVAL

PBR is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: PBR = $N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 0.5, the value for cetacean stocks with unknown population status (NMFS 2016). However, the 2016 guidelines for preparing Stock Assessment Reports (NMFS 2016) state that abundance estimates older than 8 years should not be used to calculate PBR due to a decline in confidence in the reliability of an aged abundance estimate. Therefore, the PBR for this stock is considered undetermined.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2014 and 2018 is listed, by marine mammal stock, in Young et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports. The minimum estimated mean annual level of human-caused mortality and serious injury for Gulf of Alaska harbor porpoise between 2014 and 2018 is 72 porpoise: 72 in U.S. commercial fisheries and 0.2 in unknown (commercial, recreational, or subsistence) fisheries; however, this estimate is considered a minimum because of the absence of observer placements in all of the salmon and herring fisheries operating within the range of this stock. Potential threats most likely to result in direct human-caused mortality or serious injury of this stock include entanglement in fishing gear.

Fisheries Information

Information for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is available in Appendix 3 of the Alaska Stock Assessment Reports (observer coverage) and in the NMFS List of Fisheries (LOF) and the fact sheets linked to fishery names in the LOF (observer coverage and reported incidental takes of marine mammals: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-list-fisheries, accessed December 2020).

No incidental mortality or serious injury of Gulf of Alaska harbor porpoise was observed in U.S. federal commercial fisheries between 2014 and 2018. Alaska Marine Mammal Observer Program (AMMOP) observers monitoring the State of Alaska-managed Prince William Sound salmon drift gillnet fishery in 1990 and 1991 recorded 1 mortality in 1990 and 3 in 1991, which extrapolated to 8 (95% CI: 1-23) and 32 (95% CI: 3-103) for the entire fishery, resulting in a mean annual mortality and serious injury rate of 20 porpoise (CV = 0.60) when averaged over 1990 and 1991 (Table 1; Wynne et al. 1991, 1992). The Prince William Sound salmon drift gillnet fishery has not been observed since 1991 and no additional data are available for this fishery.

In 1999 and 2000, AMMOP observers were placed on state-managed Cook Inlet salmon set and drift gillnet vessels. One harbor porpoise mortality was observed in 2000 in the Cook Inlet salmon drift gillnet fishery (Manly 2006). This single mortality extrapolates to an estimated mortality and serious injury rate of 31 porpoise for that year and an average of 16 porpoise per year when averaged over the 2 years of observer data (Table 1).

In 2002 and 2005, AMMOP observers were placed on state-managed Kodiak Island set gillnet vessels. Harbor porpoise mortality observed in this fishery (two each in both 2002 and 2005) (Manly 2007) extrapolates to an estimated mean annual mortality and serious injury rate of 36 harbor porpoise (Table 1). Although these observer data are dated, they are considered the best available data on mortality and serious injury levels in these fisheries.

Table 1. Summary of incidental mortality and serious injury of Gulf of Alaska harbor porpoise due to state-managed fisheries from 1990 through 2005 and calculation of the mean annual mortality and serious injury rate (Wynne et al. 1991, 1992; Manly 2006, 2007). Methods for calculating percent observer coverage are described in Appendix 3 of the Alaska Stock Assessment Reports.

Fishery name	Years	Data type	Percent observer coverage	Observed mortality	Estimated mortality	Mean estimated annual mortality
Prince William Sound salmon drift gillnet	1990 1991	obs data	4 5	1 3	8 32	20 (CV = 0.60)
Cook Inlet salmon drift gillnet	1999 2000	obs data	1.6 3.6	0 1	0 31	16 (CV = 1.00)
Cook Inlet salmon set gillnet	1999 2000	obs data	0.16-1.1 0.34-2.7	0 0	0 0	0
Kodiak Island salmon set gillnet	2002 2005	obs data	6.0 4.9	2 2	32 39	36 (CV = 0.68)
Minimum total estimated annual mortality						72 (CV = 0.44)

Reports to the NMFS Alaska Region marine mammal stranding network of marine mammals with fishing gear attached or with injuries caused by interactions with fishing gear are another source of mortality data. A harbor porpoise mortality, due to entanglement in unidentified fishing net near Homer, Alaska, was reported in 2014, resulting in a minimum mean annual mortality and serious injury rate of 0.2 harbor porpoise from this stock in unknown (commercial, recreational, or subsistence) fisheries between 2014 and 2018 (Table 2; Young et al. 2020). This mortality and serious injury estimate results from an actual count of verified human-caused deaths and serious injuries and is a minimum because not all entangled animals strand nor are all stranded animals found, reported, or have the cause of death determined.

Table 2. Summary of incidental mortality and serious injury of Gulf of Alaska harbor porpoise, by year and type, reported to the NMFS Alaska Region marine mammal stranding network between 2014 and 2018 (Young et al. 2020).

Cause of Injury	**2014**	**2015**	**2016**	**2017**	**2018**	**Mean annual mortality**
Entangled in unidentified net*	1	0	0	0	0	0.2
*Total in unknown (commercial, recreational, or subsistence) fisheries						0.2

A complete estimate of the total mortality and serious injury incidental to U.S. commercial fisheries is unavailable for this stock because of the absence of an observer program for all of the salmon and herring fisheries operating within the range of this stock. Based on observed mortality and serious injury in four commercial fisheries (Table 1) and a report to the NMFS Alaska Region stranding network (Table 2), the minimum estimated mean annual mortality and serious injury rate incidental to all fisheries between 2014 and 2018 is 72 harbor porpoise from this stock (72 in U.S. commercial fisheries + 0.2 in unknown fisheries).

Alaska Native Subsistence/Harvest Information

Porpoise in the Gulf of Alaska were hunted by prehistoric societies from Kodiak Island and areas around Cook Inlet and Prince William Sound (Shelden et al. 2014). Subsistence hunters have not been reported to harvest from this stock of harbor porpoise since the early 1900s (Shelden et al. 2014).

STATUS OF STOCK

Gulf of Alaska harbor porpoise are not designated as depleted under the Marine Mammal Protection Act or listed as threatened or endangered under the Endangered Species Act. The abundance estimate for this stock is unknown because the existing estimate is more than 8 years old and so the PBR level is considered undetermined. Because the PBR is undetermined and fisheries observer coverage is limited and aged, it is unknown if the minimum estimate of the mean annual mortality and serious injury rate (72 porpoise) in U.S. commercial fisheries can be considered insignificant and approaching a zero mortality and serious injury rate. NMFS considers this stock strategic because the level of mortality and serious injury would likely exceed the PBR level if we had accurate information on stock structure, a newer abundance estimate, and complete fisheries observer coverage. Population trends and status of this stock relative to its Optimum Sustainable Population are unknown.

There are key uncertainties in the assessment of the Gulf of Alaska stock of harbor porpoise. This stock likely comprises multiple, smaller stocks based on analogy with harbor porpoise populations that have been the focus of specific studies on stock structure. The most recent surveys were more than 8 years ago and, given the lack of information on population trend, the abundance estimates are not used to calculate an N_{MIN} and the PBR level is undetermined. Several commercial fisheries overlap with the range of this stock and are not observed or have not been observed in a long time; thus, the estimate of commercial fishery mortality and serious injury is expected to be a minimum estimate. Estimates of human-caused mortality and serious injury from stranding data and fisherman self-reports are underestimates because not all animals strand or are self-reported nor are all stranded animals found, reported, or have the cause of death determined.

HABITAT CONCERNS

Harbor porpoise are mostly found in nearshore areas, bays, tidal areas, and river mouths (Dahlheim et al. 2000, Hobbs and Waite 2010). As a result, harbor porpoise are vulnerable to physical modifications of nearshore habitats resulting from urban and industrial development (including waste management and nonpoint source runoff) and activities such as construction of docks and other over-water structures, filling of shallow areas, dredging, and noise (Linnenschmidt et al. 2013).

Algal toxins are a growing concern in Alaska marine food webs, in particular the neurotoxins domoic acid and saxitoxin. While saxitoxin was not detected in harbor porpoise samples collected in Alaska, domoic acid was found in 40% (2 of 5) of the samples and, notably, in maternal transfer to a fetus (Lefebvre et al. 2016).

CITATIONS

Barlow, J., C. W. Oliver, T. D. Jackson, and B. L. Taylor. 1988. Harbor porpoise, *Phocoena phocoena*, abundance estimation for California, Oregon, and Washington: II. Aerial surveys. Fish. Bull., U.S. 86:433-444.

Calambokidis, J., and J. Barlow. 1991. Chlorinated hydrocarbon concentrations and their use for describing population discreteness in harbor porpoises from Washington, Oregon, and California, p. 101-110. *In* J. E. Reynolds III and D. K. Odell (eds.), Proceedings of the Second Marine Mammal Stranding Workshop: 3-5 December 1987, Miami, Florida. U.S. Dep. Commer., NOAA Tech. Rep. NMFS-98.

Calambokidis, J., J. R. Evenson, J. C. Cubbage, S. D. Osmek, D. Rugh, and J. L. Laake. 1993. Calibration of sighting rates of harbor porpoise from aerial surveys. Final Report to the National Marine Mammal Laboratory, AFSC, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115. 55 p.

Castellote, M., K. M. Stafford, A. D. Neff, and W. Lucey. 2015. Acoustic monitoring and prey association for beluga whale, *Delphinapterus leucas*, and harbor porpoise, *Phocoena phocoena*, off two river mouths in Yakutat Bay, Alaska. Mar. Fish. Rev. 77(1):1-10.

Chivers, S. J., A. E. Dizon, P. J. Gearin, and K. M. Robertson. 2002. Small-scale population structure of eastern North Pacific harbor porpoise (*Phocoena phocoena*) indicated by molecular genetic analyses. J. Cetacean Res. Manage. 4(2):111-122.

Christman, C. L., and L. M. Aerts. 2015. Harbor porpoise (*Phocoena phocoena*) sightings from shipboard surveys in the Chukchi Sea during summer and fall, 2008-2014, p. 197. *In* Book of Abstracts, 2015 Alaska Marine Science Symposium, Anchorage, Alaska, January 19-23, 2015.

Dahlheim, M., A. York, R. Towell, J. Waite, and J. Breiwick. 2000. Harbor porpoise (*Phocoena phocoena*) abundance in Alaska: Bristol Bay to Southeast Alaska, 1991-1993. Mar. Mammal Sci. 16:28-45.

Dahlheim, M., P. A. White, and J. Waite. 2009. Cetaceans of Southeast Alaska: distribution and seasonal occurrence. J. Biogeogr. 36(3):410-426.

Dahlheim, M. E, A. N. Zerbini, J. M. Waite, and A. S. Kennedy. 2015. Temporal changes in abundance of harbor porpoise (*Phocoena phocoena*) inhabiting the inland waters of Southeast Alaska. Fish. Bull., U.S. 113(2):242-255. DOI: dx.doi.org/10.7755/FB.113.3.2 .

Gaskin, D. E. 1984. The harbor porpoise *Phocoena phocoena* (L.): regional populations, status, and information on direct and indirect catches. Rep. Int. Whal. Comm. 34:569-586.

Hobbs, R. C., and J. M. Waite. 2010. Abundance of harbor porpoise (*Phocoena phocoena*) in three Alaskan regions, corrected for observer errors due to perception bias and species misidentification, and corrected for animals submerged from view. Fish. Bull., U.S. 108(3):251-267.

Laake, J. L., J. Calambokidis, S. D. Osmek, and D. J. Rugh. 1997. Probability of detecting harbor porpoise from aerial surveys: estimating g(0). J. Wildl. Manage. 61(1):63-75.

Lefebvre, K. A., L. Quakenbush, E. Frame, K. Burek Huntington, G. Sheffield, R. Stimmelmayr, A. Bryan, P. Kendrick, H. Ziel, T. Goldstein, J. A. Snyder, T. Gelatt, F. Gulland, B. Dickerson, and V. Gill. 2016. Prevalence of algal toxins in Alaskan marine mammals foraging in a changing arctic and subarctic environment. Harmful Algae 55:13-24. DOI: dx.doi.org/10.1016/j.hal.2016.01.007 .

Linnenschmidt, M., J. Teilmann, T. Akamatsu, R. Dietz, and L. A. Miller. 2013. Biosonar, dive, and foraging activity of satellite tracked harbor porpoises (*Phocoena phocoena*). Mar. Mammal Sci. 29(2):77-97.

Manly, B. F. J. 2006. Incidental catch and interactions of marine mammals and birds in the Cook Inlet salmon driftnet and setnet fisheries, 1999-2000. Final Report to NMFS Alaska Region. 98 p.

Manly, B. F. J. 2007. Incidental take and interactions of marine mammals and birds in the Kodiak Island salmon set gillnet fishery, 2002 and 2005. Final Report to NMFS Alaska Region. 221 p.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2020.

Parsons, K. M., M. Everett, M. Dahlheim, and L. Park. 2018. Water, water everywhere: environmental DNA can unlock population structure in elusive marine species. Royal Society Open Science 5:180537. DOI: dx.doi.org/10.1098/rsos.180537 .

Rosel, P. E. 1992. Genetic population structure and systematic relationships of some small cetaceans inferred from mitochondrial DNA sequence variation. Ph.D. Dissertation, University of California San Diego. 191 p.

Rosel, P. E., A. E. Dizon, and M. G. Haygood. 1995. Variability of the mitochondrial control region in populations of the harbour porpoise, *Phocoena phocoena*, on inter-oceanic and regional scales. Can. J. Fish. Aquat. Sci. 52:1210-1219.

Rosel, P. E., R. Tiedemann, and M. Walton. 1999. Genetic evidence for limited trans-Atlantic movements of the harbor porpoise *Phocoena phocoena*. Mar. Biol. 133:583-591.

Shelden, K. E. W., B. A. Agler, J. J. Brueggeman, L. A. Cornick, S. G. Speckman, and A. Prevel-Ramos. 2014. Harbor porpoise, *Phocoena phocoena vomerina*, in Cook Inlet, Alaska. Mar. Fish. Rev. 76(1-2):22-50.

Taylor, B. L., P. R. Wade, D. P. DeMaster, and J. Barlow. 1996. Models for management of marine mammals. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/48/SM50). 12 p.

Walton, M. J. 1997. Population structure of harbour porpoises *Phocoena phocoena* in the seas around the UK and adjacent waters. Proc. R. Soc. Lond. B 264:89-94.

Westgate, A. J., and K. A. Tolley. 1999. Geographical differences in organochlorine contaminants in harbour porpoises *Phocoena phocoena* from the western North Atlantic. Mar. Ecol. Prog. Ser. 177:255-268.

Wynne, K. M., D. Hicks, and N. Munro. 1991. 1990 salmon gillnet fisheries observer programs in Prince William Sound and South Unimak Alaska. Annual Report NMFS/NOAA Contract 50ABNF000036. 65 p. Available from NMFS Alaska Region, Office of Marine Mammals, P.O. Box 21668, Juneau, AK 99802.

Wynne, K. M., D. Hicks, and N. Munro. 1992. 1991 marine mammal observer program for the salmon driftnet fishery of Prince William Sound Alaska. Annual Report NMFS/NOAA Contract 50ABNF000036. 53 p. Available from NMFS Alaska Region, Office of Marine Mammals, P.O. Box 21668, Juneau, AK 99802.

Young, N. C., B. J. Delean, V. T. Helker, J. C. Freed, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2014-2018. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-413, 142 p.

Revised 12/30/2020

HARBOR PORPOISE (*Phocoena phocoena*): Bering Sea Stock

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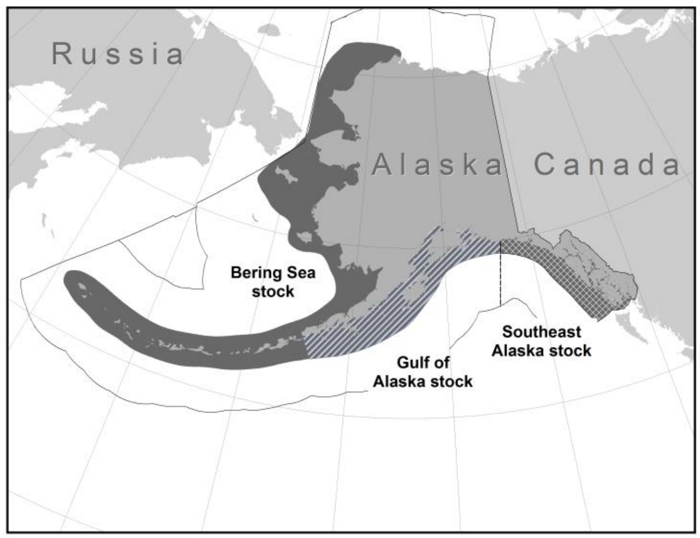

Figure 1. Approximate distribution of harbor porpoise in Alaska waters. The U.S. Exclusive Economic Zone is delineated by a black line.

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Although it is difficult to determine the true stock structure of harbor porpoise populations in the northeast Pacific, from a management standpoint it is prudent to assume that regional populations exist and that they should be managed independently (Rosel et al. 1995, Taylor et al. 1996). Based on the above information, three harbor porpoise stocks in Alaska are currently specified, recognizing that the boundaries of these three stocks are inferred primarily based upon geography or perceived areas of low porpoise density: 1) the Southeast Alaska stock - occurring from Dixon Entrance to Cape Suckling, including inland waters, 2) the Gulf of Alaska stock - occurring from Cape Suckling to Unimak Pass, and 3) the Bering Sea stock - occurring throughout the Aleutian Islands and all waters north of Unimak Pass (Fig. 1). There have been no analyses to assess the validity of these stock designations and research to assess substructure is ongoing only within the Southeast Alaska stock (see the Southeast Alaska harbor porpoise Stock Assessment Report and Parsons et al. 2018).

Harbor porpoise have been sighted during seismic surveys of the Chukchi Sea conducted in the nearshore and offshore waters by the oil and gas industry between July and November from 2006 to 2014 (Funk et al. 2010, 2011; Reiser et al. 2011; Aerts et al. 2013; Christman and Aerts 2015). Harbor porpoise were the third most frequently sighted cetacean species in the Chukchi Sea, after gray and bowhead whales, with most sightings occurring during the September to October monitoring period (Funk et al. 2011, Reiser et al. 2011, Christman and Aerts 2015). Over the 2006 to 2010 industry-sponsored monitoring period, six sightings of 11 harbor porpoise were reported in the Beaufort Sea, suggesting harbor porpoise regularly occur in both the Chukchi and Beaufort seas (Funk et al. 2011).

POPULATION SIZE

In June and July of 1999, an aerial survey covered the waters of Bristol Bay. Two types of corrections were needed for these aerial surveys: one to correct for animals available but not counted because they were missed by the observer (perception bias) and another to correct for porpoise that were submerged and not available at the surface (availability bias). The 1999 survey resulted in an observed abundance estimate for the Bering Sea harbor porpoise stock of 16,289 (coefficient of variation (CV) = 0.13: Hobbs and Waite 2010), which includes the perception bias correction factor (1.337; CV = 0.06) obtained during the survey using an independent belly window observer. Laake et al. (1997) estimated the availability bias correction factor for aerial surveys of harbor porpoise in Puget Sound to be 2.96 (CV = 0.18); the use of this correction factor is preferred to other published correction factors (e.g., Barlow et al. 1988, Calambokidis et al. 1993) because it is an empirical estimate of availability bias. Applying the Laake et al. (1997) correction factor, the corrected abundance estimate is 48,215 porpoise (16,289 × 2.96 = 48,215; CV = 0.22). The estimate for 1999 can be considered conservative for that time period, as the surveyed areas did not include known harbor porpoise range along the Aleutian Island chain, near the Pribilof Islands, or in the waters north of Cape Newenham (approximately 59°N).

Shipboard visual line-transect surveys for cetaceans were conducted on the eastern Bering Sea shelf in association with pollock stock assessment surveys in June and July of 1999, 2000, 2002, 2004, 2008, and 2010 (Moore et al. 2002; Friday et al. 2012, 2013). The entire range of the survey was completed in three of those years (2002, 2008, and 2010) and harbor porpoise abundance estimates were calculated for each of these surveys as 1,971 porpoise (CV = 0.46) for 2002, 4,056 (CV = 0.40) for 2008, and 833 (CV = 0.66) for 2010 (Friday et al. 2013). The abundance estimates provided above assume the probability of detection directly on the trackline to be unity (g(0) = 1). This assumption is typically violated in harbor porpoise surveys because observers tend to miss animals on the survey trackline. Because no estimate of g(0) was computed for the Bering Sea survey in Friday et al. (2013), their abundance estimates were corrected using an averaged estimate of g(0) (weighted by the inverse of the CV) from ship surveys for harbor porpoise in other areas off the U.S. coast (g[0] = 0.71, CV = 0.052: Barlow 1988; Palka 1995, 2000). Using this value for g(0), corrected abundance estimates for harbor porpoise in the Bering Sea are 2,276 porpoise (CV = 0.46) for 2002, 5,713 (CV = 0.40) for 2008, and 1,173 (CV = 0.66) for 2010. The 2008 ship survey estimate is used below to calculate N_{MIN} because the spatial coverage during the year of the most recent estimate (2010) was limited due to poor weather conditions and missed many habitats where harbor porpoise are known to occur in the Bering Sea (e.g., Fig. 7 in Friday et al. 2013).

Minimum Population Estimate

The minimum population estimate (N_{MIN}) for this stock is calculated using Equation 1 from the potential biological removal (PBR) guidelines (NMFS 2016): $N_{MIN} = N/\exp(0.842\times[\ln(1+[CV(N)]^2)]^{½})$. Using the 2008 ship survey partial population estimate (N) of 5,713 and its associated CV of 0.40, N_{MIN} for the Bering Sea stock of harbor porpoise is 4,130. However, this is an underestimate for the entire stock because it is based on a survey that covered only a small portion of the stock's range. Because the survey data are more than 8 years old, N_{MIN} is considered unknown.

Current Population Trend

There is no reliable information on trends in abundance for the Bering Sea stock of harbor porpoise.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate (R_{MAX}) is not available for this stock of harbor porpoise. Until additional data become available, the default cetacean maximum theoretical net productivity rate of 4% will be used (NMFS 2016).

POTENTIAL BIOLOGICAL REMOVAL

PBR is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: $PBR = N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 0.5, the value for cetacean stocks with unknown population status (NMFS 2016). However, the 2016 guidelines for preparing Stock Assessment Reports (NMFS 2016) state that abundance estimates older than 8 years should not be used to calculate PBR due to a decline in confidence in the reliability of an aged abundance estimate. Therefore, the PBR for this stock is considered undetermined.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2014 and 2018 is listed, by marine mammal stock, in Young et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports. The minimum estimated mean annual level of human-caused mortality and serious injury for Bering Sea harbor porpoise between 2014 and 2018 is 0.4 porpoise in subsistence fisheries; however, this estimate is considered a minimum because most of the fisheries likely to interact with this stock of harbor porpoise have never been monitored. Potential threats most likely to result in direct human-caused mortality or serious injury of this stock include entanglement in fishing gear.

Fisheries Information

Information for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is available in Appendix 3 of the Alaska Stock Assessment Reports (observer coverage) and in the NMFS List of Fisheries (LOF) and the fact sheets linked to fishery names in the LOF (observer coverage and reported incidental takes of marine mammals: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-list-fisheries, accessed December 2020).

Harbor porpoise mortality and serious injury is known to occur in gillnet (both drift gillnet and set gillnet) and trawl fisheries. While much of the trawl fleet has observer coverage, there are several gillnet fisheries in the Bering Sea that do not. Given the occurrence of fishery-caused mortality and serious injury of harbor porpoise in other gillnet fisheries in Alaska, it is likely that gillnet fisheries within the range of this stock also incur mortality and serious injury of harbor porpoise.

No mortality or serious injury of Bering Sea harbor porpoise was observed incidental to U.S. federal commercial fisheries between 2014 and 2018. However, a complete estimate of the total mortality and serious injury rate incidental to U.S. commercial fisheries is not available for this stock because of the absence of an observer program for all of the salmon and herring fisheries operating within the range of the stock.

Reports to the NMFS Alaska Region marine mammal stranding network of harbor porpoise entangled in fishing gear or with injuries caused by interactions with gear are another source of mortality and serious injury data (Table 1; Young et al. 2020). In 2018, two harbor porpoise entanglements were reported in the Kuskokwim, Yukon, Norton Sound, Kotzebue subsistence salmon gillnet fishery, resulting in a minimum mean annual mortality and serious injury rate of 0.4 Bering Sea harbor porpoise in this subsistence fishery between 2014 and 2018 (Table 1; Young et al. 2020). This mortality and serious injury estimate results from an actual count of verified human-caused deaths and serious injuries and is a minimum because not all entangled animals strand nor are all stranded animals found, reported, or have the cause of death determined.

Table 1. Summary of incidental mortality and serious injury of Bering Sea harbor porpoise, by year and type, reported to the NMFS Alaska Region marine mammal stranding network between 2014 and 2018 (Young et al. 2020).

Cause of injury	2014	2015	2016	2017	2018	Mean annual mortality
Entangled in Kuskokwim, Yukon, Norton Sound, Kotzebue subsistence salmon gillnet	0	0	0	0	2	0.4
Total in subsistence fisheries						0.4

Alaska Native Subsistence/Harvest Information

Subsistence hunters in Alaska have not been reported to hunt from this stock of harbor porpoise; however, when porpoise are caught incidental to subsistence or commercial fisheries, subsistence hunters may claim the carcass for subsistence use (R. Suydam, North Slope Borough, pers. comm.).

STATUS OF STOCK

Bering Sea harbor porpoise are not designated as depleted under the Marine Mammal Protection Act or listed as threatened or endangered under the Endangered Species Act. The minimum population estimate for this stock is an underestimate for the entire stock because it is based on a survey that covered only a small portion of the stock's range. Because the existing estimates are more than 8 years old, N_{MIN} is unknown and the PBR level is undetermined. Because the PBR is undetermined and most of the fisheries likely to interact with this stock have never been observed, it is unknown if the minimum estimate of the mean annual mortality and serious injury rate (0.4 porpoise from stranding data) in U.S. commercial fisheries can be considered insignificant and approaching a zero mortality and serious injury rate. NMFS considers this stock strategic because the level of mortality and serious injury would likely exceed the PBR level for this stock if we had accurate information on stock structure, a newer abundance estimate, and complete observer coverage. Population trends and status of this stock relative to its Optimum Sustainable Population are unknown.

There are key uncertainties in the assessment of the Bering Sea stock of harbor porpoise. This stock likely comprises multiple, smaller stocks based on analogy with harbor porpoise populations that have been the focus of specific studies on stock structure. The most recent surveys were more than 8 years ago and covered only a small portion of the stock's range, so N_{MIN} is unknown and the PBR level is undetermined. Several commercial fisheries overlap with the range of this stock and most have never been observed; thus, the estimate of commercial fishery mortality and serious injury is expected to be a minimum estimate. Coastal subsistence fisheries will occasionally cause incidental mortality or serious injury of a harbor porpoise; tracking these subsistence takes is challenging because there is no reporting mechanism. Estimates of human-caused mortality and serious injury from stranding data are underestimates because not all animals strand nor are all stranded animals found, reported, or have the cause of death determined.

HABITAT CONCERNS

Harbor porpoise are found over the shelf waters of the southeastern Bering Sea (Dahlheim et al. 2000, Hobbs and Waite 2010). In the nearshore waters of this region, harbor porpoise are vulnerable to physical modifications of nearshore habitats resulting from urban and industrial development (including waste management and nonpoint source runoff) and activities such as construction of docks and other over-water structures, filling of shallow areas, dredging, and noise (Linnenschmidt et al. 2013). Climate change and changes to sea-ice coverage may be opening up new habitats, or resulting in shifts in distribution, as evident by an increase in the number of reported sightings of harbor porpoise in the Chukchi Sea (Funk et al. 2010, 2011). Shipping and noise from oil and gas activities may also be a habitat concern for harbor porpoise, particularly in the Chukchi Sea.

Algal toxins are a growing concern in Alaska marine food webs, in particular the neurotoxins domoic acid and saxitoxin. While saxitoxin was not detected in harbor porpoise samples collected in Alaska, domoic acid was found in 40% (2 of 5) of the samples and, notably, in maternal transfer to a fetus (Lefebvre et al. 2016).

CITATIONS

Aerts, L. A. M., A. E. McFarland, B. H. Watts, K. S. Lomac-MacNair, P. E. Seiser, S. S. Wisdom, A. V. Kirk, and C. A. Schudel. 2013. Marine mammal distribution and abundance in an offshore sub-region of the northeastern Chukchi Sea during the open-water season. Continental Shelf Research 67:116-126. DOI: dx.doi.org/10.1016/j.csr.2013.04.020 .

Barlow, J. 1988. Harbor porpoise, *Phocoena phocoena*, abundance estimation for California, Oregon, and Washington: I. Ship surveys. Fish. Bull., U.S. 86:417-432.

Barlow, J., C. W. Oliver, T. D. Jackson, and B. L. Taylor. 1988. Harbor porpoise, *Phocoena phocoena*, abundance estimation for California, Oregon, and Washington: II. Aerial surveys. Fish. Bull., U.S. 86:433-444.

Calambokidis, J., and J. Barlow. 1991. Chlorinated hydrocarbon concentrations and their use for describing population discreteness in harbor porpoises from Washington, Oregon, and California, p. 101-110. *In* J. E. Reynolds III and D. K. Odell (eds.), Proceedings of the Second Marine Mammal Stranding Workshop: 3-5 December 1987, Miami, Florida. U.S. Dep. Commer., NOAA Tech. Rep. NMFS-98.

Calambokidis, J., J. R. Evenson, J. C. Cubbage, S. D. Osmek, D. Rugh, and J. L. Laake. 1993. Calibration of sighting rates of harbor porpoise from aerial surveys. Final Report to the National Marine Mammal Laboratory, AFSC, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115. 55 p.

Chivers, S. J., A. E. Dizon, P. J. Gearin, and K. M. Robertson. 2002. Small-scale population structure of eastern North Pacific harbor porpoise (*Phocoena phocoena*) indicated by molecular genetic analyses. J. Cetacean Res. Manage. 4(2):111-122.

Christman, C. L., and L. M. Aerts. 2015. Harbor porpoise (*Phocoena phocoena*) sightings from shipboard surveys in the Chukchi Sea during summer and fall, 2008-2014, p. 197. *In* Book of Abstracts, 2015 Alaska Marine Science Symposium, Anchorage, Alaska, January 19-23, 2015.

Dahlheim, M., A. York, R. Towell, J. Waite, and J. Breiwick. 2000. Harbor porpoise (*Phocoena phocoena*) abundance in Alaska: Bristol Bay to Southeast Alaska, 1991-1993. Mar. Mammal Sci. 16:28-45.

Dahlheim, M., P. A. White, and J. Waite. 2009. Cetaceans of Southeast Alaska: distribution and seasonal occurrence. J. Biogeogr. 36(3):410-426.

Dahlheim, M. E., A. N. Zerbini, J. M. Waite, and A. S. Kennedy. 2015. Temporal changes in abundance of harbor porpoise (*Phocoena phocoena*) inhabiting the inland waters of Southeast Alaska. Fish. Bull., U.S. 113(3):242-255.

Friday, N. A., J. M. Waite, A. N. Zerbini, and S. E. Moore. 2012. Cetacean distribution and abundance in relation to oceanographic domains on the eastern Bering Sea shelf: 1999-2004. Deep-Sea Res. II 65-70:260-272.

Friday, N. A., J. M. Waite, A. N. Zerbini, S. E. Moore, and P. J. Clapham. 2013. Cetacean distribution and abundance in relation to oceanographic domains on the eastern Bering Sea shelf, June and July of 2002, 2008 and 2010. Deep-Sea Res. II 94:244-256.

Funk, D. W., D. S. Ireland, R. Rodrigues, and W. R. Koski (eds.). 2010. Joint monitoring program in the Chukchi and Beaufort seas, open-water seasons, 2006–2008. LGL Alaska Report P1050-3, Report from LGL Alaska Research Associates, Inc., LGL, Ltd., Greeneridge Sciences, Inc., and JASCO Research, Ltd., for Shell Offshore, Inc. and Other Industry Contributors, and National Marine Fisheries Service, U.S. Fish and Wildlife Service. 499 p. + appendices.

Funk, D. W., C. M. Reiser, D. S. Ireland, R. Rodrigues, and W. R. Koski (eds.). 2011. Joint monitoring program in the Chukchi and Beaufort seas, 2006–2010. LGL Alaska Draft Report P1213-1, Report from LGL Alaska Research Associates, Inc., LGL, Ltd., Greeneridge Sciences, Inc., and JASCO Research, Ltd., for Shell Offshore, Inc., and Other Industry Contributors, and National Marine Fisheries Service, U.S. Fish and Wildlife Service. 592 p. + appendices.

Gaskin, D. E. 1984. The harbor porpoise *Phocoena phocoena* (L.): regional populations, status, and information on direct and indirect catches. Rep. Int. Whal. Comm. 34:569-586.

Hobbs, R. C., and J. M. Waite. 2010. Abundance of harbor porpoise (*Phocoena phocoena*) in three Alaskan regions, corrected for observer errors due to perception bias and species misidentification, and corrected for animals submerged from view. Fish. Bull., U.S. 108(3):251-267.

Laake, J. L., J. Calambokidis, S. D. Osmek, and D. J. Rugh. 1997. Probability of detecting harbor porpoise from aerial surveys: estimating g(0). J. Wildl. Manage. 61(1):63-75.

Lefebvre, K. A., L. Quakenbush, E. Frame, K. Burek Huntington, G. Sheffield, R. Stimmelmayr, A. Bryan, P. Kendrick, H. Ziel, T. Goldstein, J. A. Snyder, T. Gelatt, F. Gulland, B. Dickerson, and V. Gill. 2016. Prevalence of algal toxins in Alaskan marine mammals foraging in a changing arctic and subarctic environment. Harmful Algae 55:13-24. DOI: dx.doi.org/10.1016/j.hal.2016.01.007 .

Linnenschmidt, M., J. Teilmann, T. Akamatsu, R. Dietz, and L. A. Miller. 2013. Biosonar, dive, and foraging activity of satellite tracked harbor porpoises (*Phocoena phocoena*). Mar. Mammal Sci. 29(2):77-97.

Moore, S. E., J. M. Waite, N. A. Friday, and T. Honkalehto. 2002. Cetacean distribution and relative abundance on the central-eastern and the southeastern Bering Sea shelf with reference to oceanographic domains. Prog. Oceanogr. 55:249-261.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2020.

Palka, D. 1995. Abundance estimate of the Gulf of Maine harbor porpoise. Rep. Int. Whal. Comm. (Special Issue 16):27-50.

Palka, D. 2000. Abundance of the Gulf of Maine/Bay of Fundy harbor porpoise based on shipboard and aerial surveys during 1999. Northeast Fisheries Science Center Reference Document 00-07. 29 p. Available online: https://repository.library.noaa.gov/view/noaa/3290 . Accessed December 2020.

Parsons, K. M., M. Everett, M. Dahlheim, and L. Park. 2018. Water, water everywhere: environmental DNA can unlock population structure in elusive marine species. Royal Society Open Science 5:180537. DOI: dx.doi.org/10.1098/rsos.180537 .

Reiser, C. M., D. W. Funk, R. Rodrigues, and D. Hannay (eds.). 2011. Marine mammal monitoring and mitigation during marine geophysical surveys by Shell Offshore, Inc., in the Alaskan Chukchi and Beaufort seas, July–October 2010: 90-day report. LGL Report P1171E–1. Report from LGL Alaska Research Associates, Inc., Anchorage, AK, and JASCO Applied Sciences, Victoria, BC, for Shell Offshore, Inc., Houston, TX, National Marine Fisheries Service, Silver Spring, MD, and U.S. Fish and Wildlife Service, Anchorage, AK. 240 p. + appendices.

Rosel, P. E. 1992. Genetic population structure and systematic relationships of some small cetaceans inferred from mitochondrial DNA sequence variation. Ph.D. Dissertation, University of California San Diego. 191 p.

Rosel, P. E., A. E. Dizon, and M. G. Haygood. 1995. Variability of the mitochondrial control region in populations of the harbour porpoise, *Phocoena phocoena*, on inter-oceanic and regional scales. Can. J. Fish. Aquat. Sci. 52:1210-1219.

Rosel, P. E., R. Tiedemann, and M. Walton. 1999. Genetic evidence for limited trans-Atlantic movements of the harbor porpoise *Phocoena phocoena*. Mar. Biol. 133:583-591.

Shelden, K. E. W., B. A. Agler, J. J. Brueggeman, L. A. Cornick, S. G. Speckman, and A. Prevel-Ramos. 2014. Harbor porpoise, *Phocoena phocoena vomerina*, in Cook Inlet, Alaska. Mar. Fish. Rev. 76(1-2):22-50.

Taylor, B. L., P. R. Wade, D. P. DeMaster, and J. Barlow. 1996. Models for management of marine mammals. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/48/SM50). 12 p.

Walton, M. J. 1997. Population structure of harbour porpoises *Phocoena phocoena* in the seas around the UK and adjacent waters. Proc. R. Soc. Lond. B 264:89-94.

Westgate, A. J., and K. A. Tolley. 1999. Geographical differences in organochlorine contaminants in harbour porpoises *Phocoena phocoena* from the western North Atlantic. Mar. Ecol. Prog. Ser. 177:255-268.

Young, N. C., B. J. Delean, V. T. Helker, J. C. Freed, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2014-2018. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-413, 142 p.

DALL'S PORPOISE (*Phocoenoides dalli*): Alaska Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Dall's porpoise are widely distributed across the entire North Pacific Ocean (Fig. 1). They are found over the continental shelf adjacent to the slope and over deep (2,500+ m) oceanic waters (Hall 1979). They have been sighted throughout the North Pacific as far north as 65°N (Buckland et al. 1993) and as far south as 28°N in the eastern North Pacific (Leatherwood and Fielding 1974). The only apparent distribution gaps in Alaska waters are upper Cook Inlet and the shallow eastern flats of the Bering Sea. Throughout most of the eastern North Pacific they are present during all months of the year, although there may be seasonal onshore-offshore movements along the west coast of the continental U.S. (Loeb 1972, Leatherwood and Fielding 1974) and winter movements of populations out of areas with ice such as Prince William Sound (Hall 1979).

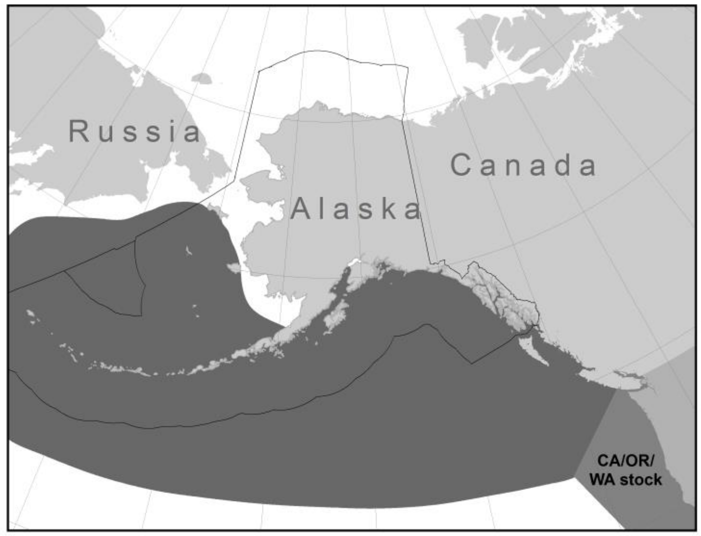

Figure 1. Approximate distribution of Dall's porpoise in the eastern North Pacific Ocean (dark shaded area). The Alaska stock is defined as the portion of the distribution in Alaska waters. The U.S. Exclusive Economic Zone is delineated by the solid black line.

Surveys on the eastern Bering Sea shelf and slope to the 1,000 m isobath in 1999, 2000, 2002, 2004, 2008, and 2010 provided information about the distribution and relative abundance of Dall's porpoise in that area (Moore et al. 2002; Friday et al. 2012, 2013). Dall's porpoise were sighted on the shelf and slope in waters deeper than 100 m in 2002, 2008, and 2010 with greater densities at the shelf break than in shallower waters (Friday et al. 2013). Ship surveys in the northeast Gulf of Alaska in 2013 and 2015 recorded Dall's porpoise throughout the study area, including the continental shelf, the slope, offshore waters, and around seamounts. Higher densities were observed on the shelf and slope (Rone et al. 2017).

The following information was considered in classifying stock structure based on the Dizon et al. (1992) phylogeographic approach: 1) Distributional data: geographic distribution continuous; 2) Population response data: differential timing of reproduction between the Bering Sea and western North Pacific; 3) Phenotypic data: unknown; and 4) Genotypic data: unknown. The stock structure of eastern North Pacific Dall's porpoise is not adequately understood at this time; however, based on patterns of stock differentiation in the western North Pacific, where they have been more intensively studied, it is expected that separate stocks will emerge when data become available (Perrin and Brownell 1994). Based primarily on the population response data (Jones et al. 1986) and preliminary genetic analyses (Winans and Jones 1988), a delineation between Bering Sea and western North Pacific stocks has been recognized. However, similar data are not available for the eastern North Pacific; thus, one stock of Dall's porpoise is currently recognized in Alaska waters. Dall's porpoise along the west coast of the continental U.S. from California to Washington comprise a separate stock and are reported in the Stock Assessment Reports for the U.S. Pacific Region.

POPULATION SIZE

Data collected from vessel surveys, performed by both U.S. fishery observers and U.S. researchers from 1987 to 1991, were analyzed to provide population estimates of Dall's porpoise throughout the North Pacific and the Bering Sea (Hobbs and Lerczak 1993). The quality of data used in analyses was determined by the procedures recommended by Boucher and Boaz (1989). Survey effort was not well distributed throughout the U.S. Exclusive Economic Zone (EEZ) in Alaska and, as a result, Bristol Bay and the northern Bering Sea received little survey effort. Only three sightings were reported between 1987 and 1991 in this area by Hobbs and Lerczak (1993), resulting in an estimate of 9,000 porpoise (CV = 0.91). In the U.S. EEZ north and south of the Aleutian Islands, Hobbs and Lerczak (1993) reported an estimated abundance of 302,000 porpoise (CV = 0.11), whereas, for the Gulf of Alaska EEZ, they reported 106,000 (CV = 0.20). Combining these three estimates (9,000 + 302,000 + 106,000) results in a total abundance estimate of 417,000 (CV = 0.097) for the Alaska stock of Dall's porpoise. Turnock and Quinn (1991) estimate that abundance estimates of Dall's porpoise are inflated by as much as five times because of vessel attraction behavior. Therefore, a corrected population estimate from 1987-1991 is 83,400 (417,000 × 0.2) for this stock. Because surveys are more than 8 years old, there are no reliable abundance estimates for the entire Alaska stock of Dall's porpoise.

Sighting surveys for cetaceans were conducted during NMFS pollock stock assessment surveys in 1999, 2000, 2002, 2004, 2008, and 2010 on the eastern Bering Sea shelf (Moore et al. 2002; Friday et al. 2012, 2013). The entire study area of the survey, which corresponded to only a fraction of the range of the Alaska stock, was fully covered in three of those years (2002, 2008, and 2010). Dall's porpoise estimates were calculated for each of these surveys (Friday et al. 2013). The abundance estimates were 35,303 porpoise (CV = 0.53) in 2002, 14,543 (CV = 0.32) in 2008, and 11,143 (CV = 0.32) in 2010. Although the 2010 estimate is the lowest of the three years, it is not statistically different from the 2002 and 2008 estimates (Friday et al. 2013).

Vessel surveys were carried out in and around a Navy Maritime Activity/Training Area in the northwestern Gulf of Alaska to document abundance and density of cetaceans in 2013 and 2015 (Rone et al. 2017). The surveys covered different, but overlapping, areas in the two years and estimated Dall's porpoise abundance as 15,432 (CV = 0.28) in 2013 and 13,110 (CV = 0.22) in 2015.

Estimates of abundance for the NMFS pollock stock assessment surveys in the Bering Sea and the 2013/2015 vessel surveys in the Gulf of Alaska did not cover the whole range of the stock and were not corrected for animals missed on the trackline (perception bias) or for animals submerged when the ship passed (availability bias). These estimates are also uncorrected for potential biases from responsive movements (ship attraction), which is known to result in severe positive bias when calculating abundance of Dall's porpoise (Turnock and Quinn 1991). Therefore, these estimates are not used as minimum population estimates.

Minimum Population Estimate

The minimum population estimate (N_{MIN}) for this stock is calculated using Equation 1 from the potential biological removal (PBR) guidelines (Wade and Angliss 1997): $N_{MIN} = N/\exp(0.842\times[\ln(1+[CV(N)]^2)]^{½})$. However, because the abundance estimate for the entire stock is based on data older than 8 years, the N_{MIN} is considered unknown.

Current Population Trend

There is no reliable information on trends in abundance for the Alaska stock of Dall's porpoise.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate (R_{MAX}) is not available for the Alaska stock of Dall's porpoise. Until additional data become available, the cetacean maximum theoretical net productivity rate of 4% will be used (Wade and Angliss 1997). However, based on life-history analyses by Ferrero and Walker (1999), Dall's porpoise reproductive strategy is not consistent with the delphinid pattern on which the default maximum theoretical net productivity rate for cetaceans is based. In contrast to the delphinids, Dall's porpoise mature earlier and reproduce annually which suggests that a higher R_{MAX} may be warranted.

POTENTIAL BIOLOGICAL REMOVAL

PBR is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: PBR = $N_{MIN} \times 0.5R_{MAX} \times F_R$. However, the 2016 guidelines for preparing Stock Assessment Reports (NMFS 2016) state that abundance estimates older than 8 years should not be used to calculate PBR due to a decline in confidence in the reliability of an aged abundance estimate. Therefore, the PBR for this stock is considered undetermined.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals in 2012-2016 is listed, by marine mammal stock, in Helker et al. (in press); however, only the mortality and serious injury data are included in the Stock Assessment Reports. The total estimated annual level of human-caused mortality and serious injury for the Alaska stock of Dall's porpoise in 2012-2016 is 38 Dall's porpoise in U.S. commercial fisheries (37 from observer data and 0.6 from fisherman self-reports). This estimate is considered a minimum because not all of the salmon and herring fisheries operating within the range of this stock have been observed. Potential threats most likely to result in direct human-caused mortality or serious injury of this stock include entanglement in fishing gear.

Fisheries Information

Information (including observer programs, observer coverage, and observed incidental takes of marine mammals) for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is presented in Appendices 3-6 of the Alaska Stock Assessment Reports.

No mortality or serious injury of the Alaska stock of Dall's porpoise was observed incidental to federally-managed U.S. commercial fisheries in 2012-2016 (Breiwick 2013; MML, unpubl. data).

The state-managed Alaska Peninsula/Aleutian Islands salmon drift gillnet fishery was monitored by Alaska Marine Mammal Observer Program (AMMOP) observers in 1990 (Wynne et al. 1991). One Dall's porpoise mortality was observed, which extrapolated to an annual (total) incidental mortality and serious injury rate of 28 Dall's porpoise (Table 1). Although these observer data are dated, they are considered the best available data on mortality and serious injury levels in this fishery.

In 2012 and 2013, the AMMOP placed observers on independent vessels in the state-managed Southeast Alaska salmon drift gillnet fishery to assess mortality and serious injury of marine mammals. Areas around and adjacent to Wrangell and Zarembo Islands (ADF&G Districts 6, 7, and 8) were observed during the 2012-2013 program (Manly 2015). In 2012, one Dall's porpoise was seriously injured. Based on the one observed serious injury, 18 serious injuries were estimated for Districts 6, 7, and 8 in 2012, resulting in an estimated mean annual mortality and serious injury rate of 9 Dall's porpoise in 2012-2013 (Table 1). Since these three districts represent only a portion of the overall fishing effort in this fishery, we expect this to be a minimum estimate of mortality for the fishery. Note that the AMMOP has not observed the Southeast Alaska salmon drift gillnet fishery in the other districts; additionally, NMFS has not observed several other gillnet fisheries that are known to interact with this stock, therefore, the total estimated mortality and serious injury is unavailable. Combining the estimates from the Alaska Peninsula/Aleutian Islands salmon drift gillnet fishery (28) and the Southeast Alaska salmon drift gillnet fishery (9) results in an estimated average annual mortality and serious injury rate of 37 Dall's porpoise from this stock.

Table 1. Summary of incidental mortality and serious injury of the Alaska stock of Dall's porpoise due to U.S. commercial fisheries in 2012-2016 (or the most recent data available) and calculation of the mean annual mortality and serious injury rate (Wynne et al. 1991; Breiwick 2013; Manly 2015; MML, unpubl. data). Methods for calculating percent observer coverage are described in Appendix 6 of the Alaska Stock Assessment Reports.

Fishery name	Years	Data type	Percent observer coverage	Observed mortality	Estimated mortality	Mean estimated annual mortality
Southeast Alaska salmon drift gillnet (Districts 6, 7, 8)	2012 2013	obs data	6.4 6.6	1 0	18 0	9 (CV = 1.0)
Alaska Peninsula/Aleutian Is. salmon drift gillnet	1990	obs data	4	1	28	28 (CV = 0.585)
Minimum total estimated annual mortality						37 (CV = 0.505)

Mortality and serious injury of Dall's porpoise due to entanglements in Prince William Sound commercial salmon drift gillnet (1 in 2013), Southeast Alaska commercial salmon drift gillnet (1 in 2014 in District 15C), and Kodiak Island commercial salmon purse seine gear (1 in 2013) was reported by Marine Mammal Authorization Program (MMAP) fisherman self-reports in 2012-2016 (Table 2; Helker et al. in press). Because observer data are not available for these fisheries, this mortality and serious injury is used to calculate mean annual mortality and serious injury rates of 0.2 Dall's porpoise for each of these fisheries (Table 2). These mortality and serious injury estimates result from an actual count of verified human-caused deaths and serious injuries and are minimums because not all entangled animals strand or are self-reported nor are all stranded animals found, reported, or have the cause of death determined.

Table 2. Summary of Alaska Dall's porpoise mortality and serious injury, by year and type, reported to the NMFS Alaska Region marine mammal stranding network and by Marine Mammal Authorization Program (MMAP) fisherman self-reports in 2012-2016 (Helker et al. in press). Only cases of serious injury were recorded in this table; animals with non-serious injuries have been excluded.

Cause of injury	**2012**	**2013**	**2014**	**2015**	**2016**	**Mean annual mortality**
Entangled in Prince William Sound commercial salmon drift gillnet	0	1[a]	0	0	0	0.2
Entangled in Southeast Alaska commercial salmon drift gillnet (District 15C)	0	0	1[a]	0	0	0.2
Entangled in Kodiak Island commercial salmon purse seine gear	0	1[a]	0	0	0	0.2
Total in commercial fisheries						0.6

[a]MMAP fisherman self-report.

A complete estimate of the total mortality and serious injury incidental to U.S. commercial fisheries is unavailable for this stock because not all of the salmon and herring fisheries operating within the range of this stock have been observed. Based on observed mortality and serious injury in two commercial fisheries (Table 1) and by MMAP fisherman self-reports (Table 2), the minimum estimated mean annual mortality and serious injury rate incidental to commercial fisheries in 2012-2016 is 38 Dall's porpoise from this stock.

Alaska Native Subsistence/Harvest Information

There are no reports of subsistence take of Dall's porpoise in Alaska.

STATUS OF STOCK

Dall's porpoise are not designated as depleted under the Marine Mammal Protection Act or listed as threatened or endangered under the Endangered Species Act. The minimum abundance estimate for this stock is unknown because the most recent abundance estimate is more than 8 years old and so the PBR level is considered undetermined. Because the PBR is undetermined and fisheries observer coverage is limited and aged, it is unknown if the minimum estimate of the mean annual mortality and serious injury rate (38 porpoise) in U.S. commercial fisheries can be considered insignificant and approaching zero mortality and serious injury rate. The Alaska stock of Dall's porpoise is not classified as a strategic stock. Population trends and status of this stock relative to its Optimum Sustainable Population are unknown.

There are key uncertainties in the assessment of the Alaska stock of Dall's porpoise. The most recent surveys of the entire range of this stock were more than 8 years ago, so the related abundance estimates are not used to calculate an N_{MIN} and the PBR level is undetermined. There is no information on population trend. Several commercial fisheries overlap with the range of this stock and are not observed or have not been observed in a long time; thus, the estimate of commercial fishery mortality and serious injury is expected to be a minimum estimate. Estimates of human-caused mortality and serious injury from stranding data and fisherman self-reports are underestimates because not all animals strand or are self-reported nor are all stranded animals found, reported, or have the cause of death determined.

HABITAT CONCERNS

While the majority of Dall's porpoise are found throughout the North Pacific, there are also significant numbers found in shelf break and deeper nearshore areas. Thus, they are subject to a variety of habitat impacts. Of particular concern are nearshore areas, bays, channels, and inlets where some Dall's porpoise are vulnerable to physical modifications of nearshore habitats and noise (Linnenschmidt et al. 2013). Climate change and changes to sea-ice coverage may be opening up new habitats, or resulting in shifts in habitat, as evident by an increase in the number of reported sightings of Dall's porpoise in the Chukchi Sea (Funk et al. 2010, 2011). Shipping and noise from oil and gas activities may also be a habitat concern for Dall's porpoise, particularly in the Chukchi Sea.

CITATIONS

Boucher, G. C., and C. J. Boaz. 1989. Documentation for the marine mammal sightings database of the National Marine Mammal Laboratory. U.S. Dep. Commer., NOAA Tech. Memo. NMFS F/NWC-159, 60 p.

Breiwick, J. M. 2013. North Pacific marine mammal bycatch estimation methodology and results, 2007-2011. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-260, 40 p.

Buckland, S. T., K. L. Cattanach, and R. C. Hobbs. 1993. Abundance estimates of Pacific white-sided dolphin, northern right whale dolphin, Dall's porpoise and northern fur seal in the North Pacific, 1987/90, p. 387-407. *In* W. Shaw, R. L. Burgner, and J. Ito (eds.), Biology, distribution and stock assessment of species caught in the high seas driftnet fisheries in the North Pacific Ocean. International North Pacific Fisheries Commission Symposium; 4-6 November 1991, Tokyo, Japan.

Dizon, A. E., C. Lockyer, W. F. Perrin, D. P. DeMaster, and J. Sisson. 1992. Rethinking the stock concept: a phylogeographic approach. Conserv. Biol. 6:24-36.

Ferrero, R. C., and W. A. Walker. 1999. Age, growth, and reproductive patterns of Dall's porpoise (*Phocoenoides dalli*) in the central North Pacific Ocean. Mar. Mammal Sci. 15(2):273-313.

Friday, N. A., J. M. Waite, A. N. Zerbini, and S. E. Moore. 2012. Cetacean distribution and abundance in relation to oceanographic domains on the eastern Bering Sea shelf: 1999-2004. Deep-Sea Res. II 65-70:260-272.

Friday, N. A., A. N. Zerbini, J. M. Waite, S. E. Moore, and P. J. Clapham. 2013. Cetacean distribution and abundance in relation to oceanographic domains on the eastern Bering Sea shelf in June and July of 2002, 2008, and 2010. Deep-Sea Res. II 94:244-256.

Funk, D. W., D. S. Ireland, R. Rodrigues, and W. R. Koski (eds.). 2010. Joint monitoring program in the Chukchi and Beaufort seas, open-water seasons, 2006–2008. LGL Alaska Report P1050-3, Report from LGL Alaska Research Associates, Inc., LGL, Ltd., Greeneridge Sciences, Inc., and JASCO Research, Ltd., for Shell Offshore, Inc., and Other Industry Contributors, and National Marine Fisheries Service, U.S. Fish and Wildlife Service. 499 p. + appendices.

Funk, D. W., C. M. Reiser, D. S. Ireland, R. Rodrigues, and W. R. Koski (eds.). 2011. Joint monitoring program in the Chukchi and Beaufort seas, 2006–2010. LGL Alaska Draft Report P1213-1, Report from LGL Alaska Research Associates, Inc., LGL, Ltd., Greeneridge Sciences, Inc., and JASCO Research, Ltd., for Shell Offshore, Inc., and Other Industry Contributors, and National Marine Fisheries Service, U.S. Fish and Wildlife Service. 592 p. + appendices.

Hall, J. 1979. A survey of cetaceans of Prince William Sound and adjacent waters - their numbers and seasonal movements. Unpubl. report to Alaska Outer Continental Shelf Environmental Assessment Programs. NOAA OCSEAP Juneau Project Office, Juneau, AK. 37 p.

Helker, V. T., M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. In press. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2012-2016. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-XXX, XXX p.

Hobbs, R. C., and J. A. Lerczak. 1993. Abundance of Pacific white-sided dolphin and Dall's porpoise in Alaska estimated from sightings in the North Pacific Ocean and the Bering Sea during 1987 through 1991. Annual Report to the MMPA Assessment Program, Office of Protected Resources, NMFS, NOAA, 1335 East-West Highway, Silver Spring, MD 20910.

Jones, L. L., J. M. Breiwick, G. C. Boucher, and B. J. Turnock. 1986. Untitled document. Submitted as NOAA-2 in Docket #MMPAH - 1986-01 in Seattle Administrative Building, 1986.

Leatherwood, J. S., and M. R. Fielding. 1974. A survey of distribution and movements of Dall's porpoise, *Phocoenoides dalli*, off southern California and Baja California. Working paper No. 42, FAO, United Nations, ACMRR Meeting, La Jolla, CA.

Linnenschmidt, M., J. Teilmann, T. Akamatsu, R. Dietz, and L. A. Miller. 2013. Biosonar, dive, and foraging activity of satellite tracked harbor porpoises (*Phocoena phocoena*). Mar. Mammal Sci. 29(2):77-97.

Loeb, V. J. 1972. A study of the distribution and feeding habits of the Dall's porpoise in Monterey Bay, CA. MA Thesis, San Jose State University, CA. 62 p.

Manly, B. F. J. 2015. Incidental takes and interactions of marine mammals and birds in districts 6, 7, and 8 of the Southeast Alaska salmon drift gillnet fishery, 2012 and 2013. Final Report to NMFS Alaska Region. 52 p.

Moore, S. E., J. M. Waite, N. A. Friday, and T. Honkalehto. 2002. Distribution and comparative estimates of cetacean abundance on the central and south-eastern Bering Sea shelf with observations on bathymetric and prey associations. Prog. Oceanogr. 55(1-2):249-262.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2018.

Perrin, W. F., and R. L. Brownell, Jr. 1994. A brief review of stock identity in small marine cetaceans in relation to assessment of driftnet mortality in the North Pacific. Rep. Int. Whal. Comm. (Special Issue 15):393-401.

Rone, B. K., A. N. Zerbini, A. B. Douglas, D. W. Weller, and P. J. Clapham. 2017. Abundance and distribution of cetaceans in the Gulf of Alaska. Mar. Biol. 164:23. DOI: dx.doi.org/10.1007/s00227-016-3052-2 .

Turnock, B. J., and T. J. Quinn. 1991. The effect of responsive movement on abundance estimation using line transect sampling. Biometrics 47:701-715.

Wade, P. R., and R. Angliss. 1997. Guidelines for assessing marine mammal stocks: report of the GAMMS Workshop April 3-5, 1996, Seattle, Washington. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-OPR-12, 93 p.

Winans, G. A., and L. L. Jones. 1988. Electrophoretic variability in Dall's porpoise (*Phocoenoides dalli*) in the North Pacific Ocean and Bering Sea. J. Mammal. 69(1):14-21.

Wynne, K. M., D. Hicks, and N. Munro. 1991. 1990 salmon gillnet fisheries observer programs in Prince William Sound and South Unimak Alaska. Annual Report NMFS/NOAA Contract 50ABNF000036. 65 p. Available from NMFS, Alaska Region, Office of Marine Mammals, P.O. Box 21668, Juneau, AK 99802.

SPERM WHALE (*Physeter macrocephalus*): North Pacific Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The sperm whale is one of the most widely distributed marine mammal species, perhaps exceeded in its global range only by the killer whale and humpback whale (Rice 1989). In the North Pacific Ocean, sperm whales were depleted by extensive commercial whaling over a period of more than a hundred years, and the species was the primary target of illegal Soviet whaling in the second half of the 20th century (Ivashchenko et al. 2013, 2014). Systematic illegal catches were also made on a large scale by Japan in both the North Pacific and Antarctic in at least the late 1960s (Ivashchenko and Clapham 2015, Clapham and Ivashchenko 2016).

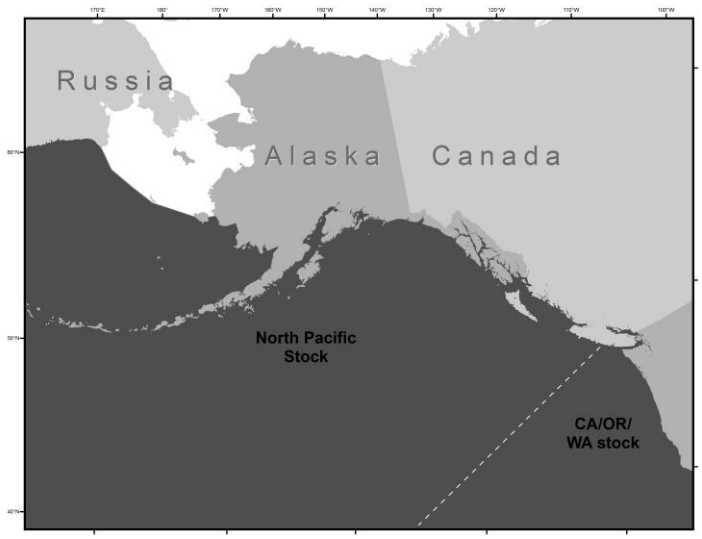

Figure 1. The approximate distribution of sperm whales in the North Pacific Ocean includes deep waters south of 62°N to the equator.

Sperm whales feed primarily on medium-sized to large-sized squids but also consume substantial quantities of large demersal and mesopelagic sharks, skates, and fishes (Rice 1989). In the North Pacific, sperm whales are distributed widely (Fig. 1). Although females and young sperm whales were thought to remain in tropical and temperate waters year-round, Mizroch and Rice (2006) and Ivashchenko et al. (2014) showed that there were extensive catches of female sperm whales above 50°N; Soviet catches of females were made as far north as Olyutorsky Bay (62°N) in the western Bering Sea, as well as in the western Aleutian Islands. Mizroch and Rice (2013) also showed movements by females into the Gulf of Alaska and western Aleutians. During summer, males are found in the Gulf of Alaska, Bering Sea, and waters around the Aleutian Islands (Kasuya and Miyashita 1988, Mizroch and Rice 2013, Ivashchenko et al. 2014). Sighting surveys conducted by the Alaska Fisheries Science Center's Marine Mammal Laboratory (MML) in the summer months between 2001 and 2010 found sperm whales to be the most frequently sighted large cetacean in the coastal waters around the central and western Aleutian Islands (MML, unpubl. data). Acoustic surveys, from fixed autonomous hydrophones, detected the presence of sperm whales year-round in the Gulf of Alaska, although they appear to be approximately two times as common in summer than in winter (Mellinger et al. 2004). This seasonality of detections is consistent with the hypothesis that sperm whales generally move to higher latitudes in summer and to lower latitudes in winter (Whitehead and Arnbom 1987).

Discovery tags implanted in sperm whales in the 1960s could, when recovered from a dead whale, provide useful information on historical movements. Mizroch and Rice (2013) examined 261 Discovery tag recoveries from the days of commercial whaling and found extensive movements from U.S. and Canadian coastal waters into the Gulf of Alaska and Bering Sea/Aleutian Islands region. The U.S. tagged 176 sperm whales from 1962 to 1969 off southern California and northern Baja California (Mizroch and Rice 2013). Seven of those tagged whales were recovered in locations ranging from offshore California, Oregon, and British Columbia to the western Gulf of Alaska. A male sperm whale tagged by Canadian researchers moved from near Vancouver Island, British Columbia, to the Aleutian Islands near Adak. A whale tagged by Soviet researchers moved from coastal Michoacán, mainland Mexico, to a location about 1,300 km offshore of Washington State. Similar extensive movements have also been demonstrated by satellite-tagging studies (Straley et al. 2014). Three adult males satellite tagged off southeastern Alaska moved far south: one to coastal Baja California, one into the north-central Gulf of California, and the third to a location near the Mexico-Guatemala border (Straley et al. 2014).

Mizroch and Rice (2013) analyzed whaling data and found that males and females historically concentrated seasonally along oceanic frontal zones, for example, in the subtropical frontal zone (approximately 28-34°N) and the subarctic frontal zones (approximately 40-43°N). Males also concentrated seasonally near the Aleutian Islands and along the Bering Sea shelf edge. More current research suggests sperm whales are likely relatively nomadic, with movements linked to geographical and temporal variations in the abundance of pelagic squids (Mizroch and Rice 2013). The authors also found no indication from Discovery tag or whaling data to indicate apparent divisions between separate demes or stocks within the North Pacific (Mizroch and Rice 2013). Analysis of Soviet catch data by Ivashchenko et al. (2014) showed broad agreement with these results, although they identified a sharp division at Amchitka Pass in the Aleutians, with mature males to the east and males and family groups to the west. There were four main areas of concentration in the Soviet catches: a large pelagic area (30-50°N) in the eastern North Pacific, including the Gulf of Alaska and western coast of North America; the northeastern and southwestern central North Pacific; and the southern Kuril Islands. Some of the catch distribution was similar to that of 19th-century Yankee whaling catches plotted by Townsend (1935), notably in the "Japan Ground" (in the pelagic western Pacific) and the "Coast of Japan Ground." Many females were caught in Olyutorsky Bay (western Bering Sea) and around the Commander Islands.

More recently, an International Whaling Commission (IWC)-sponsored survey operated by the Government of Japan recorded 284 sightings of sperm whales across the entire North Pacific between 2010 and 2016, but an abundance estimate was not calculated (IWC 2017).

The following information was considered in classifying stock structure based on the Dizon et al. (1992) phylogeographic approach: 1) Distributional data: no apparent discontinuities based on Discovery tag data; 2) Population response data: unknown; 3) Phenotypic data: unknown; and 4) Genotypic data: genetic studies indicate the possibility of a "somewhat" discrete U.S. coastal stock (Mesnick et al. 2011). For management purposes, the IWC recognizes two management units of sperm whales in the North Pacific (eastern and western). However, the IWC has not reviewed its sperm whale stock boundaries in recent years (Donovan 1991). For management purposes, three stocks of sperm whales are currently recognized in U.S. waters: 1) Alaska (North Pacific stock) (Fig. 1); 2) California/Washington/Oregon; and 3) Hawaii. Mizroch and Rice (2013) suggest that this should be reviewed and updated to reflect additional data, but there is insufficient information to propose a reasonable alternative structure. The California/Oregon/Washington and Hawaii sperm whale stocks are reported in the Stock Assessment Reports for the U.S. Pacific Region.

POPULATION SIZE

Current and historical abundance estimates of sperm whales in the North Pacific are based on limited data and are considered unreliable; caution should be exercised in interpreting published estimates. Further, sperm whales are far-ranging and exhibit sex segregation and stock overlap that together make population size estimation difficult. The existing estimates are caveated and do not cover consistent areas, making comparisons difficult. The abundance of sperm whales in the North Pacific was estimated to be 1,260,000 prior to exploitation, which by the late 1970s was thought to have been reduced to 930,000 whales (Rice 1989). Confidence intervals for these estimates do not exist. These estimates include whales from the California/Oregon/Washington stock, for which a separate abundance estimate is available (see the Stock Assessment Reports for the U.S. Pacific Region). Estimates for a large area of the eastern temperate North Pacific were produced from line-transect and acoustic survey data by Barlow and Taylor (2005); the acoustic data produced an estimate of 32,100 sperm whales (coefficient of variation (CV) = 0.36). However, no more recent estimate exists for other areas, including for the central or western North Pacific.

Kato and Miyashita (1998) reported 102,112 sperm whales (CV = 0.155) in the western North Pacific, with the caveat that their estimate is likely positively biased. From surveys in the Gulf of Alaska in 2009 and 2015, Rone et al. (2017) estimated 129 (CV = 0.44) and 345 sperm whales (CV = 0.43) in each year, respectively. These estimates are for a small area that was unlikely to include females and juveniles and they do not account for animals missed on the trackline; therefore, they are not considered reliable estimates.

As the data used in estimating the abundance of sperm whales in the entire North Pacific are more than 8 years old, a reliable estimate of abundance for the entire North Pacific stock is considered unavailable.

Minimum Population Estimate

A minimum population estimate (N_{MIN}) for this stock can be calculated according to Equation 1 from the potential biological removal (PBR) guidelines (NMFS 2016): $N_{MIN} = N/\exp(0.842\times[\ln(1+[CV(N)]^2)]^{½})$. Using the estimate (N) of 345 from surveys in the Gulf of Alaska in 2015 (Rone et al. 2017), and the associated CV(N) of 0.43, results in an N_{MIN} of 244 sperm whales. However, this is an underestimate for the entire stock because it is based on surveys of a small portion of the stock's extensive range and it does not account for animals missed on the trackline or for females and juveniles in tropical and subtropical waters.

Current Population Trend

There is no reliable information on trends in abundance for this stock (Braham 1992).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate (R_{MAX}) is not available for the North Pacific stock of sperm whales. Until additional data become available, the default cetacean maximum theoretical net productivity rate of 4% will be used for this stock (NMFS 2016).

POTENTIAL BIOLOGICAL REMOVAL

Potential biological removal (PBR) is defined as the product of the minimum population estimate (N_{MIN}), one-half the maximum theoretical net productivity rate, and a recovery factor: PBR = $N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 0.1, the value for cetacean stocks that are classified as endangered (NMFS 2016). Using the estimate of 345 (CV = 0.43) from surveys in the Gulf of Alaska in 2015 (Rone et al. 2017), and the associated N_{MIN} of 244, PBR is calculated to be 0.5 sperm whales (244 × 0.02 × 0.1). However, because the N_{MIN} is for only a small portion of the stock's range and does not account for females and juveniles in tropical and subtropical waters, the calculated PBR is not a reliable index for the entire stock.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2014 and 2018 is listed, by marine mammal stock, in Young et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports. Injury events lacking detailed injury information are assigned prorated values following injury determination guidelines described in NMFS (2012). A summary of information used to determine whether an injury was serious or non-serious, as well as a table of prorate values used for large whale reports with incomplete information, is reported in Young et al. (2020). A minimum estimate of the mean annual level of human-caused mortality and serious injury for North Pacific sperm whales between 2014 and 2018 is 3.5 whales: 3.3 in U.S. commercial fisheries and 0.2 due to ship strikes. Sperm whales have been observed depredating both halibut and sablefish longline fisheries in the Gulf of Alaska and this is particularly common in sablefish longline fisheries in the central and eastern Gulf of Alaska; this depredation can lead to mortality or serious injury if hooking or entanglement occurs. Potential threats most likely to result in direct human-caused mortality or serious injury of this stock include entanglement in fishing gear and ship strikes due to increased vessel traffic (from increased shipping in higher latitudes).

Fisheries Information

Information for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is available in Appendix 3 of the Alaska Stock Assessment Reports (observer coverage) and in the NMFS List of Fisheries (LOF) and the fact sheets linked to fishery names in the LOF (observer coverage and reported incidental takes of marine mammals: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-list-fisheries, accessed December 2020).

Between 2014 and 2018, mortality and serious injury of sperm whales was observed in the Bering Sea/Aleutian Islands halibut longline fishery (one serious injury in 2015, prorated at 0.75), the Aleutian Islands sablefish pot fishery (one mortality in 2018), and the Gulf of Alaska sablefish longline fishery (one serious injury in 2016, prorated at 0.75). The mortality and serious injury was extrapolated to fishery-wide estimates when possible, resulting in a minimum estimated mean annual mortality and serious injury rate of 3.3 sperm whales in U.S. commercial fisheries between 2014 and 2018 (Table 1; Breiwick 2013; MML, unpubl. data).

Table 1. Summary of incidental mortality and serious injury of North Pacific sperm whales due to U.S. commercial fisheries between 2014 and 2018 and calculation of the mean annual mortality and serious injury rate (Breiwick 2013; MML, unpubl. data). Methods for calculating percent observer coverage are described in Appendix 3 of the Alaska Stock Assessment Reports. Injury events lacking detailed injury information are assigned prorated values following injury determination guidelines described in NMFS (2012). A summary of information used to determine whether an injury was serious or non-serious, as well as a table of prorate values used for large whale reports with incomplete information, is reported in Young et al. (2020).

Fishery name	Years	Data type	Percent observer coverage	Observed mortality	Estimated mortality (CV)	Mean estimated annual mortality
Bering Sea/Aleutian Is. halibut longline	2014 2015 2016 2017 2018	obs data	11 13 10 6.9 8.2	0 0.75 0 0 0	0 10 (0.98) 0 0 0	2.0 (CV=0.98)
Aleutian Is. sablefish pot	2014 2015 2016 2017 2018	obs data	0 86 88 33 55	0 0 0 0 0 (+1)[a]	0 0 0 0 0 (+1)[b]	0 (+0.2)[c] (CV = N/A)
Gulf of Alaska sablefish longline	2014 2015 2016 2017 2018	obs data	19 20 14 12 9.8	0 0 0.75 0 0	0 0 5.7 (0.93) 0 0	1.1 (CV = 0.93)
Minimum total estimated annual mortality						3.3 (CV = 0.71)

[a]Total mortality and serious injury observed in 2018: 0 whales in sampled hauls + 1 whale in an unsampled haul.
[b]Total estimate of mortality and serious injury in 2018: 0 whales (extrapolated estimate from 0 whales observed in sampled hauls) + 1 whale (1 whale observed in an unsampled haul).
[c]Mean annual mortality and serious injury for fishery: 0 whales (mean of extrapolated estimate from 0 whales observed in sampled hauls) + 0.2 whales (mean of number observed in unsampled hauls).

Alaska Native Subsistence/Harvest Information

Sperm whales have never been reported to be taken by subsistence hunters (Rice 1989).

Other Mortality

Sperm whales were the dominant species killed by the commercial whaling industry as it developed in the North Pacific in the years after World War II (Mizroch and Rice 2006, Ivashchenko et al. 2014). Between 1946 and 1967, most of the sperm whales were caught in waters near Japan and in the Bering Sea/Aleutian Islands region. The Bering Sea/Aleutian Islands catches were dominated by males. After 1967, whalers moved out of the Bering Sea/Aleutian Islands region and began to catch even larger numbers of sperm whales farther south in the North Pacific between 30° and 50°N latitude (Mizroch and Rice 2006: Figs. 7-9). The reported catch of sperm whales taken by commercial whalers operating in the North Pacific between 1912 and 2006 equaled 261,148 sperm whales, of which, 259,120 were taken between 1946 and 1987 (Allison 2012). This value underestimates the actual kill in the North Pacific as a result of under-reporting by U.S.S.R. and Japanese pelagic whaling operations. Berzin (2008) described extreme under-reporting and misreporting of Soviet sperm whale catches from the mid-1960s into the early 1970s, including enormous (and under-reported) whaling pressure on female sperm whales in the latter years of whaling. More recently, Ivashchenko et al. (2013, 2014) estimate that 157,680 sperm whales were killed by the U.S.S.R. in the North Pacific between 1948 and 1979, of which, 25,175 were unreported; the Soviets also extensively misreported the sex and length of catches. In addition, it is known that Japanese land-based whaling operations also misreported the number and sex of sperm whale catches during the post-World War II era (Kasuya 1999), and other studies indicate that falsifications also occurred on a large scale in the Japanese pelagic fishery (Cooke et al. 1983, Ivashchenko and Clapham 2015). The last year that the U.S.S.R. reported catches of sperm whales was in 1979 and the last year that Japan reported substantial catches was in 1987, but Japanese whalers reported catches of 48 sperm whales between 2000 and 2009 (IWC, BIWS catch data, October 2010 version, unpubl.). Although the Soviet data on catches of this species in the North Pacific have now been largely corrected (Ivashchenko et al. 2013), the North Pacific sperm whale data in the IWC's Catch Database (Allison 2012) are known to be incorrect (i.e., too low) because of falsified catch information from both the Japanese coastal and pelagic fisheries (Kasuya 1999, Ivashchenko and Clapham 2015).

Reports to the NMFS Alaska Region marine mammal stranding network are another source of information on sperm whale mortality and serious injury (Table 2; Young et al. 2020). One sperm whale mortality due to a ship strike was reported in 2017, resulting in a mean annual mortality and serious injury rate of 0.2 sperm whales due to ship strikes between 2014 and 2018.

Table 2. Summary of mortality and serious injury of North Pacific sperm whales, by year and type, reported to the NMFS Alaska Region marine mammal stranding network between 2014 and 2018 (Young et al. 2020).

Cause of Injury	**2014**	**2015**	**2016**	**2017**	**2018**	**Mean annual mortality**
Ship strike	0	0	0	1	0	0.2
Total due to ship strikes						0.2

Other Issues

NMFS observers aboard longline vessels targeting both sablefish and halibut have documented sperm whales feeding off longline gear in the Gulf of Alaska (Hill and Mitchell 1998, Hill et al. 1999, Perez 2006, Sigler et al. 2008). Fishery observers recorded several instances between 1995 and 1997 in which sperm whales were deterred by fishermen (i.e., throwing seal bombs in the water).

Annual longline surveys have been recording sperm whale depredation on catch since 1998 (Hanselman et al. 2008). Sperm whale depredation in the sablefish longline fishery is widespread in the central and eastern Gulf of Alaska but rarely observed in the Bering Sea; interaction rates are increasing significantly in the East Yakutat/Southeast Alaska and Central Gulf management areas (Hanselman et al. 2018). More recent research suggests that sperm whales impacted catch rates at a more significant rate than earlier studies suggested (Straley et al. 2005, Sigler et al. 2008), and sperm whales are estimated to reduce commercial fishery and NMFS annual longline survey catch rates by approximately 15% - 26% (Peterson and Hanselman 2017, Hanselman et al. 2018).

STATUS OF STOCK

Sperm whales are listed as endangered under the Endangered Species Act of 1973 and, therefore, designated as depleted under the MMPA. As a result, this stock is classified as a strategic stock. However, on the basis of total abundance, current distribution, and regulatory measures that are in place, it is unlikely that this stock is in danger of extinction (Braham 1992). Reliable estimates of the minimum population, population trends, PBR, and status of the stock relative to its Optimum Sustainable Population are not available. A minimum estimate of the mean annual level of human-caused mortality and serious injury is 3.5 whales. The minimum estimate of the mean annual U.S. commercial fishery-related mortality and serious injury rate (3.3 whales) is more than 10% of the PBR (10% of PBR = 0.05) calculated from the 2015 abundance estimate (Rone et al. 2017) for a small portion of the stock's range. However, because the calculated PBR level is based on an N_{MIN} which is known to be an underestimate of the abundance of the population, the PBR level is considered unreliable.

There are key uncertainties in the assessment of the North Pacific stock of sperm whales. There is little current information about the broad-scale distribution of sperm whales in Alaska waters, and there is no current abundance estimate, N_{MIN}, PBR level, or trend in abundance for the entire stock.

HABITAT CONCERNS

Potential habitat concerns for this stock include elevated levels of sound from anthropogenic sources (e.g., shipping, military exercises), possible changes in prey distribution and quality with climate change, entanglement in fishing gear, ship strikes due to increased vessel traffic (e.g., from increased shipping in higher latitudes), and oil and gas activities.

CITATIONS

Allison, C. 2012. IWC Catch Database, version 5.3 (25 October 2012). Available from International Whaling Commission, Cambridge, UK.

Barlow, J., and B. L. Taylor. 2005. Estimates of sperm whale abundance in the northeastern temperate Pacific from a combined acoustic and visual survey. Mar. Mammal Sci. 21:429-445.

Berzin, A. A. 2008. The truth about Soviet whaling: a memoir. Mar. Fish. Rev. 70(2):4-59.

Braham, H. 1992. Endangered whales: status update. Working document presented at A Workshop on the Status of California Cetacean Stocks (SOCCS/14). 35 p. + tables. Available from Marine Mammal Laboratory, AFSC, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Breiwick, J. M. 2013. North Pacific marine mammal bycatch estimation methodology and results, 2007-2011. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-260, 40 p.

Clapham, P. J., and Y. V. Ivashchenko. 2016. Stretching the truth: length data highlight extensive falsification of Japanese sperm whale catch statistics in the Southern Hemisphere. Royal Society Open Science 3:160506. DOI: dx.doi.org/10.1098/rsos.160506 .

Cooke, J. G., W. K. de la Mare, and J. R. Beddington. 1983. Some aspects of the reliability of the length data for the western North Pacific stock of sperm whales. Rep. Int. Whal. Comm. 33:265-267.

Dizon, A. E., C. Lockyer, W. F. Perrin, D. P. DeMaster, and J. Sisson. 1992. Rethinking the stock concept: a phylogeographic approach. Conserv. Biol. 6:24-36.

Donovan, G. P. 1991. A review of IWC stock boundaries. Rep. Int. Whal. Comm. (Special Issue 13):39-68.

Hanselman, D. H., C. R. Lunsford, J. T. Fujioka, and C. J. Rodgveller. 2008. Assessment of the sablefish stock in Alaska, Section 3, p. 303-420. *In* Stock assessment and fishery evaluation report for the groundfish resources of the Gulf of Alaska. North Pacific Fisheries Management Council, Anchorage, AK.

Hanselman, D. H., B. J. Pyperb, and M. J. Peterson. 2018. Sperm whale depredation on longline surveys and implications for the assessment of Alaska sablefish. Fish. Res. 200:75-83. DOI: dx.doi.org/10.1016/j.fishres.2017.12.017 .

Hill, P. S., and E. Mitchell. 1998. Sperm whale interactions with longline vessels in Alaska waters during 1997. Unpubl. doc. submitted to Fish. Bull., U.S. Available from Marine Mammal Laboratory, AFSC, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Hill, P. S., J. L. Laake, and E. Mitchell. 1999. Results of a pilot program to document interactions between sperm whales and longline vessels in Alaska waters. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-108, 42 p.

International Whaling Commission (IWC). 2017. Report of the planning meeting for the 2017 IWC-POWER cruise in the North Pacific with initial discussions for the 2018 and 2019 cruises. Document SC/67a/Rep 01. Available from International Whaling Commission, Cambridge, UK.

Ivashchenko, Y. V., and P. J. Clapham. 2015. What's the catch? Validity of whaling data for Japanese catches of sperm whales in the North Pacific. Royal Society Open Science 2:150177. DOI: dx.doi.org/10.1098/rsos.150177 .

Ivashchenko, Y. V., P. J. Clapham, and R. L. Brownell, Jr. 2013. Soviet catches of whales in the North Pacific: revised totals. J. Cetacean Res. Manage. 13(1):59-71.

Ivashchenko, Y. V., R. L. Brownell, Jr., and P. J. Clapham. 2014. Distribution of Soviet catches of sperm whales *Physeter macrocephalus* in the North Pacific. Endang. Species Res. 25:249-263.

Kasuya, T. 1999. Examination of the reliability of catch statistics in the Japanese coastal sperm whale fishery. J. Cetacean Res. Manage. 1:109-122.

Kasuya T., and T. Miyashita. 1988. Distribution of sperm whale stocks in the North Pacific. Sci. Rep. Whales Res. Inst. 39: 31-75.

Kato, H., and T. Miyashita. 1998. Current status of North Pacific sperm whales and its preliminary abundance estimates. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/50/CAWS/52). 6 p.

Mellinger, D. K., K. M. Stafford, and C. G. Fox. 2004. Seasonal occurrence of sperm whale (*Physeter macrocephalus*) sounds in the Gulf of Alaska, 1999-2001. Mar. Mammal Sci. 20(1):48-62.

Mesnick, S. L., B. L. Taylor, F. I. Archer, K. K. Martien, S. Escorza Treviño, B. L. Hancock-Hanser, S. C. Moreno Medina, V. L. Pease, K. M. Robertson, J. M. Straley, R. W. Baird, J. Calambokidis, G. S. Schorr, P. Wade, V. Burkanov, C. R. Lunsford, L. Rendell, and P. A. Morin. 2011. Sperm whale population structure in the eastern and central North Pacific inferred by the use of single nucleotide polymorphisms, microsatellites and mitochondrial DNA. Mol. Ecol. Res. 11(Suppl. 1):278-298.

Mizroch, S. A., and D. W. Rice. 2006. Have North Pacific killer whales switched prey species in response to depletion of the great whale populations? Mar. Ecol. Prog. Ser. 310:235-246.

Mizroch, S. A., and D. W. Rice. 2013. Ocean nomads: distribution and movements of sperm whales in the North Pacific shown by whaling data and Discovery marks. Mar. Mammal Sci. 29(2):E136-E165. DOI: dx.doi.org/10.1111/j.1748-7692.2012.00601.x .

National Marine Fisheries Service (NMFS). 2012. Process for distinguishing serious from non-serious injury of marine mammals. 42 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-policies-guidance-and-regulations . Accessed December 2020.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2020.

Perez, M. A. 2006. Analysis of marine mammal bycatch data from the trawl, longline, and pot groundfish fisheries of Alaska, 1998-2004, defined by geographic area, gear type, and target groundfish catch species. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-167, 194 p.

Peterson, M. J., and D. Hanselman. 2017. Sablefish mortality associated with whale depredation in Alaska. ICES J. Mar. Sci. 74 (5):1382-1394. DOI: dx.doi.org/10.1093/icesjms/fsw239 .

Rice, D. W. 1989. Sperm whale, *Physeter macrocephalus*, p. 177-233. *In* S. H. Ridgway and R. Harrison (eds.), Handbook of Marine Mammals. Vol. 4. River Dolphins and the Larger Toothed Whales. Academic Press, New York.

Rone, B. K., A. N. Zerbini, A. B. Douglas, D. W. Weller, and P. J. Clapham. 2017. Abundance and distribution of cetaceans in the Gulf of Alaska. Mar. Biol. 164:23. DOI: dx.doi.org/10.1007/s00227-016-3052-2 .

Sigler, M. F., C. R. Lunsford, J. M. Straley, and J. B. Liddle. 2008. Sperm whale depredation of sablefish longline gear in the northeast Pacific Ocean. Mar. Mammal Sci. 24(1):16-27.

Straley, J., T. O'Connell, S. Mesnick, L. Behnken, and J. Liddle. 2005. Sperm whale and longline fisheries interactions in the Gulf of Alaska. North Pacific Research Board R0309 Final Report. 15 p.

Straley, J. M., G. S. Schorr, A. M. Thode, J. Calambokidis, C. R. Lunsford, E. M. Chenoweth, V. M. O'Connell, and R. D. Andrews. 2014. Depredating sperm whales in the Gulf of Alaska: local habitat use and long distance movements across putative population boundaries. Endang. Species Res. 24:125-135.

Townsend, C. 1935. The distribution of certain whales as shown by logbook records of American whaleships. Zoologica 19:1-50.

Whitehead, H., and T. Arnbom. 1987. Social organization of sperm whales off the Galapagos Islands, February-April 1985. Can. J. Zool. 65(4):913-919.

Young, N. C., B. J. Delean, V. T. Helker, J. C. Freed, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2014-2018. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-413, 142 p.

Revised 5/15/2013

BAIRD'S BEAKED WHALE (*Berardius bairdii*): Alaska Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Baird's beaked, or giant bottlenose, whale inhabits the North Pacific Ocean and adjacent seas (Bering Sea, Okhotsk Sea, Sea of Japan, and the Sea of Cortez in the southern Gulf of California, Mexico), with the best-known populations occurring in the coastal waters around Japan (Balcomb 1989) and the Commander Islands (Fedutin et al. 2012). Within the North Pacific Ocean, Baird's beaked whales have been sighted in virtually all areas north of 30°N in deep waters over the continental shelf, particularly in regions with submarine escarpments and seamounts (Ohsumi 1983, Kasuya and Ohsumi 1984, Kasuya 2002). The range of the species extends north from Cape Navarin (62° N) and the central Sea of Okhotsk (57° N) to St. Matthew Island, the Pribilof Islands in the Bering Sea, and the northern Gulf of Alaska (Rice 1986, Rice 1998, Kasuya 2002) (Fig. 1). An apparent break in distribution occurs in the eastern Gulf of Alaska, but from the mid-Gulf to the Aleutian Islands and in the southern Bering Sea there are numerous sighting records (Kasuya and Ohsumi 1984, Forney and Brownell 1996, Moore et al. 2002). In the Sea of Okhotsk and the Bering Sea, Baird's beaked whales arrive in April-May, are numerous during the summer, and decrease in October (Tomilin 1957, Kasuya 2002). Observations during 2007-2011 in the western Bering Sea were made in all months except winter (December to March) around the Commander Islands, with encounters peaking in April-June and to a lesser extent in August-November (Fedutin et al. 2012). During winter months, they are rarely found in offshore waters and their winter distribution is unknown (Kasuya 2002). However, acoustic detections of Baird's beaked whales from November through January (and no detections in July-October) in the northern Gulf of Alaska suggest that this region may be wintering habitat for some Baird's beaked whales (Baumann-Pickering et al. 2012b). There were no detections of this species from early June to late August 2010 off Kiska Island (Baumann-Pickering et al. 2012a). They are the most commonly seen beaked whales within their range, perhaps because they are relatively large and gregarious, traveling in schools of a few to several dozen, making them more noticeable to observers than other beaked whale species. Baird's beaked whales are migratory, arriving in continental slope waters during summer and fall months when surface water temperatures are the highest (Dohl et al. 1983, Kasuya 1986). Photo-identification analysis of animals sighted between 2007-2011 revealed resightings of some individuals around the Commander Islands and confirmed associations of individuals over several years in this species (Fedutin et al. 2012).

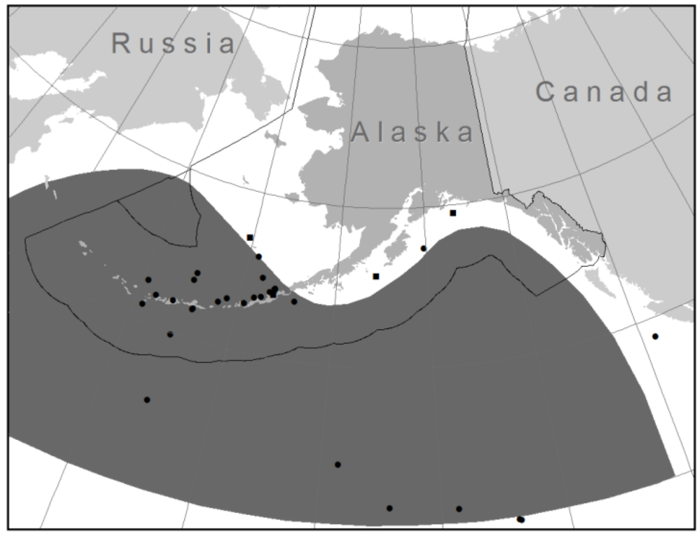

Figure 1. Approximate distribution of Baird's beaked whales in the eastern North Pacific (shaded area). Sightings (circles) and strandings (squares) within the last 10 years are also depicted. (Forney and Brownell 1996, Moore et al. 2002, NMFS unpublished data). Note: Distribution updated based on Kasuya 2002.

There are insufficient data to apply the phylogeographic approach to stock structure (Dizon et al. 1992) for Baird's beaked whale. Therefore, Baird's beaked whale stocks are defined as the two non-contiguous areas within Pacific U. S. waters where they are found: 1) Alaska and 2) California/Oregon/Washington. These two stocks were defined in this manner because of: 1) the large distance between the two areas in conjunction with the lack of any information about whether animals move between the two areas, 2) the somewhat different oceanographic habitats found in the two areas, and 3) the different fisheries that operate within portions of those two areas, with bycatch of Baird's beaked whales only reported from the California/Oregon thresher shark and swordfish drift gillnet fishery. The California/Oregon/Washington Baird's beaked whale stock is reported separately in the Stock Assessment Reports for the Pacific Region.

POPULATION SIZE

Reliable estimates of abundance for this stock are currently unavailable.

Minimum Population Estimate

At this time, it is not possible to produce a reliable minimum population estimate (N_{MIN}) for this stock, as current estimates of abundance are unavailable.

Current Population Trend

No reliable estimates of abundance are available for this stock; therefore, reliable data on trends in population abundance are unavailable.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate is currently unavailable for the Alaska stock of Baird's beaked whale. Hence, until additional data become available, it is recommended that the cetacean maximum theoretical net productivity rate (R_{MAX}) of 4% be employed (Wade and Angliss 1997).

POTENTIAL BIOLOGICAL REMOVAL

Under the 1994 reauthorized Marine Mammal Protection Act (MMPA), the potential biological removal (PBR) is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: $PBR = N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for these stocks is 0.5, the value for cetacean stocks with unknown population status (Wade and Angliss 1997). However, in the absence of a reliable estimate of minimum abundance, the PBR for this stock is unknown.

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

Fisheries Information

Twenty-two different commercial fisheries operating within the potential range of the Alaska stock of Baird's beaked whale were monitored for incidental take by fisheries observers from 2007-2011 (see 76 FR 73912, final List of Fisheries for 2012) . There were no serious injuries or mortalities of Baird's beaked whales incidental to observed commercial fisheries reported between 2007-2011 (Breiwick 2013). The estimated annual mortality rate incidental to commercial fisheries is zero.

Subsistence/Native Harvest Information

There is no known subsistence harvest of Baird's beaked whales by Alaska Natives.

Other Mortality

Between 1925 and 1987, 618 Baird's beaked whales were reported taken throughout the North Pacific (International Whaling Commission, BWIS catch data, February 2003 version, unpublished). The annual quota of Baird's beaked whales for small-type whaling in Japan was 62 from 1999-2004, which increased temporarily to 66 from 2005-2010 and will remain a permanent increase (Kasuya 2011). Due to the unknown stock structure and migratory patterns in the North Pacific, it is unclear whether these animals belong to the Alaska stock of Baird's beaked whales.

STATUS OF STOCK

Baird's beaked whales are not designated as "depleted" under the MMPA or listed as "threatened" or "endangered" under the Endangered Species Act. Reliable estimates of the minimum population, population trends, PBR, and status of the stock relative to its Optimum Sustainable Population size are currently not available. Because the PBR is unknown, the level of annual U.S. commercial fishery-related mortality that can be considered insignificant and approaching zero mortality and serious injury rate is unknown. However, the estimated annual rate of human-caused mortality and serious injury seems minimal for this stock. Thus, the Alaska stock of Baird's beaked whale is not classified as strategic.

Habitat concerns

Disturbance by anthropogenic noise is an increasing habitat concern for most species of beaked whales, particularly in areas of oil and gas activities or where shipping or military activities are high. Shipping noise and the use of military sonars have been found to alter dive behavior and movements, as well as vocal activity in some species of beaked whales (Aguilar de Soto et al. 2006, McCarthy et al. 2011, Tyack et al. 2011). Little is known about the effects of noise on beaked whales in Alaska. Ingestion of marine debris, particularly plastics, is a concern; plastic is occasionally found in the stomach contents of stranded beaked whales, including Baird's beaked whales (Smithsonian Institution, Cetacean Distributional Database, accessed 04 June 2012).

CITATIONS

Aguilar deSoto, N., M. Johnson, P. T. Madsen, P. L. Tyack, A. Bocconcelli and F. Borsani. 2006. Does intense ship noise disrupt foraging in deep diving Cuvier's beaked whales (*Ziphius cavirostris*)? Mar. Mamm. Sci., 22 (3), 690-699.

Angliss, R. P., and D. P. DeMaster. 1998. Differentiating serious and non-serious injury of marine mammals taken incidental to commercial fishing operations. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-OPR-13, 48 p.

Andersen, M. S., K. A. Forney, T. V. N. Cole, T. Eagle, R. Angliss, K. Long, L. Barre, L. Van Atta, D. Borggaard, T. Rowles, B. Norberg, J. Whaley, and L. Engleby. 2008. Differentiating Serious and Non-Serious Injury of Marine Mammals: Report of the Serious Injury Technical Workshop, 10-13 September 2007, Seattle, Washington. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-OPR-39. 94 p.

Balcomb, K. C. 1989. Baird's beaked whale, *Berardius bairdii* Stejneger, 1883: Arnoux's beaked whale *Berardius arnouxii* Douvernoy, 1851. Pp. 261-288 *In* S. H. Ridgway and R. Harrison (eds.), Handbook of marine mammals: River dolphins and the larger toothed whales. Academic Press, New York.

Baumann-Pickering S., A. E. Simonis, S. M. Wiggins, et al. 2012a. Aleutian Islands beaked whale echolocation signals. Mar. Mamm. Sci. 29:221–227. doi: 10.1111/j.1748-7692.2011.00550.x

Baumann-Pickering, S., A. Širović, J. Hildebrand, A. Debich, R. Gottlieb, S. Johnson, S. Kerosky, L. Roche, A. S. Berga, L. Wakefield, and S. Wiggins. 2012b. Passive Acoustic Monitoring for Marine Mammals in the Gulf of Alaska Temporary Maritime Activities Area 2011-2012. Marine Physical Laboratory, Scripps Institute of Oceanography. MPL Technical Memorandum # 538.

Breiwick, J. M. 2013. North Pacific marine mammal bycatch estimation methodology and results, 2007-2011. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-260, 40 p.

Dizon, A. E., C. Lockyer, W. F. Perrin, D. P. DeMaster, and J. Sisson. 1992. Rethinking the stock concept: a phylogeographic approach. Conserv. Biol. 6:24-36.

Dohl, T., R. Guess, M. Duman, and R. Helm. 1983. Cetaceans of central and northern California, 1980-1983: status, abundance, and distribution. Rep. Outer Continental Shelf Study, MMS 84-0045, U.S. Dep. Interior.

Fedutin, I. D., O. A. Filatove, E. G. Mamaev, E. I. Chekalski, A. M. Burdin, and E. Hoyt. 2012. The results of long-term monitoring and first evidence of stable social associations in Baird's beaked whales (*Berardius bairdii*) in the waters of the Commander Islands, Russian Far East, (SC/64/SM5). 11 pp.

Forney, K. A., and R. L. Brownell. 1996. Preliminary report of the 1994 Aleutian Island marine mammal survey. Unpubl. doc. submitted to Int. Whal. Comm. (SC/48/O11). 15 pp.

Kasuya, T. 2011. Conservation Biology of Small Cetacean around Japan. University of Tokyo Press. In Japanese.

Kasuya, T. 1986. Distribution and behavior of Baird's beaked whales off the Pacific coast of Japan. Sci. Rep. Whales Res. Inst. 37:61-83.

Kasuya, T. 2002. Giant beaked whales. Pp. 519-522 *In* William F. Perrin, Bernd Würsig and J. G. M. Thewissen editors, Encyclopedia of marine mammals. Academic Press, San Diego, CA.

Kasuya, T., and Ohsumi, S. 1984. Further analysis of the Baird's beaked whale stock in the western North Pacific. Rep. Int. Whal. Comm. 34:587-595.

McCarthy E., D. Moretti, L. Thomas, N. DiMarzio, R. Morrissey, et al. 2011. Changes in spatial and temporal distribution and vocal behavior of Blainville's beaked whales (*Mesoplodon densirostris*) during multiship exercises with mid-frequency sonar. Marine Mammal Science 27: E206–E226.

Moore, S. E., J. M. Waite, N. A. Friday and T. Honkalehto. 2002. Distribution and comparative estimates of cetacean abundance on the central and south-eastern Bering Sea shelf with observations on bathymetric and prey associations. Progr. Oceanogr. 55(1-2):249-262.

NOAA. 2012. Federal Register 77:3233. National Policy for Distinguishing Serious From Non-Serious Injuries of Marine Mammals. Available online: http://www.nmfs.noaa.gov/op/pds/documents/02/238/02-238-01.pdf.

Ohsumi, S. 1983. Population assessment of Baird's beaked whales in the waters adjacent to Japan. Rep. Int. Whal. Comm. 33:633-641.

Rice, D. W. 1986. Beaked whales. Pp. 102-109 *In* D. Haley (ed.), Marine mammals of the eastern North Pacific and Arctic waters. Pacific Search Press, Seattle.

Rice, D. W. 1998. Marine mammals of the world: Systematics and distribution. The Society for Marine Mammalogy, Special pub. 4, Allen Press, Lawrence, KS, 231 pp.

Tomilin, A. G. 1957. Mammals of the USSR and Adjacent Countries. vol. 9. Cetacea. Izdatel'stvo Akademi Nauk SSSR, Moscow. 756pp. (English translation by Israel Program Sci. Transl. 1967. 717pp. Available from U.S. Dep. Commer., Natl. Tech. Info. Serv., Springfield, VA, as TT 65-50086.)

Tyack P. L., Zimmer W. M. X., Moretti D., Southall B. L., Claridge D. E., Durban J. W., Clark C. W., D'Amico A., DiMarzio N., Jarvis S., McCarthy E., Morrissey R., Ward J., Boyd I. L. 2011. Beaked Whales Respond to Simulated and Actual Navy Sonar. PLoS ONE 6(3):e17009. doi:10.1371/journal.pone.0017009

Wade, P. R., and R. Angliss. 1997. Guidelines for assessing marine mammal stocks: report of the GAMMS workshop April 3-5, 1996, Seattle, Washington. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-OPR-12, 93 pp.

CUVIER'S BEAKED WHALE (*Ziphius cavirostris*): Alaska Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The distribution of Cuvier's beaked, or goosebeak, whale (Fig. 1) is known primarily from strandings, which indicate that it is the most widespread of the beaked whales and is distributed in all oceans and most seas except in the high polar waters (Moore 1963). In the Pacific, they range north to the northern Gulf of Alaska, the Aleutian Islands, and the Commander Islands (Rice 1986, 1998). In the northeastern Pacific from Alaska to Baja California, no obvious pattern of seasonality to strandings has been identified (Mitchell 1968). Strandings of Cuvier's beaked whales are the most numerous of all beaked whales, indicating that they are probably not as rare as originally thought (Heyning 1989). Observations reveal that the blow is low, diffuse, and directed forward (Backus and Schevill 1961, Norris and Prescott 1961), making sightings more difficult, and there is some evidence that they avoid vessels by diving (Heyning 1989). Relatively few (4 total) acoustic detections of Cuvier's beaked whales were recorded off Kiska Island (1 in summer) and in the offshore Gulf of Alaska (3 total detections, 1 in October and 2 in January; Baumann-Pickering et al. 2012a, 2012b).

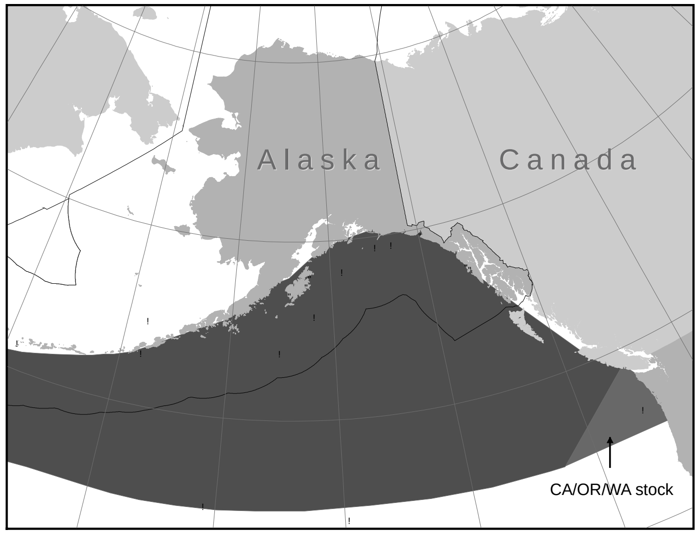

Figure 1. Approximate distribution of Cuvier's beaked whales in the eastern North Pacific (shaded area). Sightings (circles) and strandings (squares) within the last 10 years are also depicted (Forney and Brownell 1996, NMFS unpublished data).

Mitchell (1968) examined skulls of stranded whales for geographical differences and thought that there was probably one panmictic population in the northeastern Pacific. Otherwise, there are insufficient data to apply the phylogeographic approach to stock structure (Dizon et al. 1992) for the Cuvier's beaked whale. Therefore, Cuvier's beaked whale stocks are defined as the three non-contiguous areas within Pacific U. S. waters where they are found: 1) Alaska, 2) California/Oregon/Washington, and 3) Hawaii. These three stocks were defined in this way because of: 1) the large distance between the areas in conjunction with the lack of any information about whether animals move between the three areas, 2) the different oceanographic habitats found in the three areas, and 3) the different fisheries that operate within portions of those three areas, with bycatch of Cuvier's beaked whales only reported from the California/Oregon thresher shark and swordfish drift gillnet fishery. The California/Oregon/Washington and Hawaiian Baird's beaked whale stocks are reported separately in the Stock Assessment Reports for the Pacific Region.

POPULATION SIZE

Reliable estimates of abundance for this stock are currently unavailable.

Minimum Population Estimate

At this time, it is not possible to produce a reliable minimum population estimate (N_{MIN}) for this stock, as current estimates of abundance are unavailable.

Current Population Trend

No reliable estimates of abundance are available for this stock; therefore, reliable data on trends in population abundance are unavailable.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate is currently unavailable for the Alaska stock of Cuvier's beaked whale. Hence, until additional data become available, it is recommended that the cetacean maximum theoretical net productivity rate (R_{MAX}) of 4% be employed (Wade and Angliss 1997).

POTENTIAL BIOLOGICAL REMOVAL

Under the 1994 reauthorized Marine Mammal Protection Act (MMPA), the potential biological removal (PBR) is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: $PBR = N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 0.5, the value for cetacean stocks with unknown population status (Wade and Angliss 1997). However, in the absence of a reliable estimate of minimum abundance, the PBR for this stock is unknown.

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

Fisheries Information

Twenty-two different commercial fisheries operating within the potential range of the Alaska stock of Cuvier's beaked whale were monitored for incidental take by fishery observers from 2007-2011 (see 76 FR 73912, final List of Fisheries for 2012) . There were no serious injuries or mortalities of Cuvier's beaked whales incidental to observed commercial fisheries reported between 2007-2011 (Breiwick 2013). The estimated annual mortality rate incidental to commercial fisheries is zero.

Subsistence/Native Harvest Information

There is no known subsistence harvest of Cuvier's beaked whales.

Other Mortality

Unknown levels of injuries and mortality of Cuvier's beaked whales may occur as a result of anthropogenic noise, such as military sonars (U.S. Dept. of Commerce and Secretary of the Navy 2001) or other commercial and scientific activities producing high-energy sound. The use of active sonar from military vessels has been implicated or coincident with mass strandings of beaked whales (Cox et al. 2006, Frantzis 1998, Martel 2002, Jepson et al. 2003, Simmonds and Lopez-Jurado 1991, U.S. Dept. of Commerce and Secretary of the Navy 2001), and all atypical single and mixed-species mass strandings involved Cuvier's beaked whales (D'Amico et al. 2009). There is concern regarding the potential effects of underwater sounds from seismic operations on beaked whales, although investigations of causation of atypical strandings of Cuvier's beaked whales and nearby seismic air gun operations have been inconclusive (Gentry 2002, Gordon et al. 2003/2004, Malakoff 2002). Changes in dive behavior, particularly a quick ascent from deep dives, in response to sound exposure may result in injuries related to bubble growth during decompression (Cox et al. 2006, Tyack et al. 2011, Hooker et al. 2011). Such injuries or mortality would rarely be documented due to the remote nature of many of these activities and the low probability that an injured or dead beaked whale would strand. No estimates of potential mortality or serious injury are available for Cuvier's beaked whales in Alaska waters.

STATUS OF STOCK

Cuvier's beaked whales are not designated as "depleted" under the MMPA or listed as "threatened" or "endangered" under the Endangered Species Act. Reliable estimates of the minimum population, population trends, PBR, and status of the stock relative to its Optimum Sustainable Population size are currently not available. Because the PBR is unknown, the level of annual U.S. commercial fishery-related mortality that can be considered insignificant and approaching zero mortality and serious injury rate is unknown. However, the estimated annual rate of human-caused mortality and serious injury seems minimal for this stock. Thus, the Alaska stock of Cuvier's beaked whale is not classified as strategic.

Habitat concerns

Disturbance by anthropogenic noise is an increasing habitat concern for most species of beaked whales, particularly in areas of oil and gas activities or where shipping or military activities are high. Shipping noise may disrupt the behavior of Cuvier's beaked whales (Aguilar de Soto et al. 2006), and the use of military sonars has been found to alter dive behavior and movements, as well as vocal activity in some species of beaked whales (McCarthy et al. 2011, Tyack et al. 2011). Moore and Barlow (2013) report impacts of anthropogenic sound and ecosystem change as the most plausible hypotheses for declining abundance of *Ziphius* and *Mesoplodon* spp. in the California Current large marine ecosystem. Little is known about the effects of noise or ecosystem change on beaked whales in Alaska, and the lack of abundance estimates hinder the detection of any population trends. Ingestion of marine debris, particularly plastics, is a concern; plastic is occasionally found in the stomach contents of stranded beaked whales, including Cuvier's beaked whales. (Smithsonian Institution, Cetacean Distributional Database, accessed 04 June 2012).

CITATIONS

Aguilar deSoto, N., Johnson, M., Madsen, P. T., Tyack, P. L., Bocconcelli, A. & Borsani, F. 2006. Does intense ship noise disrupt foraging in deep diving Cuvier's beaked whales (*Ziphius cavirostris*)? Mar. Mamm. Sci. 22 (3), 690-699.

Angliss, R. P., and D. P. DeMaster. 1998. Differentiating serious and non-serious injury of marine mammals taken incidental to commercial fishing operations. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-OPR-13, 48 p.

Andersen, M. S., K. A. Forney, T. V. N. Cole, T. Eagle, R. Angliss, K. Long, L. Barre, L. Van Atta, D. Borggaard, T. Rowles, B. Norberg, J. Whaley, and L. Engleby. 2008. Differentiating Serious and Non-Serious Injury of Marine Mammals: Report of the Serious Injury Technical Workshop, 10-13 September 2007, Seattle, Washington. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-OPR-39. 94 p.

Backus, R. H., and W. E. Schevill. 1961. The stranding of a Cuvier's beaked whale (*Ziphius cavirostris*) in Rhode Island, USA. Norsk Hval. 50:177-181.

Baumann-Pickering S., A. E. Simonis, S. M. Wiggins, et al. 2012a. Aleutian Islands beaked whale echolocation signals. Mar. Mamm. Sci. 29:221–227. doi: 10.1111/j.1748-7692.2011.00550.x

Baumann-Pickering, S., A. Širović, J. Hildebrand, A. Debich, R. Gottlieb, S. Johnson, S. Kerosky, L. Roche, A. S. Berga, L. Wakefield, and S. Wiggins. 2012b. Passive Acoustic Monitoring for Marine Mammals in the Gulf of Alaska Temporary Maritime Activities Area 2011-2012. Marine Physical Laboratory, Scripps Institute of Oceanography. MPL Technical Memorandum # 538.

Breiwick, J. M. 2013. North Pacific marine mammal bycatch estimation methodology and results, 2007-2011. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-260, 40 p.

Cox, T. M., T. J. Ragen, A. J. Read, E. Vos, R. W. Baird, K. Balcomb, J. Barlow, J. Caldwell, T. Cranford, L. Crum, A. D'Amico, G. D'Spain, A. Fern´andez, J. Finneran, R. Gentry, W. Gerth, F. Gulland, J. Hildebrand, D. Houser, T. Hullar, P. D. Jepson, D. Ketten, C. D. MacLeod, P. Miller, S. Moore, D. C. Mountain, D. Palka, P. Ponganis, S. Rommel, T. Rowles, B. Taylor, P. Tyack, D. Wartzok, R. Gisiner, J. Mead and L. Benner. 2006. Understanding the impacts of anthropogenic sound on beaked whales. J. Cetacean Res. Manage. 7:177–187.

D'Amico, A., Gisiner, R. C., Ketten, D. R., Hammock, J. A., Johnson, C., Tyack, P. L., and Mead, J. 2009. Beaked whale strandings and naval exercises. Aquat. Mamm. 34: 452–472.

Dizon, A. E., C. Lockyer, W. F. Perrin, D. P. DeMaster, and J. Sisson. 1992. Rethinking the stock concept: a phylogeographic approach. Conserv. Biol. 6:24-36.

Forney, K. A., and R. L. Brownell. 1996. Preliminary report of the 1994 Aleutian Island marine mammal survey. Unpubl. doc. submitted to Int. Whal. Comm. (SC/48/O11). 15 pp.

Frantzis, A. 1998. Does acoustic testing strand whales? Nature 392:29.

Gentry, R. L. 2002. Mass stranding of beaked whales in the Galapagos Islands, April 2000. http://www.nmfs.noaa.gov/prot_res/PR2/Health_and_Stranding_Response_Program/Mass_Galapagos_Islands.htm.

Gordon, J., D. Gillespie, J. Potter, A. Frantzis, M. P. Simmonds, R. Swift, D. Thompson. 2003/2004. A review of the effects of seismic surveys on marine mammals. Mar. Tech. Soc. J. 37(4): 16-34.

Heyning, J. E. 1989. Cuvier's beaked whale - *Ziphius cavirostris* G. Cuvier, 1823. Pp. 289-308 *In* S. H. Ridgway and R. Harrison (eds.), Handbook of marine mammals: River dolphins and the larger toothed whales. Academic Press, New York.

Hooker, S. K., A. Fahlman, M. J. Moore, N. Aguilar de Soto, Y. Bernaldo de Quirós, A. O. Brubakk, D. P. Costa, A. M. Costidis, S. Dennison, K. J. Falke, A. Fernandez, M. Ferrigno, J. R. Fitz-Clarke, M. M. Garner, D. S. Houser, P. D. Jepson, D. R. Ketten, P. H. Kvadsheim, P. T. Madsen, N. W. Pollock, D. S. Rotstein, T. K. Rowles, S. E. Simmons, W. Van Bonn, P. K. Weathersby, M. J. Weise, T. M. Williams and P. L. Tyack. 2011. Deadly diving? Physiological and behavioral management of decompression stress in diving mammals. Proc. R. Soc. B. doi: 10.1098/rspb.2011.2088.

Jepson, P. D., M. Arbelo, R. Deaville, I. A. P. Patterson, P. Castro, J. R. Baker, E. Degollada, H. M. Ross, P. Herraez, A. M. Pocknell, F. Rodriguez, F. E. Howie, A. Espinosa, R. J. Reid, J. R. Jaber, V. Martin, A. A. Cunningham, and A. Fernández. 2003. Gas-bubble lesions in stranded animals: Was sonar responsible for a spate of whale deaths after an Atlantic military exercise? Nature 425(6958):575-76.

Malakoff, D. 2002. Suit ties whale deaths to research cruise. Science 298:722-723.

Martel, V. M. 2002. Summary of the report on the atypical mass stranding of beaked whales in the Canary Islands in September 2002 during naval exercises. Society for the Study of the Cetaceans in the Canary Archipelago (SECAC). Unpublished report. 11p.

McCarthy E., D. Moretti, L. Thomas, N. DiMarzio, R. Morrissey, et al. 2011. Changes in spatial and temporal distribution and vocal behavior of Blainville's beaked whales (*Mesoplodon densirostris*) during multiship exercises with mid-frequency sonar. Mar. Mamm. Sci. 27: E206–E226.

Mitchell, E. 1968. Northeast Pacific stranding distribution and seasonality of Cuvier's beaked whale, *Ziphius cavirostris.* Can. J. Zool. 46:265-279.

Moore, J. C. 1963. The goose-beaked whale, where in the world? Bull. Chicago Nat. Hist. Mus. 34:2-3, 8.

Moore J. E., J. P. Barlow. 2013. Declining Abundance of Beaked Whales (Family Ziphiidae) in the California Current Large Marine Ecosystem. PLoS ONE 8(1): e52770. doi:10.1371/journal.pone.0052770

Norris, K. S., and J. H. Prescott. 1961. Observations on Pacific cetaceans of California and Mexican waters. Univ. Calif. Pub. Zool. 63:291-370.

NOAA. 2012. Federal Register 77:3233. National Policy for Distinguishing Serious From Non-Serious Injuries of Marine Mammals. http://www.nmfs.noaa.gov/op/pds/documents/02/238/02-238-01.pdf.

Rice, D. W. 1986. Beaked whales. Pp. 102-109 *In* D. Haley (ed.), Marine mammals of the eastern North Pacific and Arctic waters. Pacific Search Press, Seattle.

Rice, D. W. 1998. Marine mammals of the world: Systematics and distribution. The Society for Marine Mammalogy, Special pub. 4, Allen Press, Lawrence, KS, 231 pp.

Simmonds, M. P., and L. F. Lopez-Jurado. 1991. Whales and the military. Nature 351:448.

Tyack P. L., Zimmer W. M. X., Moretti D., Southall B. L., Claridge D. E., Durban J. W., Clark C. W., D'Amico A., DiMarzio N., Jarvis S., McCarthy E., Morrissey R., Ward J., Boyd I. L. 2011. Beaked Whales Respond to Simulated and Actual Navy Sonar. PLoS ONE 6(3):e17009. doi:10.1371/journal.pone.0017009

United States Department of Commerce and United States Navy. 2001. Joint interim report on the Bahamas marine mammal stranding event of 15-16 March 2000 (December 2001). NOAA unpublished report. 59 pp. [Available at http://www.nmfs.noaa.gov/pr/pdfs/health/stranding_bahamas2000.pdf].

Wade, P. R., and R. Angliss. 1997. Guidelines for assessing marine mammal stocks: report of the GAMMS workshop April 3-5, 1996, Seattle, WA. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-OPR-12, 93 pp.

Revised 2/11/2013

STEJNEGER'S BEAKED WHALE (*Mesoplodon stejnegeri*): Alaska Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Stejneger's, or Bering Sea, beaked whale is rarely seen at sea, and its distribution generally has been inferred from stranded specimens (Loughlin and Perez 1985, Mead 1989, Walker and Hanson 1999). It is endemic to the cold-temperate waters of the North Pacific Ocean, Sea of Japan, and deep waters of the southwest Bering Sea (Fig. 1). The range of Stejneger's beaked whale extends along the coast of North America from Cardiff, California, north through the Gulf of Alaska to the Aleutian Islands, into the Bering Sea to the Pribilof Islands and Commander Islands, and, off Asia, south to Akita Beach on Noto Peninsula, Honshu, in the Sea of Japan (Loughlin and Perez 1985). Near the central Aleutian Islands, groups of 3-15 Stejneger's beaked whales have been sighted on a number of occasions (Rice 1986). The species is not known to enter the Arctic Ocean and is the only species of *Mesoplodon* known to occur in Alaska waters. The distribution of *M. stejnegeri* in the North Pacific corresponds closely, in occupying the same cold-temperate niche and position, to that of *M. bidens* in the North Atlantic. It lies principally between 50°and 60°N and extends only to about 45°N in the eastern Pacific, but to about 40°N in the western Pacific (Moore 1963, 1966). Acoustic signals believed to be produced by Stejneger's beaked whales (based on frequency characteristics, interpulse interval and geographic location, Baumann-Pickering et al. 2012a) were recorded 2-5 times a week in July off Kiska Island and almost weekly from July 2011 to February 2012 in the northern Gulf of Alaska (Baumann-Pickering et al. 2012b).

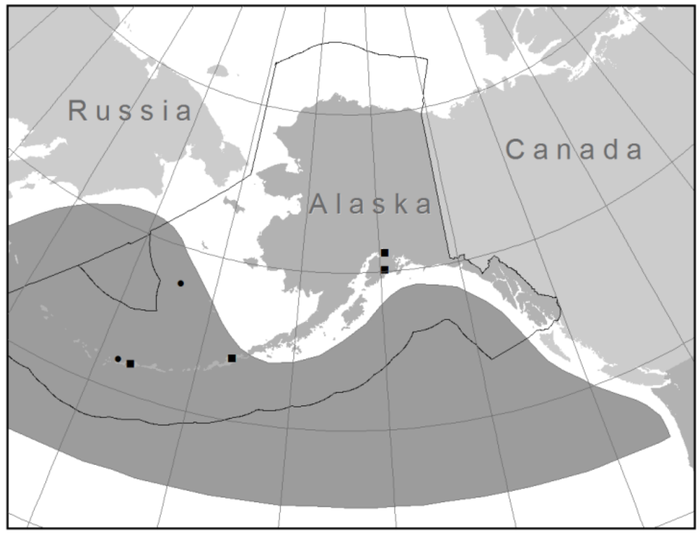

Figure 1. Approximate distribution of Stejneger's beaked whales in the eastern North Pacific (shaded area). Sightings (circles) and strandings (squares) within the last 10 years are also depicted (Walker and Hanson 1999, NMFS unpublished data).

There are insufficient data to apply the phylogeographic approach to stock structure (Dizon et al. 1992) for Stejneger's beaked whale. The Alaska Stejneger's beaked whale stock is recognized separately from *Mesoplodon* spp. off California, Oregon, and Washington because of: 1) the distribution of Stejneger's beaked whale and the different oceanographic habitats found in the two areas, 2) the large distance between the two non-contiguous areas of U.S. waters in conjunction with the lack of any information about whether animals move between the two areas, and 3) the different fisheries that operate within portions of those two areas, with bycatch of *Mesoplodon* spp. only reported from the California/Oregon thresher shark and swordfish drift gillnet fishery. The California/Oregon/Washington stock of all *Mesoplodon* spp. and a *Mesoplodon densirostris* stock in Hawaiian waters are reported separately in the Stock Assessment Reports for the Pacific Region.

POPULATION SIZE

Reliable estimates of abundance for this stock are currently unavailable.

Minimum Population Estimate

At this time, it is not possible to produce a reliable minimum population estimate (N_{MIN}) for this stock, as current estimates of abundance are unavailable.

Current Population Trend

No reliable estimates of abundance are available for this stock; therefore, reliable data on trends in population abundance are unavailable.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate is currently unavailable for the Alaska stock of Stejneger's beaked whale. Hence, until additional data become available, it is recommended that the cetacean maximum theoretical net productivity rate (R_{MAX}) of 4% be employed (Wade and Angliss 1997).

POTENTIAL BIOLOGICAL REMOVAL

Under the 1994 reauthorized Marine Mammal Protection Act (MMPA), the potential biological removal (PBR) is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: $PBR = N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 0.5, the value for cetacean stocks with unknown population status (Wade and Angliss 1997). However, in the absence of a reliable estimate of minimum abundance, the PBR for this stock is unknown.

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

Fisheries Information

Twenty-two different commercial fisheries operating within the potential range of the Alaska stock of Cuvier's beaked whale were monitored for incidental take by fishery observers from 2007-2011 (see 76 FR 73912, final List of Fisheries for 2012) . There were no serious injuries or mortalities of Stejneger's beaked whales incidental to observed commercial fisheries reported between 2007-2011 (Breiwick 2013). The estimated annual mortality rate incidental to commercial fisheries is zero.

Subsistence/Native Harvest Information

There is no known subsistence harvest of Stejneger's beaked whales.

STATUS OF STOCK

Stejneger's beaked whales are not designated as "depleted" under the MMPA or listed as "threatened" or "endangered" under the Endangered Species Act. Reliable estimates of the minimum population, population trends, PBR, and status of the stock relative to its Optimum Sustainable Population size are currently not available. Because the PBR is unknown, the level of annual U.S. commercial fishery-related mortality that can be considered insignificant and approaching zero mortality and serious injury rate is unknown. However, the estimated annual rate of human-caused mortality and serious injury seems minimal for this stock. Thus, the Alaska stock of Stejneger's beaked whale is not classified as strategic.

Habitat concerns

Disturbance by anthropogenic noise is an increasing habitat concern for most species of beaked whales, particularly in areas of oil and gas activities or where shipping or military activities are high. Shipping noise and the use of military sonars have been found to alter dive behavior and movements, as well as vocal activity in some species of beaked whales (Aguilar de Soto et al. 2006, McCarthy et al. 2011, Tyack et al. 2011). Moore and Barlow (2013) report impacts of anthropogenic sound and ecosystem change as the most plausible hypotheses for declining abundance of *Ziphius* and *Mesoplodon* spp., including *M. stejnegeri*, in the California Current large marine ecosystem. Little is known about the effects of noise on beaked whales in Alaska. Ingestion of marine debris, particularly plastics, is a concern; plastic is occasionally found in the stomach contents of stranded beaked whales, including Stejneger's beaked whales. (Smithsonian Institution, Cetacean Distributional Database, accessed 04 June 2012).

CITATIONS

Aguilar deSoto, N., Johnson, M., Madsen, P. T., Tyack, P. L., Bocconcelli, A. & Borsani, F. 2006. Does intense ship noise disrupt foraging in deep diving Cuvier's beaked whales (*Ziphius cavirostris*)? Mar. Mamm. Sci. 22 (3), 690-699.

Angliss, R. P., and D. P. DeMaster. 1998. Differentiating serious and non-serious injury of marine mammals taken incidental to commercial fishing operations. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-OPR-13, 48 p.

Andersen, M. S., K. A. Forney, T. V. N. Cole, T. Eagle, R. Angliss, K. Long, L. Barre, L. Van Atta, D. Borggaard, T. Rowles, B. Norberg, J. Whaley, and L. Engleby. 2008. Differentiating Serious and Non-Serious Injury of Marine Mammals: Report of the Serious Injury Technical Workshop, 10-13 September 2007, Seattle, Washington. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-OPR-39. 94 p.

Baumann-Pickering S., A. E. Simonis, S. M. Wiggins, et al. 2012a. Aleutian Islands beaked whale echolocation signals. Mar. Mamm. Sci. 29:221–227. doi: 10.1111/j.1748-7692.2011.00550.x

Baumann-Pickering, S., A. Širović, J. Hildebrand, A. Debich, R. Gottlieb, S. Johnson, S. Kerosky, L. Roche, A. S. Berga, L. Wakefield, and S. Wiggins. 2012b. Passive Acoustic Monitoring for Marine Mammals in the Gulf of Alaska Temporary Maritime Activities Area 2011-2012. Marine Physical Laboratory, Scripps Institute of Oceanography. MPL Technical Memorandum # 538.

Breiwick, J. M. 2013. North Pacific marine mammal bycatch estimation methodology and results, 2007-2011. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-260, 40 p.

Dizon, A. E., C. Lockyer, W. F. Perrin, D. P. DeMaster, and J. Sisson. 1992. Rethinking the stock concept: a phylogeographic approach. Conserv. Biol. 6:24-36.

Loughlin, T. R., and M. A. Perez. 1985. *Mesoplodon stejnegeri*. Mammalian Species, No. 250.

McCarthy E., D. Moretti, L. Thomas, N. DiMarzio, R. Morrissey, et al. 2011. Changes in spatial and temporal distribution and vocal behavior of Blainville's beaked whales (*Mesoplodon densirostris*) during multiship exercises with mid-frequency sonar. Mar. Mamm. Sci. 27: E206–E226.

Mead, J. G. 1989. Beaked whales of the genus - *Mesoplodon*. Pp. 349-430 *In* S. H. Ridgway and R. Harrison (eds.), Handbook of marine mammals: River dolphins and the larger toothed whales. Academic Press, New York.

Moore, J. C. 1963. Recognizing certain species of beaked whales of the Pacific Ocean. Amer. Midl. Nat. 70:396-428.

Moore, J. C. 1966. Diagnoses and distributions of beaked whales of the genus *Mesoplodon* known from North American waters. Pp. 32-61 *In* K. S. Norris (ed.), Whales, dolphins and porpoises. Univ. California Press, Berkeley.

Moore J. E., J. P. Barlow. 2013. Declining Abundance of Beaked Whales (Family Ziphiidae) in the California Current Large Marine Ecosystem. PLoS ONE 8(1): e52770. doi:10.1371/journal.pone.0052770

NOAA. 2012. Federal Register 77:3233. National Policy for Distinguishing Serious From Non-Serious Injuries of Marine Mammals. http://www.nmfs.noaa.gov/op/pds/documents/02/238/02-238-01.pdf.

Rice, D. W. 1986. Beaked whales. Pp. 102-109 *In* D. Haley (ed.), Marine mammals of the eastern North Pacific and Arctic waters. Pacific Search Press, Seattle.

Tyack P. L., W. M. X. Zimmer, D. Moretti., B. L. Southall, D. E. Claridge, J. W. Durban, C. W. Clark, A. D'Amico, N. DiMarzio, S. Jarvis, E. McCarthy, R. Morrissey, J. Ward, I. L. Boyd. 2011. Beaked Whales Respond to Simulated and Actual Navy Sonar. PLoS ONE 6(3):e17009. doi:10.1371/journal.pone.0017009

Wade, P. R., and R. Angliss. 1997. Guidelines for assessing marine mammal stocks: report of the GAMMS workshop April 3-5, 1996, Seattle, WA. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-OPR-12, 93 pp.

Walker, W. A., and M. B. Hanson. 1999. Biological observations on Stejneger's beaked whale, *Mesoplodon stejnegeri*, from strandings on Adak Island, Alaska. Mar. Mammal Sci. 15(4): 1314-1329.

HUMPBACK WHALE (*Megaptera novaeangliae*): Western North Pacific Stock

NOTE – NMFS is in the process of reviewing humpback whale stock structure under the Marine Mammal Protection Act (MMPA) in light of the 14 Distinct Population Segments established under the Endangered Species Act (ESA) (81 FR 62259, 8 September 2016). A complete revision of the humpback whale stock assessments will be postponed until this review is complete. In the interim, new information on humpback whale mortality and serious injury is provided within this report.

STOCK DEFINITION AND GEOGRAPHIC RANGE

The humpback whale is distributed worldwide in all ocean basins. In winter, most humpback whales occur in the subtropical and tropical waters of the Northern and Southern Hemispheres. Humpback whales in the high latitudes of the North Pacific Ocean are seasonal migrants that feed on euphausiids and small schooling fishes (Nemoto 1957, 1959; Clapham and Mead 1999). The humpback whale population was considerably reduced as a result of intensive commercial exploitation during the 20th century.

A large-scale study of humpback whales throughout the North Pacific was conducted from 2004 to 2006 (the Structure of Populations, Levels of Abundance, and Status of Humpbacks (SPLASH) project). Results from this project (Calambokidis et al. 2008, Barlow et al. 2011), including abundance estimates and movement information, have been reported in Baker et al. (2008, 2013) and are also summarized in Fleming and Jackson (2011); however, these results are still being considered for stock structure analysis.

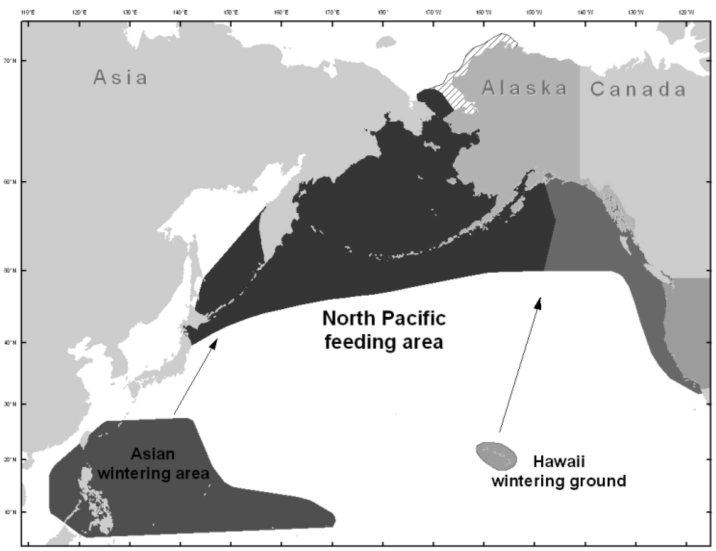

Figure 1. Approximate distribution of humpback whales in the western North Pacific (dark shaded areas). Feeding and wintering grounds are presented above (see text). Area within the hash lines is a probable distribution area based on sightings in the Beaufort Sea (Hashagen et al. 2009). See Figure 1 in the Central North Pacific humpback whale Stock Assessment Report for humpback whale distribution in the eastern North Pacific.

The historical summer feeding range of humpback whales in the North Pacific encompassed coastal and inland waters around the Pacific Rim from Point Conception, California, north to the Gulf of Alaska and the Bering Sea, and west along the Aleutian Islands to the Kamchatka Peninsula and into the Sea of Okhotsk and north of the Bering Strait (Zenkovich 1954, Nemoto 1957, Tomlin 1967, Johnson and Wolman 1984). Historically, the Asian wintering area extended from the South China Sea east through the Philippines, Ryukyu Retto, Ogasawara Gunto, Mariana Islands, and Marshall Islands (Rice 1998). Humpback whales are currently found throughout this historical range (Clarke et al. 2013b), with sightings during summer months occurring as far north as the Beaufort Sea (Hashagen et al. 2009). Most of the current winter range of humpback whales in the North Pacific is relatively well known, with aggregations of whales in Japan, the Philippines, Hawaii, Mexico, and Central America. The winter range includes the main islands of the Hawaiian archipelago, with the greatest concentration along the west side of Maui. In Mexico, the winter breeding range includes waters around the southern part of the Baja California peninsula, the central portions of the Pacific coast of mainland Mexico, and the Revillagigedo Islands off the mainland coast. The winter range also extends from southern Mexico into Central America, including Guatemala, El Salvador, Nicaragua, and Costa Rica (Calambokidis et al. 2008).

Photo-identification data, distribution information, and genetic analyses have indicated that in the North Pacific there are at least three breeding populations (Asia, Hawaii, and Mexico/Central America) that all migrate between their respective winter/spring calving and mating areas and their summer/fall feeding areas (Calambokidis et al. 1997, Baker et al. 1998). Calambokidis et al. (2001) further suggested that there may be as many as six subpopulations on the wintering grounds. From photo-identification and Discovery tag information there are known connections between Asia and Russia, between Hawaii and Alaska, and between Mexico/Central America and California (Darling 1991, Darling and Cerchio 1993, Calambokidis et al. 1997, Baker et al. 1998). This information led to the designation of three stocks of humpback whales in the North Pacific: 1) the California/Oregon/Washington and Mexico stock, consisting of winter/spring populations in coastal Central America and coastal Mexico which migrate to the coast of California and as far north as southern British Columbia in summer/fall (Calambokidis et al. 1989, 1993; Steiger et al. 1991); 2) the Central North Pacific stock, consisting of winter/spring populations of the Hawaiian Islands which migrate primarily to northern British Columbia/Southeast Alaska, the Gulf of Alaska, and the Bering Sea/Aleutian Islands (Baker et al. 1990, Perry et al. 1990, Calambokidis et al. 1997); and 3) the Western North Pacific stock, consisting of winter/spring populations off Asia which migrate primarily to Russia and the Bering Sea/Aleutian Islands (Fig. 1).

Information from the SPLASH project largely confirms this view of humpback whale distribution and movements in the North Pacific. For example, the SPLASH results confirm low rates of interchange between the three principal wintering regions (Asia, Hawaii, and Mexico). However, the full SPLASH results suggest that the current view of population structure is incomplete. The overall pattern of movements is complex but indicates a high degree of population structure. Whales from wintering areas at the extremes of their range on both sides of the Pacific migrate to coastal feeding areas that are on the same side of the Pacific: whales from Asia in the west migrate to Russia and whales from mainland Mexico and Central America in the east migrate to coastal waters off California/Oregon.

The SPLASH data now show that Revillagigedo whales are seen in all sampled feeding areas except northern California/Oregon and the south side of the Aleutians. They are primarily distributed in the Bering Sea, Gulf of Alaska, and Southeast Alaska/northern British Columbia but are also found in Russia and southern British Columbia/Washington. The migratory destinations of humpback whales from Hawaii were found to be quite similar, and a significant number of matches (14) were seen during SPLASH between Hawaii and the Revillagigedos (Calambokidis et al. 2008).

The winter distribution of humpback whales in the Western stock includes several island chains in the western North Pacific. In the Ogasawara Islands, humpback whale sampling during SPLASH was conducted at the three main island groups of Chichi-jima, Haha-jima, and Muko-jima, separated from each other by approximately 50-70 km. SPLASH sampling in Okinawa (southwest of Honshu) occurred at the Okinawa mainland and Zamami in the Kerama Islands (40 km from the Okinawa mainland), and in the Philippines SPLASH sampling occurred only at the northern tip of the archipelago around the Babuyan Islands. Humpback whales are reported to also occur in the South China Sea north of the Philippines near Taiwan, and east of Ogasawara in the Marshall and Mariana Islands (Rice 1998), but there were no known areas of high density in these regions that could be efficiently sampled.

The SPLASH project also found that whales from the Aleutian Islands and Bering Sea, and perhaps the Gulf of Anadyr and the Chukotka Peninsula on the west side of the Bering Strait in Russia, have an unusually low resighting rate in winter areas compared to whales from other feeding areas. It is now believed that some of these whales have a winter migratory destination that was not sampled during the SPLASH project. Given the location of these feeding areas, the most parsimonious explanation would be that some of these whales winter somewhere between Hawaii and Asia, which would include the possibility of the Mariana Islands (southwest of the Ogasawara Islands), the Marshall Islands (approximately half-way between the Mariana Islands and the Hawaiian Islands), and the Northwestern Hawaiian Islands. Subsequent to the SPLASH project, a survey in 2007 documented humpback whales from a number of locations in the Northwestern Hawaiian Islands at relatively low densities (Johnston et al. 2007), but no sampling occurred there during the SPLASH project. Some humpback whales, including mother/calf pairs, have also been found in the Mariana Islands (Hill et al. 2016). Both of these locations are plausible migratory destinations for whales from the Aleutian Islands and Bering Sea. Which stock that whales in these locations would belong to is currently unknown.

The migratory destination of Western North Pacific humpback whales is not completely known. Discovery tag recoveries have indicated movement of whales between Ogasawara and Okinawa and feeding areas in the Bering Sea, on the southern side of the Aleutian Islands, and in the Gulf of Alaska (Omura and Ohsumi 1964, Nishiwaki 1966, Ohsumi and Masaki 1975). Research on humpback whales at the Ogasawara Islands has documented movements of whales between there and British Columbia (Darling et al. 1996), the Kodiak Archipelago in the central Gulf of Alaska (Calambokidis et al. 2001), and the Shumagin Islands in the western Gulf of Alaska (Witteveen et al. 2004), but no photo-identification studies had previously been conducted in Russia. Individual movement information from the SPLASH study documents that Russia is likely the primary migratory destination for whales in Okinawa and the Philippines but also reconfirms that some Asian whales go to Ogasawara, the

Aleutian Islands, Bering Sea, and Gulf of Alaska (Calambokidis et al. 2008). A small amount of inter-yearly interchange was also found between the wintering areas (Philippines, Okinawa, and Ogasawara).

During the SPLASH study in Russia, humpback whales were primarily found along the Pacific east side of the Kamchatka Peninsula, near the Commander Islands between Kamchatka and the Aleutian Islands, and in the Gulf of Anadyr just southwest of the Bering Strait. Analysis of whaling data shows historical catches of humpback whales well into the Bering Sea and catches in the Bering Strait and Chukchi Sea in August-October in the 1930s (Mizroch and Rice 2007), but no survey effort occurred during SPLASH north of the Bering Strait. Humpback whales are increasingly seen north of the Bering Strait into the northeastern Chukchi Sea (Clarke et al. 2013a, 2013b), with some indication that more humpback whales are seen on the Russian side north of the Bering Strait (Clarke et al. 2013b). Humpback whales are the most commonly recorded cetacean on hydrophones just north of the Bering Strait and occurred from September into early November from 2009 to 2012 (K. Stafford, Applied Physics Laboratory-University of Washington, Seattle, WA, pers. comm.). Other locations in the far western Pacific where humpback whales have been seen in summer include the northern Kuril Islands (V. Burkanov, NMFS-AFSC-MML, pers. comm.), far offshore southeast of the Kamchatka Peninsula and south of the Commander Islands (Miyashita 2006), and along the north coast of the Chukotka Peninsula in the Chukchi Sea (Melnikov 2000).

These results indicate humpback whales from the Western North Pacific (Asian) breeding stock overlap broadly on summer feeding grounds with whales from the Central North Pacific breeding stock, as well as with whales that winter in the Revillagigedos in Mexico. Given the relatively small size of the Asian population, Asian whales probably represent a small fraction of all the whales found in the Aleutian Islands, Bering Sea, and Gulf of Alaska, which are primarily whales from Hawaii and the Revillagigedos. The only feeding area that appears to be primarily (or exclusively) composed of Asian whales is along the Kamchatka Peninsula in Russia. The initial SPLASH abundance estimates for Asia ranged from about 900 to 1,100, and the estimates for Kamchatka in Russia ranged from about 100 to 700, suggesting a large portion of the Asian population migrates to Kamchatka. This also shows that Asian whales that migrate to feeding areas besides Russia would be only a small fraction of the total number of whales in those areas, given the much larger abundance estimates for the Bering Sea and Aleutian Islands (6,000-14,000) and the Gulf of Alaska (3,000-5,000) (Calambokidis et al. 2008). A full description of the distribution and density of humpback whales in the Aleutian Islands, Bering Sea, and Gulf of Alaska is in the Stock Assessment Report for the Central North Pacific stock of humpback whales.

In summary, information from a variety of sources indicates that humpback whales from the Western and Central North Pacific stocks mix to a limited extent on summer feeding grounds ranging from British Columbia through the central Gulf of Alaska and up to the Bering Sea.

NMFS conducted a global Status Review of humpback whales (Bettridge et al. 2015) and revised the ESA listing of the species (81 FR 62259, 8 September 2016); the effects of the ESA-listing final rule on the status of the stock are discussed below.

POPULATION SIZE

In the SPLASH study, fluke photographs were collected by over 400 researchers in all known feeding areas from Russia to California and in all known wintering areas from Okinawa and the Philippines to the coast of Central America and Mexico from 2004 to 2006. Over 18,000 fluke identification photographs were collected, and these have been used to estimate the abundance of humpback whales in the entire North Pacific Basin. A total of 566 unique individuals were seen in the Asian wintering areas during the 2-year period (3 winter field seasons) of the SPLASH study. Based on a comparison of all winter identifications to all summer identifications, the Chapman-Petersen estimate of abundance was 21,808 (CV = 0.04) (Barlow et al. 2011). A simulation study identifies significant biases in this estimate from violations of the closed population assumption (+5.3%), exclusion of calves (-10.3%), failure to achieve random geographic sampling (+1.5%), and missed matches (+9.8%) (Barlow et al. 2011). Sex-biased sampling favoring males in wintering areas does not add significant bias if both sexes are proportionately sampled in the feeding areas. The bias-corrected estimate is 20,800 after accounting for a net positive bias of 4.8%. This estimate is likely to be lower than the true abundance due to two additional sources of bias: individual heterogeneity in the probability of being sampled (unquantified) and the likely existence of an unknown and unsampled wintering area (-7.2%).

During the SPLASH study, surveys were conducted in three winter field seasons (2004 to 2006). The total numbers of unique individuals found in each area during the study were 77 in the Philippines, 215 in Okinawa, and 294 in the Ogasawara Islands. There was a total of 20 individuals seen in more than one area, leaving a total of 566 unique individuals seen in the Asian wintering areas (Calambokidis et al. 2008). For abundance in winter or summer areas, a multistrata Hilborn mark-recapture model was used, which is a form of a spatially-stratified model that explicitly estimates movement rates between winter and summer areas. Two broad categories of models were used making different assumptions about the movement rates, and four different models were used for capture probability. Point estimates of abundance for Asia (combined across the three areas) were relatively consistent across models, ranging from 938 to 1,107. The model that fit the data the best (as selected by AICc) gave an estimate of 1,107 for the Ogasawara Islands, Okinawa, and the Philippines. No confidence limits or coefficients of variation (CVs) were calculated for the SPLASH abundance estimates. Although no other high density aggregations of humpback whales are known on the Asian wintering ground, whales have been seen in other locations, indicating this is likely to represent an underestimate of the stock's true abundance to an unknown degree. This estimate is more than 8 years old and is outdated for use in stock assessments; however, this population increased between estimates for 1991 to 1993 and 2004 to 2006 (Calambokidis et al. 2008), and this is still considered a valid minimum population estimate (NMFS 2016).

On the summer feeding grounds, the initial SPLASH abundance estimates for Kamchatka in Russia ranged from about 100 to 700, suggesting a large portion of the Asian population occurs near Kamchatka. No separate estimates are available for the other areas in Russia, the Gulf of Anadyr and the Commander Islands; abundance from those areas is included in the estimate of abundance for the Bering Sea and Aleutian Islands, which ranged from about 6,000 to 14,000. Abundance estimates for the Gulf of Alaska and for Southeast Alaska/northern British Columbia both ranged from 3,000 to 5,000 (Calambokidis et al. 2008).

Minimum Population Estimate

Point estimates of abundance for Asia ranged from 938 to 1,107 (for 2004 to 2006), but no associated CV was calculated. The 1991 to 1993 abundance estimate for Asia using similar (though likely less) data had a CV of 0.084. Therefore, it is unlikely the CV of a SPLASH estimate would be greater than 0.300. The minimum population estimate (N_{MIN}) for this stock is calculated according to Equation 1 from the potential biological removal (PBR) guidelines (NMFS 2016): $N_{MIN} = N/\exp(0.842\times[\ln(1+[CV(N)]^2)]^{½})$. Using the SPLASH population estimate (N) of 1,107 from the best fit model and an assumed conservative CV(N) of 0.300 would result in an N_{MIN} for this humpback whale stock of 865. The 2016 guidelines for preparing Stock Assessment Reports (NMFS 2016) recommend that N_{MIN} be considered unknown if the abundance estimate is more than 8 years old, unless there is compelling evidence that the stock has not declined since the last estimate. This population increased between estimates for 1991 to 1993 and 2004 to 2006 (Calambokidis et al. 2008), and this is still considered a valid minimum population estimate.

Current Population Trend

The SPLASH abundance estimate for Asia represents a 6.7% annual rate of increase over the 1991 to 1993 abundance estimate (Calambokidis et al. 2008). However, the 1991 to 1993 estimate was for Ogasawara and Okinawa only, whereas the SPLASH estimate includes the Philippines, so the annual rate of increase is unknown.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Utilizing a birth-interval model, Barlow and Clapham (1997) have estimated a population growth rate of 6.5% (SE = 1.2%) for the well-studied humpback whale population in the Gulf of Maine, although there are indications that this rate subsequently slowed (Clapham et al. 2003). Mobley et al. (2001) estimated a trend of 7% per year for 1993-2000 using data from aerial surveys that were conducted in a consistent manner for several years across all of the Hawaiian Islands and were developed specifically to estimate a trend for the Central North Pacific stock. Mizroch et al. (2004) estimated survival rates for North Pacific humpback whales using mark-recapture methods, and a Pradel model fit to data from Hawaii for the years 1980-1996 resulted in an estimated rate of increase of 10% per year (95% CI: 3-16%). For shelf waters of the northern Gulf of Alaska, Zerbini et al. (2006) estimated an annual rate of increase for humpback whales from 1987 to 2003 of 6.6% (95% CI: 5.2-8.6%). The SPLASH abundance estimate for the total North Pacific represents an annual increase of 4.9% over the most complete estimate for the North Pacific for 1991 to 1993. Comparisons of SPLASH abundance estimates for Hawaii to estimates for 1991 to 1993 gave estimates of annual increase that ranged from 5.5 to 6.0% (Calambokidis et al. 2008). No confidence limits were calculated for these rates of increase from SPLASH data.

Estimates of observed rates of increase can be used to estimate maximum net productivity rates (R_{MAX}), although in most cases these estimates may be biased low, as maximum net productivity rates are only achieved at very low population sizes. However, if the observed rates of increase are greater than the default value for R_{MAX}, it would be reasonable to use a higher value based on those observations. The rates of increase summarized above include estimates for the North Pacific of 7%, 10%, and 6.6%. Although there is no estimate of R_{MAX} for just the Western stock (i.e., from trends in abundance in the Asia breeding areas), it is reasonable to assume that R_{MAX} for this stock would be at least 7% based on the other observations from the North Pacific. Until additional data become available for the Western North Pacific humpback whale stock, 7% will be used as R_{MAX} for this stock (NMFS 2016).

POTENTIAL BIOLOGICAL REMOVAL

PBR is defined as the product of the minimum population estimate, one-half the maximum estimated net productivity rate, and a recovery factor: PBR = $N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 0.1, the value for cetacean stocks listed as endangered under the ESA (NMFS 2016; see Status of Stock section below regarding ESA listing status). Using the N_{MIN} of 865 calculated from the SPLASH abundance estimate for 2004 to 2006, of 1,107 with an assumed CV of 0.300, the PBR is calculated to be 3.0 whales (865 × 0.035 × 0.1).

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2014 and 2018 is listed, by marine mammal stock, in Young et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports. Injury events lacking detailed injury information are assigned prorated values following injury determination guidelines described in NMFS (2012). A summary of information used to determine whether an injury was serious or non-serious, as well as a table of prorate values used for large whale reports with incomplete information, is reported in Young et al. (2020). The minimum estimated mean annual level of human-caused mortality and serious injury for Western North Pacific humpback whales between 2014 and 2018 is 2.8 whales: 0.9 in U.S. commercial fisheries, 0.4 in recreational fisheries, 0.4 in unknown (commercial, recreational, or subsistence) fisheries, 0.6 in marine debris, and 0.5 due to other causes (ship strikes and an intentional unauthorized take); however, this estimate is considered a minimum because there are no data concerning fishery-related mortality and serious injury in Japanese, Russian, or international waters. Assignment of mortality and serious injury to both the Western North Pacific and Central North Pacific stocks of humpback whales, when the stock is unknown and events occur within the area where the stocks are known to overlap, may result in overestimating stock specific mortality and serious injury. Potential threats most likely to result in direct human-caused mortality or serious injury of this stock include entanglement in fishing gear and ship strikes due to increased vessel traffic (from increased shipping in higher latitudes with changes in sea-ice coverage).

Fisheries Information

Information for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is available in Appendix 3 of the Alaska Stock Assessment Reports (observer coverage) and in the NMFS List of Fisheries (LOF) and the fact sheets linked to fishery names in the LOF (observer coverage and reported incidental takes of marine mammals: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-list-fisheries, accessed December 2020).

In 2018, one humpback whale mortality occurred in the Bering Sea/Aleutian Islands pollock trawl fishery (Table 1; Breiwick 2013; MML, unpubl. data). Because the stock is unknown, and the event occurred within the area where the Western North Pacific and Central North Pacific stocks are known to overlap, the mortality in this fishery was assigned to both stocks of humpback whales. The minimum estimated mean annual mortality and serious injury rate from observed U.S. commercial fisheries between 2014 and 2018 is 0.2 Western North Pacific humpback whales (Table 1).

Table 1. Summary of incidental mortality and serious injury of Western North Pacific humpback whales due to U.S. commercial fisheries between 2014 and 2018 and calculation of the mean annual mortality and serious injury rate (Breiwick 2013; MML, unpubl. data). Methods for calculating percent observer coverage are described in Appendix 3 of the Alaska Stock Assessment Reports.

Fishery name	Years	Data type	Percent observer coverage	Observed mortality	Estimated mortality (CV)	Mean estimated annual mortality
Bering Sea/Aleutian Is. pollock trawl*	2014 2015 2016 2017 2018	obs data	98 99 99 99 99	0 0 0 0 1	0 0 0 0 1.0 (0.11)	0.2 (CV = 0.11)
Minimum total estimated annual mortality						0.2 (CV = 0.11)

*Mortality and serious injury in this fishery was assigned to both the Western North Pacific and Central North Pacific stocks of humpback whales, because the stock is unknown and the two stocks overlap within the area of operation of the fishery.

Mortality and serious injury reported to the NMFS Alaska Region marine mammal stranding network, for fisheries in which observer data are not available, resulted in a minimum mean annual mortality and serious injury rate of 0.7 humpback whales in U.S. commercial fisheries between 2014 and 2018 (Table 2; Young et al. 2020). Because all of these events occurred in the area where the two stocks overlap, this mortality and serious injury was assigned to both the Western North Pacific and Central North Pacific stocks of humpback whales (NMFS 2016). These mortality and serious injury estimates result from an actual count of verified human-caused deaths and serious injuries and are minimums because not all entangled animals strand or are self-reported nor are all stranded animals found, reported, or have the cause of death determined.

The minimum estimate of the mean annual mortality and serious injury rate incidental to U.S. commercial fisheries between 2014 and 2018 is 0.9 Western North Pacific humpback whales, based on observer data (0.2) and reports to the NMFS Alaska Region stranding network in which the commercial fishery can be confirmed (0.7). However, this estimate is considered a minimum because there are no data concerning fishery-related mortality and serious injury in Japanese, Russian, or international waters.

Reports to the NMFS Alaska Region marine mammal stranding network of swimming, floating, or beachcast humpback whales entangled in fishing gear or with injuries caused by interactions with gear, which may be from commercial, recreational, or subsistence fisheries, are another source of fishery-related mortality and serious injury data (Table 2). These mortality and serious injury estimates result from an actual count of verified human-caused deaths and serious injuries and are minimums because not all entangled animals strand nor are all stranded animals found, reported, or have the cause of death determined. Because all of these events occurred in the area where the two stocks overlap, the mortality and serious injury was assigned to both the Western North Pacific and Central North Pacific stocks of humpback whales. Between 2014 and 2018, two humpback whales (each with a serious injury prorated at 0.75) entangled in recreational pot fisheries gear, resulting in a minimum mean annual mortality and serious injury rate of 0.4 whales in recreational gear (Table 2; Young et al. 2020). Additional entanglements in Prince William Sound shrimp pot gear and unidentified fishing gear resulted in a minimum mean annual mortality and serious injury rate of 0.4 humpback whales in unknown (commercial, recreational, or subsistence) fisheries between 2014 and 2018 (Table 2; Young et al. 2020).

The minimum mean annual mortality and serious injury rate due to interactions with all fisheries between 2014 and 2018 is 1.7 Western North Pacific humpback whales (0.9 in commercial fisheries + 0.4 in recreational fisheries + 0.4 in unknown fisheries).

Table 2. Summary of mortality and serious injury of Western North Pacific humpback whales, by year and type, reported to the NMFS Alaska Region marine mammal stranding network between 2014 and 2018 (Young et al. 2020). All events occurred within the area of known overlap between the Western North Pacific and Central North Pacific humpback whale stocks. Because the stock is unknown, the mortality and serious injury is reflected in the Stock Assessment Reports for both stocks. Injury events lacking detailed injury information are assigned prorated values following injury determination guidelines described in NMFS (2012). A summary of information used to determine whether an injury was serious or non-serious, as well as a table of prorate values used for large whale reports with incomplete information, is reported in Young et al. (2020).

Cause of injury	**2014**	**2015**	**2016**	**2017**	**2018**	**Mean annual mortality**
Entangled in Kodiak Island commercial salmon set gillnet	0	0.75	0	0	0	0.2
Entangled in Bering Sea/Aleutian Is. commercial pot gear	0	1	0	0	0	0.2
Entangled in Alaska State-managed commercial cod pot gear (parallel fishery)	0	0	0	1	0	0.2
Ship strike by AK/WA/OR/CA commercial passenger fishing vessel	0	0	0	0.52	0	0.1
Entangled in Gulf of Alaska recreational Dungeness crab pot gear	0	0.75	0	0	0	0.2
Entangled in Gulf of Alaska recreational shrimp pot gear	0	0.75	0	0	0	0.2
Entangled in Prince William Sound shrimp pot gear*	1	0	0	0	0	0.2
Entangled in unidentified fishing gear*	0	0	0	0	0.75	0.2
Entangled in marine debris	0.75	0	2	0	0	0.6
Ship strike	1.2	0	0.2	0	0	0.3
Intentional unauthorized take	0	0	1	0	0	0.2
Total in commercial fisheries						0.7
Total in recreational fisheries						0.4
*Total in unknown (commercial, recreational, or subsistence) fisheries						0.4
Total in marine debris						0.6
Total due to other causes (ship strike, intentional unauthorized take)						0.5

Brownell et al. (2000) compiled records of bycatch in Japanese and Korean commercial fisheries between 1993 and 2000. From 1995 to 1999, there were six humpback whales indicated as "bycatch." In addition, two strandings were reported during this period. Furthermore, genetic analysis of four samples from meat found in markets indicated that humpback whale meat was being sold. It is not known whether any or all strandings were caused by incidental interactions with commercial fisheries; similarly, it is not known whether the humpback whales identified in market samples were killed as a result of incidental interactions with commercial fisheries. It is also not known which fishery may be responsible for the bycatch. Regardless, these data indicate a minimum mortality level of 1.1 per year (using bycatch data only) to 2.4 per year (using bycatch, stranding, and market data) in the waters of Japan and Korea. Because many mortalities pass unreported, the actual rate in these areas is likely higher. An analysis of entanglement rates from photographs collected for the SPLASH study found a minimum entanglement rate of 31% for humpback whales from the Asia breeding grounds (Cascadia Research NFWF Report #2003-0170-019).

Alaska Native Subsistence/Harvest Information

Subsistence hunters in Alaska are not authorized to take humpback whales from this stock. An intentional unauthorized take of a humpback whale by Alaska Natives in 2016 in Toksook Bay is reported in the Other Mortality section of this report.

Other Mortality

In 2015, increased mortality of large whales was observed along the western Gulf of Alaska (including the areas around Kodiak Island, Afognak Island, Chirikof Island, the Semidi Islands, and the southern shoreline of the Alaska Peninsula) and along the central British Columbia coast (from the northern tip of Haida Gwaii to southern Vancouver Island). NMFS declared an Unusual Mortality Event (UME) for large whales that occurred from 22 May to 31 December 2015 in the western Gulf of Alaska and from 23 April 2015 to 16 April 2016 in British Columbia (https://www.fisheries.noaa.gov/national/marine-life-distress/active-and-closed-unusual-mortality-events, accessed December 2020). Forty-six large whale deaths attributed to the UME included 12 fin whales and 22 humpback whales in Alaska and 5 fin whales and 7 humpback whales in British Columbia. Based on the findings from the investigation, the UME was likely caused by ecological factors (i.e., the 2015 El Niño, Warm Water Blob, and Pacific Coast Domoic Acid Bloom).

Entanglements in marine debris and ship strikes by vessels unrelated to fisheries reported to the NMFS Alaska Region marine mammal stranding network account for minimum mean annual mortality and serious injury rates of 0.6 and 0.3 Western North Pacific humpback whales, respectively, between 2014 and 2018 (Table 2; Young et al. 2020). Because all of these events occurred in the area where the stocks overlap, the mortality and serious injury was assigned to both the Western North Pacific and Central North Pacific stocks of humpback whales. These mortality and serious injury estimates result from an actual count of verified human-caused deaths and serious injuries and are minimums because not all animals strand nor are all stranded animals found, reported, or have the cause of death determined.

An intentional unauthorized take of a humpback whale by Alaska Natives in Toksook Bay in 2016 resulted in a mean annual mortality and serious injury rate of 0.2 whales between 2014 and 2018 (Table 2).

HISTORICAL WHALING

Rice (1978) estimated that the number of humpback whales in the North Pacific may have been approximately 15,000 individuals prior to exploitation; however, this was based upon incomplete data and, given the level of known catches (legal and illegal) since World War II, may be an underestimate. Intensive commercial whaling removed more than 28,000 animals from the North Pacific during the 20th century (Rice 1978). A total of 3,277 reported catches occurred in Asia between 1910 and 1964, with 817 catches from Ogasawara between 1924 and 1944 (Nishiwaki 1966, Rice 1978). After World War II, substantial catches occurred in Asia near Okinawa (including 970 between 1958 and 1961), as well as around the main islands of Japan and the Ogasawara Islands. On the feeding grounds, substantial catches occurred around the Commander Islands and western Aleutian Islands, as well as in the Gulf of Anadyr (Springer et al. 2006).

Humpback whales in the North Pacific were theoretically fully protected in 1965, but illegal catches by the U.S.S.R. continued until 1972 (Ivashchenko et al. 2013). From 1948 to 1971, 7,334 humpback whales were killed by the U.S.S.R., and 2,654 of these were illegally taken and not reported to the IWC (Ivashchenko et al. 2013). Many animals during this period were taken from the Gulf of Alaska and Bering Sea (Doroshenko 2000); additional illegal catches were made across the North Pacific, from the Kuril Islands to Haida Gwaii, and other takes may have gone unrecorded. The Soviet factory ship *Aleut* is known to have taken 535 humpback whales from 1933 to 1947 (Ivashchenko et al. 2013).

STATUS OF STOCK

The minimum estimated mean annual level of human-caused mortality and serious injury of 2.8 Western North Pacific humpback whales is less than the calculated PBR level for this stock (3.0). The minimum estimate of the mean annual U.S. commercial fishery-related mortality and serious injury rate for this stock (0.9 whales) exceeds 10% of the PBR (10% of PBR = 0.3) and cannot be considered insignificant and approaching a zero mortality and serious injury rate. In addition, there is a lack of information about fisheries bycatch from Russia, Japan, Korea, and international waters, as well as earlier evidence of bycatch in Japan and Korea (Brownell et al. 2000: 1.1 to 2.4 whales per year based on bycatch, stranding, and market data). The humpback whale ESA listing final rule (81 FR 62259, 8 September 2016) established 14 Distinct Population Segments (DPSs) with different listing statuses. The DPSs that occur in waters under the jurisdiction of the United States do not equate to the existing MMPA stocks. Some of the listed DPSs partially coincide with the currently defined Western North Pacific stock. Because we cannot manage one portion of an MMPA stock as ESA-listed and another portion of a stock as not ESA-listed, until such time as the MMPA stock delineations are reviewed in light of the DPS designations and Bettridge et al. (2015), NMFS continues to use the existing MMPA stock structure and considers this stock to be endangered and depleted for MMPA management purposes (e.g., selection of a recovery factor, stock status). As a result, the Western North Pacific stock of humpback whales is classified as a strategic stock.

There are key uncertainties in the assessment of the Western North Pacific stock of humpback whales. New DPSs were identified under the ESA; however, stocks have not yet been revised. The feeding areas of the Western North Pacific stock and the Central North Pacific stock overlap in waters from British Columbia to the Bering Sea, so human-related mortality and serious injury estimates must be assigned to or prorated to multiple stocks. The migratory destination of the Western North Pacific stock is not well understood. The population estimate was based on studies from the Asian wintering grounds; although no other large aggregations of whales are known, the estimate is likely conservative relative to the actual abundance. An estimate of variance is not available. The abundance estimate is calculated using data collected from 2004 to 2006; however, the population increased between estimates for 1991 to 1993 and 2004 to 2006 (Calambokidis et al. 2008), and the N_{MIN} is still considered a valid minimum population estimate (NMFS 2016). Estimates of human-caused mortality and serious injury from stranding data and fisherman self-reports are underestimates because not all animals strand or are self-reported nor are all stranded animals found, reported, or have the cause of death determined.

HABITAT CONCERNS

Potential concerns for this stock include elevated levels of sound from anthropogenic sources (e.g., shipping, military sonars), harmful algal blooms (Geraci et al. 1989), possible changes in prey distribution with climate change, entanglement in fishing gear, ship strikes due to increased vessel traffic (e.g., from increased shipping in higher latitudes), and oil and gas activities.

The overall trend for most humpback whale populations found in U.S. waters is positive and points toward recovery (81 FR 62259; 8 September 2016); however, this may not be uniform for all breeding areas. A sharp decline in observed reproduction and encounter rates of humpback whales from the Central North Pacific stock between 2013 and 2018 has been related to oceanographic anomalies and consequent impacts on prey resources (Cartwright et al. 2019), suggesting that humpback whales are vulnerable to major environmental changes.

CITATIONS

Baker, C. S., S. R. Palumbi, R. H. Lambertsen, M. T. Weinrich, J. Calambokidis, and S. J. O'Brien. 1990. Influence of seasonal migration on geographic distribution of mitochondrial DNA haplotypes in humpback whales. Nature 344:238-240.

Baker, C. S., L. Medrano-Gonzalez, J. Calambokidis, A. Perry, F. Pichler, H. Rosenbaum, J. M. Straley, J. Urban-Ramirez, M. Yamaguchi, and O. von Ziegesar. 1998. Population structure of nuclear and mitochondrial DNA variation among humpback whales in the North Pacific. Mol. Ecol. 7:695-707.

Baker, C. S., D. Steel, J. Calambokidis, J. Barlow, A. M. Burdin, P. J. Clapham, E. A. Falcone, J. K. B. Ford, C. M. Gabriele, U. González-Peral, R. LeDuc, D. Mattila, T. J. Quinn, L. Rojas-Bracho, J. M. Straley, B. L. Taylor, J. Urbán-R., M. Vant, P. Wade, D. Weller, B. H. Witteveen, K. Wynne, and M. Yamaguchi. 2008. *geneSPLASH*: an initial, ocean-wide survey of mitochondrial (mt) DNA diversity and population structure among humpback whales in the North Pacific. Final Report for Contract 2006-0093-008 to the National Fish and Wildlife Foundation.

Baker, C. S., D. Steel, J. Calambokidis, E. Falcone, U. González-Peral, J. Barlow, A. M. Burdin, P. J. Clapham, J. K. B. Ford, C. M. Gabriele, D. Mattila, L. Rojas-Bracho, J. M. Straley, B. L. Taylor, J. Urbán, P. R. Wade, D. Weller, B. H. Witteveen, and M. Yamaguchi. 2013. Strong maternal fidelity and natal philopatry shape genetic structure in North Pacific humpback whales. Mar. Ecol. Prog. Ser. 494:291-306. DOI: dx.doi.org/10.3354/meps10508 .

Barlow, J., and P. J. Clapham. 1997. A new birth-interval approach to estimating demographic parameters of humpback whales. Ecology 78(2):535-546.

Barlow, J., J. Calambokidis, E. A. Falcone, C. S. Baker, A. M. Burdin, P. J. Clapham, J. K. B. Ford, C. M. Gabriele, R. LeDuc, D. K. Mattila, T. J. Quinn II, L. Rojas-Bracho, J. M. Straley, B. L. Taylor, J. Urbán R., P. Wade, D. Weller, B. H. Witteveen, and M. Yamaguchi. 2011. Humpback whale abundance in the North Pacific estimated by photographic capture-recapture with bias correction from simulation studies. Mar. Mammal Sci. 27:793-818.

Bettridge, S., C. S. Baker, J. Barlow, P. J. Clapham, M. Ford, D. Gouveia, D. K. Mattila, R. M. Pace III, P. E. Rosel, G. K. Silber, and P. R. Wade. 2015. Status review of the humpback whale (*Megaptera novaeangliae*) under the Endangered Species Act. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-SWFSC-540, 240 p.

Breiwick, J. M. 2013. North Pacific marine mammal bycatch estimation methodology and results, 2007-2011. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-260, 40 p.

Brownell, R. L., T. Kasuya, W. P. Perrin, C. S. Baker, F. Cipriano, J. Urban R., D. P. DeMaster, M. R. Brown, and P. J. Clapham. 2000. Unknown status of the western North Pacific humpback whale population: a new conservation concern. Unpubl. report to the International Whaling Commission. 5 p.

Calambokidis, J., G. H. Steiger, J. C. Cubbage, K. C. Balcomb III, and P. Bloedel. 1989. Biology of humpback whales in the Gulf of the Farallones. Report to Gulf of the Farallones National Marine Sanctuary, San Francisco, CA, by Cascadia Research Collective, 218½ West Fourth Avenue, Olympia, WA. 93 p.

Calambokidis, J., G. H. Steiger, and J. R. Evenson. 1993. Photographic identification and abundance estimates of humpback and blue whales off California in 1991-92. Final Contract Report 50ABNF100137 to Southwest Fisheries Science Center, 8901 La Jolla Shores Drive, La Jolla, CA 92037. 67 p.

Calambokidis, J., G. H. Steiger, J. M. Straley, T. Quinn, L. M. Herman, S. Cerchio, D. R. Salden, M. Yamaguchi, F. Sato, J. Urban R., J. Jacobson, O. von Ziegesar, K. C. Balcomb, C. M. Gabriele, M. E. Dahlheim, N. Higashi, S. Uchida, J. K. B. Ford, Y. Miyamura, P. Ladrón de Guevara, S. A. Mizroch, L. Schlender, and K. Rasmussen. 1997. Abundance and population structure of humpback whales in the North Pacific basin. Final Contract Report 50ABNF500113 to Southwest Fisheries Science Center, 8901 La Jolla Shores Drive, La Jolla, CA 92037. 72 p.

Calambokidis, J., G. H. Steiger, J. M. Straley, L. M. Herman, S. Cerchio, D. R. Salden, J. Urbán R., J. K. Jacobsen, O. von Ziegesar, K. C. Balcomb, C. M. Gabriele, M. E. Dahlheim, S. Uchida, G. Ellis, Y. Miyamura, P. Ladrón de Guevara P., M. Yamaguchi, F. Sato, S. A. Mizroch, L. Schlender, K. Rasmussen, J. Barlow, and T. J. Quinn II. 2001. Movements and population structure of humpback whales in the North Pacific. Mar. Mammal Sci. 17(4):769-794.

Calambokidis, J., E. A. Falcone, T. J. Quinn, A. M. Burdin, P. J. Clapham, J. K. B. Ford, C. M. Gabriele, R. LeDuc, D. Mattila, L. Rojas-Bracho, J. M. Straley, B. L. Taylor, J. Urbán R., D. Weller, B. H. Witteveen, M. Yamaguchi, A. Bendlin, D. Camacho, K. Flynn, A. Havron, J. Huggins, and N. Maloney. 2008. SPLASH: Structure of Populations, Levels of Abundance and Status of Humpback Whales in the North Pacific. Final Report for Contract AB133F-03-RP-00078, U.S. Dep. Commer., Western Administrative Center, Seattle, WA. Available online: http://www.cascadiaresearch.org/files/publications/SPLASH-contract-Report-May08.pdf . Accessed December 2020.

Cartwright, R., A. Venema, V. Hernandez, C. Wyels, J. Cesere, and D. Cesere. 2019. Fluctuating reproductive rates in Hawaii's humpback whales, *Megaptera novaeangliae*, reflect recent climate anomalies in the North Pacific. Roy. Soc. Open Sci. 6:181463. DOI: dx.doi.org/10.1098/rsos.181463 .

Clapham, P. J., and J. G. Mead. 1999. *Megaptera novaeangliae*. Mamm. Species 604:1-9.

Clapham, P. J., J. Barlow, M. Bessinger, T. Cole, D. Mattila, R. Pace, D. Palka, J. Robbins, and R. Seton. 2003. Abundance and demographic parameters of humpback whales from the Gulf of Maine, and stock definition relative to the Scotian Shelf. J. Cetacean Res. Manage. 5:13-22.

Clarke, J. T., C. L. Christman, A. A. Brower, and M. C. Ferguson. 2013a. Distribution and relative abundance of marine mammals in the northeastern Chukchi and western Beaufort seas, 2012. Annual Report, OCS Study BOEM 2013-00117. Marine Mammal Laboratory, AFSC, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Clarke, J., K. Stafford, S. E. Moore, B. Rone, L. Aerts, and J. Crance. 2013b. Subarctic cetaceans in the southern Chukchi Sea: evidence of recovery or response to a changing ecosystem. Oceanography 26(4):136-149.

Darling, J. D. 1991. Humpback whales in Japanese waters. Ogasawara and Okinawa. Fluke identification catalog 1987-1990. Final Contract Report, World Wide Fund for Nature, Japan. 22 p.

Darling, J. D., and S. Cerchio. 1993. Movement of a humpback whale (*Megaptera novaeangliae*) between Japan and Hawaii. Mar. Mammal Sci. 1:84-89.

Darling, J. D., J. Calambokidis, K. C. Balcomb, P. Bloedel, K. Flynn, A. Mochizuki, K. Mori, F. Sato, H. Suganuma, and M. Yamaguchi. 1996. Movement of a humpback whale (*Megaptera novaeangliae*) from Japan to British Columbia and return. Mar. Mammal Sci. 12(2):281-287.

Doroshenko, N. V. 2000. Soviet catches of humpback whales (*Megaptera novaeangliae*) in the North Pacific, p. 48-95. *In* A. V. Yablokov and V. A. Zemsky (eds.), Soviet Whaling Data (1949-1979). Center for Russian Environmental Policy, Marine Mammal Council, Moscow.

Fleming, A., and J. Jackson. 2011. Global review of humpback whales (*Megaptera novaeangliae*). U.S. Dep. Commer., NOAA Tech. Memo. NMFS-SWFSC-474, 206 p.

Geraci, J. R., D. M. Anderson, R. J. Timperi, D. St. Aubin, G. A. Early, J. H. Prescott, and C. A. Mayo. 1989. Humpback whales (*Megaptera novaeangliae*) fatally poisoned by dinoflagellate toxin. Can. J. Fish. Aquat. Sci. 46(11):1895-1898. DOI: dx.doi.org/10.1139/f89-238 .

Hashagen, K. A., G. A. Green, and B. Adams. 2009. Observations of humpback whales, *Megaptera novaeangliae*, in the Beaufort Sea, Alaska. Northwest. Nat. 90:160-162.

Hill, M. C., E. M. Oleson, S. Baumann-Pickering, A. M. Van Cise, A. D. Ligon, A. R. Bendlin, A. C. Ü, J. S. Trickey, and A. L. Bradford. 2016. Cetacean monitoring in the Mariana Islands Range Complex, 2015. Prepared for the U.S. Pacific Fleet Environmental Readiness Office. PIFSC Data Report DR-16-01. 36 p. + appendix.

Ivashchenko, Y. V., R. L. Brownell, Jr., and P. J. Clapham. 2013. Soviet whaling in the North Pacific: revised catch totals. J. Cetacean Res. Manage. 13:59-71.

Johnson, J. H., and A. A. Wolman. 1984. The humpback whale, *Megaptera novaeangliae*. Mar. Fish. Rev. 46(4):30-37.

Johnston, D. W., M. E. Chapla, L. E. Williams, and D. K. Mattila. 2007. Identification of humpback whale *Megaptera novaeangliae* wintering habitat in the Northwestern Hawaiian Islands using spatial habitat modeling. Endang. Species Res. 3:249-257.

Melnikov, V. V. 2000. Humpback whales *Megaptera novaeangliae* off Chukchi Peninsula. Russ. J. Oceanogr. 4:844-849.

Miyashita, T. 2006. Cruise report of the sighting survey in the waters east of the Kuril Islands and the Kamchatka Peninsula in 2005. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/58/NPM5).

Mizroch, S. A., and D. W. Rice. 2007. Distribution and movements of humpback whales (*Megaptera novaeangliae*) in the North Pacific Ocean. Presented at the 17th Biennial Conference on the Biology of Marine Mammals, Cape Town, South Africa.

Mizroch, S. A., L. M. Herman, J. M. Straley, D. Glockner-Ferrari, C. Jurasz, J. Darling, S. Cerchio, C. Gabriele, D. Salden, and O. von Ziegesar. 2004. Estimating the adult survival rate of central North Pacific humpback whales. J. Mammal. 85(5):963-972.

Mobley, J. M., S. Spitz, R. Grotefendt, P. Forestell, A. Frankel, and G. Bauer. 2001. Abundance of humpback whales in Hawaiian waters: Results of 1993-2000 aerial surveys. Report to the Hawaiian Islands Humpback Whale National Marine Sanctuary. 16 p.

National Marine Fisheries Service (NMFS). 2012. Process for distinguishing serious from non-serious injury of marine mammals. 42 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-policies-guidance-and-regulations . Accessed December 2020.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2020.

Nemoto, T. 1957. Foods of baleen whales in the northern Pacific. Sci. Rep. Whales Res. Inst. Tokyo 12:33-89.

Nemoto, T. 1959. Food of baleen whales with reference to whale movements. *S*ci. Rep. Whales Res. Inst. 14**:**149-290.

Nishiwaki, M. 1966. Distribution and migration of the larger cetaceans in the North Pacific as shown by Japanese whaling results, p. 172-191. *In* K. S. Norris (ed.), Whales, Dolphins and Porpoises. University of California Press, Berkeley, CA.

Ohsumi, S., and Y. Masaki. 1975. Japanese whale marking in the North Pacific, 1963-1972. Bull. Far Seas Fish. Res. 12:171-219.

Omura, H., and S. Ohsumi. 1964. A review of Japanese whale marking in the North Pacific to the end of 1962, with some information on marking in the Antarctic. Norsk Hvalfangst-Tidende 4:90-112.

Perry, A., C. S. Baker, and L. M. Herman. 1990. Population characteristics of individually identified humpback whales in the central and eastern North Pacific: a summary and critique. Rep. Int. Whal. Comm. (Special Issue 12):307-317.

Rice, D. W. 1978. The humpback whale in the North Pacific: distribution, exploitation and numbers, Appendix 4, p. 29-44. *In* K. S. Norris and R. R. Reeves (eds.), Report on a workshop on problems related to humpback whales (*Megaptera novaeangliae*) in Hawaii. U.S. Dep. Commer., Nat. Tech. Info. Serv. PB-280 794. Springfield, VA.

Rice, D. W. 1998. Marine Mammals of the World: Systematics and Distribution. Soc. Mar. Mammal. Spec. Publ. No. 4.

Springer, A. M., G. B. van Vliet, J. F. Piatt, and E. M. Danner. 2006. Whales and whaling in the North Pacific Ocean and Bering Sea: oceanographic insights and ecosystem impacts, p. 245-261. *In* J. A. Estes, R. L. Brownell, Jr., D. P DeMaster, D. F. Doak, and T. M. Williams (eds.), Whales, Whaling and Ocean Ecosystems. University of California Press. 418 p.

Steiger, G. H., J. Calambokidis, R. Sears, K. C. Balcomb, and J. C. Cubbage. 1991. Movement of humpback whales between California and Costa Rica. Mar. Mammal Sci. 7:306-310.

Tomlin, A. G. 1967. Mammals of the USSR and adjacent countries. Vol. 9. Cetacea. Israel Program for Scientific Translations No. 1124, Natl. Tech. Info. Serv. TT 65-50086. Springfield, VA. 717 p. (Translation of Russian text published in 1957.)

Witteveen, B. H., J. M. Straley, O. von Ziegesar, D. Steel, and C. S. Baker. 2004. Abundance and mtDNA differentiation of humpback whales (*Megaptera novaeangliae*) in the Shumagin Islands, Alaska. Can. J. Zool. 82:1352-1359.

Young, N. C., B. J. Delean, V. T. Helker, J. C. Freed, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2014-2018. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-413, 142 p.

Zenkovich, B. A. 1954. Vokrug sveta za kitami, Vol. Gosudarstvennoe Izdatel’stvo Geograficheskoi Literatury, Moscow.

Zerbini, A. N., J. M. Waite, J. L. Laake, and P. R. Wade. 2006. Abundance, trends and distribution of baleen whales off western Alaska and the central Aleutian Islands. Deep-Sea Res. I 53(11):1772-1790.

Revised 12/30/2020

HUMPBACK WHALE (*Megaptera novaeangliae*): Central North Pacific Stock

NOTE – NMFS is in the process of reviewing humpback whale stock structure under the Marine Mammal Protection Act (MMPA) in light of the 14 Distinct Population Segments established under the Endangered Species Act (ESA) (81 FR 62259, 8 September 2016). A complete revision of the humpback whale stock assessments will be postponed until this review is complete. In the interim, new information on humpback whale mortality and serious injury is provided within this report.

STOCK DEFINITION AND GEOGRAPHIC RANGE

The humpback whale is distributed worldwide in all ocean basins. In winter, most humpback whales occur in the subtropical and tropical waters of the Northern and Southern Hemispheres. Humpback whales in the high latitudes of the North Pacific Ocean are seasonal migrants that feed on euphausiids and small schooling fishes (Nemoto 1957, 1959; Clapham and Mead 1999). The humpback whale population was considerably reduced as a result of intensive commercial exploitation during the 20th century.

A large-scale study of humpback whales throughout the North Pacific was conducted from 2004 to 2006 (the Structure of Populations, Levels of Abundance, and Status of Humpbacks (SPLASH) project). Results from this project (Calambokidis et al. 2008, Barlow et al. 2011), including abundance estimates and movement information, have been reported in Baker et al. (2008, 2013) and are also summarized in Fleming and Jackson (2011); however, these results are still being considered for stock structure analysis.

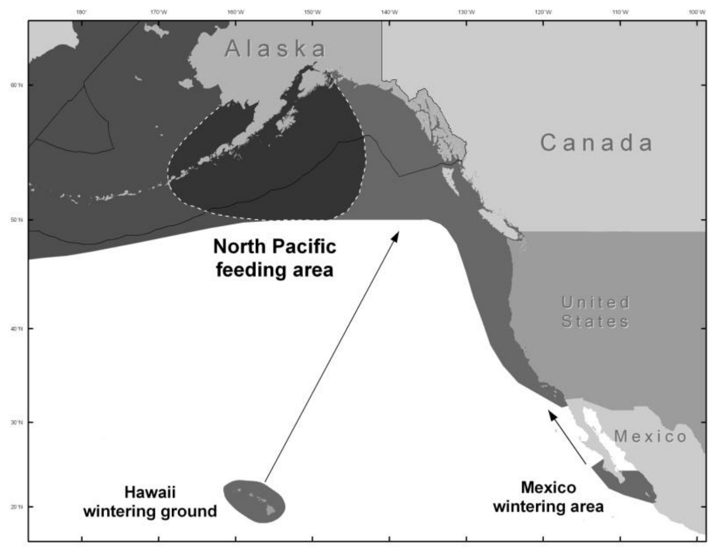

Figure 1. Approximate distribution of humpback whales in the eastern North Pacific (dark shaded areas). Feeding and wintering areas are presented above (see text). Area within the dotted line is known to be an area where the Central North Pacific and Western North Pacific stocks overlap. See Figure 1 in the Western North Pacific humpback whale Stock Assessment Report for distribution of humpback whales in the western North Pacific.

The historical summer feeding range of humpback whales in the North Pacific encompassed coastal and inland waters around the Pacific Rim from Point Conception, California, north to the Gulf of Alaska and the Bering Sea, and west along the Aleutian Islands to the Kamchatka Peninsula and into the Sea of Okhotsk and north of the Bering Strait (Zenkovich 1954, Nemoto 1957, Tomlin 1967, Johnson and Wolman 1984). Historically, the Asian wintering area extended from the South China Sea east through the Philippines, Ryukyu Retto, Ogasawara Gunto, Mariana Islands, and Marshall Islands (Rice 1998). Humpback whales are currently found throughout this historical range (Clarke et al. 2013), with sightings during summer months occurring as far north as the Beaufort Sea (Hashagen et al. 2009). Most of the current winter range of humpback whales in the North Pacific is relatively well known, with aggregations of whales in Japan, the Philippines, Hawaii, Mexico, and Central America. The winter range includes the main islands of the Hawaiian archipelago, with the greatest concentration along the west side of Maui. In Mexico, the winter breeding range includes waters around the southern part of the Baja California peninsula, the central portions of the Pacific coast of mainland Mexico, and the Revillagigedo Islands off the mainland coast. The winter range also extends from southern Mexico into Central America, including Guatemala, El Salvador, Nicaragua, and Costa Rica (Calambokidis et al. 2008).

Photo-identification data, distribution information, and genetic analyses have indicated that in the North Pacific there are at least three breeding populations (Asia, Hawaii, and Mexico/Central America) that all migrate between their respective winter/spring calving and mating areas and their summer/fall feeding areas (Calambokidis et al. 1997, Baker et al. 1998). Calambokidis et al. (2001) further suggested that there may be as many as six subpopulations on the wintering grounds. From photo-identification and Discovery tag information there are known connections between Asia and Russia, between Hawaii and Alaska, and between Mexico/Central America and California (Darling 1991, Darling and Cerchio 1993, Calambokidis et al. 1997, Baker et al. 1998). This information led to the designation of three stocks of humpback whales in the North Pacific: 1) the California/Oregon/Washington and Mexico stock, consisting of winter/spring populations in coastal Central America and coastal Mexico which migrate to the coast of California and as far north as southern British Columbia in summer/fall (Calambokidis et al. 1989, 1993; Steiger et al. 1991); 2) the Central North Pacific stock, consisting of winter/spring populations of the Hawaiian Islands which migrate primarily to northern British Columbia/Southeast Alaska, the Gulf of Alaska, and the Bering Sea/Aleutian Islands (Baker et al. 1990, Perry et al. 1990, Calambokidis et al. 1997) (Fig. 1) ; and 3) the Western North Pacific stock, consisting of winter/spring populations off Asia which migrate primarily to Russia and the Bering Sea/Aleutian Islands.

Information from the SPLASH project largely confirms this view of humpback whale distribution and movements in the North Pacific. For example, the SPLASH results confirm low rates of interchange between the three principal wintering regions (Asia, Hawaii, and Mexico). However, the full SPLASH results suggest that the current view of population structure is incomplete. The overall pattern of movements is complex but indicates a high degree of population structure. Whales from wintering areas at the extremes of their range on both sides of the Pacific migrate to coastal feeding areas that are on the same side of the Pacific: whales from Asia in the west migrate to Russia and whales from mainland Mexico and Central America in the east migrate to coastal waters off California/Oregon.

The SPLASH data now show that Revillagigedo whales are seen in all sampled feeding areas except northern California/Oregon and the south side of the Aleutians. They are primarily distributed in the Bering Sea, Gulf of Alaska, and Southeast Alaska/northern British Columbia but are also found in Russia and southern British Columbia/Washington. The migratory destinations of humpback whales from Hawaii were found to be quite similar, and a significant number of matches (14) were seen during SPLASH between Hawaii and the Revillagigedos (Calambokidis et al. 2008). The SPLASH project also found that whales from the Aleutian Islands and Bering Sea, and perhaps the Gulf of Anadyr and the Chukotka Peninsula on the west side of the Bering Strait in Russia, have an unusually low resighting rate in winter areas compared to whales from other feeding areas. It is now believed that some of these whales have a winter migratory destination that was not sampled during the SPLASH project. Given the location of these feeding areas, the most parsimonious explanation would be that some of these whales winter somewhere between Hawaii and Asia, which would include the possibility of the Mariana Islands (southwest of the Ogasawara Islands), the Marshall Islands (approximately half-way between the Mariana Islands and the Hawaiian Islands), and the Northwestern Hawaiian Islands. Subsequent to the SPLASH project, a survey in 2007 documented humpback whales from a number of locations in the Northwestern Hawaiian Islands at relatively low densities (Johnston et al. 2007), but no sampling occurred there during the SPLASH project. Some humpback whales, including mother/calf pairs, have also been found in the Mariana Islands (Hill et al. 2016). Both of these locations are plausible migratory destinations for whales from the Aleutian Islands and Bering Sea. Which stock that whales in these locations would belong to is currently unknown.

The winter distribution of the Central North Pacific stock is primarily in the Hawaiian archipelago. In the SPLASH study, sampling occurred on Kauai, Oahu, Penguin Bank (off the southwest tip of the island of Molokai), Maui, and the island of Hawaii (the Big Island). Interchange within Hawaii was extensive. Although most of the Hawaii identifications came from the Maui sub-area, identifications from the Big Island and Kauai at the eastern and western end of the region showed a high rate of interchange with Maui.

In summer, the majority of whales from the Central North Pacific stock are found in the Aleutian Islands, Bering Sea, Gulf of Alaska, and Southeast Alaska/northern British Columbia. High densities of humpback whales are found in the eastern Aleutian Islands, particularly along the northern side of Unalaska Island, and along the Bering Sea shelf edge and break to the north towards the Pribilof Islands. Small numbers of humpback whales are known from a few locations not sampled during the SPLASH study, including northern Bristol Bay and the Chukchi and Beaufort seas. In the Gulf of Alaska, high densities of humpback whales are found in the Shumagin Islands, south and east of Kodiak Island, and from the Barren Islands through Prince William Sound. Although densities in any particular location are not high, humpback whales are also found in deep waters south of the continental shelf from the eastern Aleutians through the Gulf of Alaska. Relatively high densities of humpback whales occur throughout much of Southeast Alaska and northern British Columbia.

NMFS conducted a global Status Review of humpback whales (Bettridge et al. 2015) and revised the ESA listing of the species (81 FR 62259, 8 September 2016); the effects of the ESA-listing final rule on the status of the stock are discussed below.

POPULATION SIZE

Prior to the SPLASH study, the most complete estimate of abundance for humpback whales in the North Pacific was from data collected from 1991 to 1993, with a best mark-recapture estimate of 6,010 (coefficient of variation (CV) = 0.08) for the entire North Pacific, using a winter-to-winter comparison (Calambokidis et al. 1997). Estimates for Hawaii and Mexico were higher, using marks from summer feeding areas with recaptures on the winter grounds, and totaled almost 10,000 summed across all winter areas. In the SPLASH study, fluke photographs were collected by over 400 researchers in all known feeding areas from Russia to California and in all known wintering areas from Okinawa and the Philippines to the coast of Central America and Mexico from 2004 to 2006. Over 18,000 fluke identification photographs were collected, and these have been used to estimate the abundance of humpback whales in the entire North Pacific Basin. Based on a comparison of all winter identifications to all summer identifications, the Chapman-Petersen estimate of abundance is 21,808 (CV = 0.04) (Barlow et al. 2011). A simulation study identifies significant biases in this estimate from violations of the closed population assumption (+5.3%), exclusion of calves (-10.3%), failure to achieve random geographic sampling (+1.5%), and missed matches (+9.8%) (Barlow et al. 2011). Sex-biased sampling favoring males in wintering areas does not add significant bias if both sexes are proportionately sampled in the feeding areas. The bias-corrected estimate is 20,800 after accounting for a net positive bias of 4.8%. This estimate is likely to be lower than the true abundance due to two additional sources of bias: individual heterogeneity in the probability of being sampled (unquantified) and the likely existence of an unknown and unsampled wintering area (-7.2%).

The Central North Pacific stock of humpback whales winters in Hawaiian waters (Baker et al. 1986). Preliminary mark-recapture abundance estimates from the SPLASH data were calculated in Calambokidis et al. (2008), using a multistrata Hilborn model. The best estimate for Hawaii (as chosen by AICc) was 10,103; no confidence limit or coefficient of variation (CV) was calculated for that estimate. This estimate is more than 8 years old and is outdated for use in stock assessments; however, because this population is increasing in localized areas in Alaska, e.g., Prince William Sound (Teerlink et al. 2015), this is still considered a valid minimum population estimate (NMFS 2016).

In the SPLASH study, the number of unique identifications in different regions during 2004 and 2005 included 63 in the Aleutian Islands (defined as everything on the south side of the islands), 491 in the Bering Sea, 301 in the western Gulf of Alaska (including the Shumagin Islands), and 1,038 in the northern Gulf of Alaska (including Kodiak and Prince William Sound), with a few whales seen in more than one area (Calambokidis et al. 2008). The SPLASH combined estimates ranged from 6,000 to 19,000 for the Aleutian Islands, Bering Sea, and Gulf of Alaska, a considerable increase from previous estimates that were available (e.g., Waite et al. 1999, Moore et al. 2002, Witteveen et al. 2004, Zerbini et al. 2006). However, the SPLASH surveys covered areas not covered in those previous surveys, such as parts of Russian waters (Gulf of Anadyr and Commander Islands), the western and central Aleutian Islands, offshore waters in the Gulf of Alaska and Aleutian Islands, and Prince William Sound. Additionally, mark-recapture estimates can be higher than line-transect estimates because they estimate the total number of whales that have used the study area during the study period, whereas, line-transect surveys provide a snapshot of average abundance in the survey area at the time of the survey. For the Aleutian Islands and Bering Sea (including the Commander Islands and Gulf of Anadyr in Russia), the SPLASH estimates ranged from 2,889 to 13,594; for the Gulf of Alaska (from Prince William Sound to the Shumagin Islands, including Kodiak Island), the SPLASH estimates ranged from 2,845 to 5,122. Given known overlap in the distribution of the Western and Central North Pacific humpback whale stocks, estimates for these feeding areas may include whales from the Western North Pacific stock.

The SPLASH study showed a relatively high rate of interchange between Southeast Alaska and northern British Columbia, so they are considered together. Humpback whale studies have been conducted since the late 1960s in Southeast Alaska. Straley et al. (2009) examined data for the northern portion of Southeast Alaska from 1994 to 2000 and provided an updated abundance estimate of 961 (CV = 0.12). Using 1992 to 2006 photo-identification data and an SIR Jolly-Seber model, Ford et al. (2009) estimated an abundance of 2,145 humpback whales (95% CI: 1,970-2,331) in British Columbia waters. During the SPLASH study, 1,115 unique identifications were made in Southeast Alaska and 583 in northern British Columbia, for a total of 1,669 individual whales, after subtracting whales seen in both areas (1,115+583-13-16 = 1,669) (Calambokidis et al. 2008). From the SPLASH study, the estimates of abundance for Southeast Alaska/northern British Columbia ranged from 2,883 to 6,414. The estimates from SPLASH are considerably larger than the estimate from Straley et al. (2009). This is because the SPLASH estimates included areas not part of the Straley et al. (2009) estimate, including southern Southeast Alaska, northern British Columbia, and offshore waters of both British Columbia and Southeast Alaska.

Minimum Population Estimate

A total of 2,367 unique individuals were seen in the Hawaiian wintering areas during the 2-year period (3 winter field seasons, 2004 to 2006) of the SPLASH study. As discussed above, point estimates of abundance for Hawaii from SPLASH ranged from 7,469 to 10,103: the estimate from the best model was 10,103, but no associated CV was calculated. The 1991 to 1993 abundance estimate for Hawaii using similar (but less) data had a CV of 0.095. Therefore, it is unlikely the CV of a SPLASH estimate would be greater than 0.300. The minimum population estimate (N_{MIN}) for this stock is calculated according to Equation 1 from the potential biological removal (PBR) guidelines (NMFS 2016): $N_{MIN} = N/\exp(0.842\times[\ln(1+[CV(N)]^2)]^{½})$. Using the population estimate (N) of 10,103 from the best fit model and an assumed conservative CV(N) of 0.300 results in an N_{MIN} for the Central North Pacific humpback whale stock of 7,891. The 2016 guidelines for preparing Stock Assessment Reports (NMFS 2016) recommend that N_{MIN} be considered unknown if the abundance estimate is more than 8 years old, unless there is compelling evidence that the stock has not declined since the last estimate. Because this population is increasing in localized areas in Alaska, e.g., Prince William Sound (Teerlink et al. 2015), this is still considered a valid minimum population estimate.

Although the Southeast Alaska/northern British Columbia feeding aggregation is not formally considered a stock, the calculation of what a PBR would be for this area is useful for management purposes. The total number of unique individuals seen during the SPLASH study was 1,669 (1,115 in Southeast Alaska). The abundance estimate of Straley et al. (2009) had a CV of 0.12, and the SPLASH abundance estimates are unlikely to have a much higher CV. Using the lowest population estimate (N) of 2,883 and an assumed worst case CV(N) of 0.300, N_{MIN} for this aggregation is 2,252. Similarly, for the Aleutian Islands and Bering Sea, using the lowest SPLASH estimate of 2,889 with an assumed worst-case CV of 0.300 results in an N_{MIN} of 2,256. For the Gulf of Alaska (from Prince William Sound to the Shumagin Islands, including Kodiak Island), using the lowest SPLASH estimate of 2,845 with an assumed worst-case CV of 0.300 results in an N_{MIN} of 2,222. Estimates for these feeding areas may include whales from the Western North Pacific stock and the Mexican breeding population.

Current Population Trend

Comparison of the estimate for the entire stock provided by Calambokidis et al. (1997) with the 1981 estimate of 1,407 (95% CI: 1,113-1,701) from Baker et al. (1987) suggests that abundance increased in Hawaii between the early 1980s and early 1990s. Mobley et al. (2001) estimated a trend of 7% per year for 1993 to 2000 using data from aerial surveys that were conducted in a consistent manner for several years across all of the Hawaiian Islands and were developed specifically to estimate a trend for the Central North Pacific stock. Mizroch et al. (2004) estimated survival rates for North Pacific humpback whales using mark-recapture methods, and a Pradel model fit to data from Hawaii for the years 1980 to 1996 resulted in an estimated rate of increase of 10% per year (95% CI: 3-16%). For shelf waters of the northern Gulf of Alaska, Zerbini et al. (2006) estimated an annual rate of increase for humpback whales from 1987 to 2003 of 6.6% (95% CI: 5.2-8.6%). The SPLASH abundance estimate for the total North Pacific represents an annual increase of 4.9% over the most complete estimate for the North Pacific for 1991 to 1993. Comparisons of SPLASH abundance estimates for Hawaii to estimates for 1991 to 1993 gave estimates of annual increase that ranged from 5.5 to 6.0% (Calambokidis et al. 2008). No confidence limits were calculated for these rates of increase from SPLASH data. It is also clear that the abundance has increased in Southeast Alaska, although a trend for the Southeast Alaska portion of this stock cannot be estimated from the data because of differences in methods and areas covered.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Using a birth-interval model, Barlow and Clapham (1997) have estimated a population growth rate of 6.5% (SE = 1.2%) for the well-studied humpback whale population in the Gulf of Maine, although there are indications that this rate has slowed over the last decade (Clapham et al. 2003). Estimated rates of increase for the Central North Pacific stock include values for Hawaii of 7.0% (from aerial surveys), 5.5-6.0% (from mark-recapture abundance estimates), and 10% (95% CI: 3-16%) (from a model fit to mark-recapture data) and a value for the northern Gulf of Alaska of 6.6% (95% CI: 5.2-8.6%) from ship surveys (Calambokidis et al. 2008). Although there is no estimate of the maximum net productivity rate (R_{MAX}) for the Central North Pacific stock, it is reasonable to assume that R_{MAX} for this stock would be at least 7%. Until additional data become available for the Central North Pacific humpback whale stock, 7% will be used as R_{MAX} for this stock.

POTENTIAL BIOLOGICAL REMOVAL

PBR is defined as the product of the minimum population estimate, one-half the maximum estimated net productivity rate, and a recovery factor: $PBR = N_{MIN} \times 0.5R_{MAX} \times F_R$. The default recovery factor (F_R) for this stock is 0.1, the recommended value for cetacean stocks listed as endangered under the ESA (NMFS 2016; see Status of Stock section below regarding ESA listing status); however, a recovery factor of 0.3 is used in calculating the PBR for this stock based on the suggested guidelines of Taylor et al. (2003). The default value of 4% for R_{MAX} is replaced by 7%, which is the best estimate of the current rate of increase and is considered a conservative estimate of R_{MAX}. For the Central North Pacific stock of humpback whales, using the SPLASH study abundance estimate from the best fit model for 2004 to 2006 for Hawaii of 10,103 with an assumed CV of 0.300 and its associated N_{MIN} of 7,891, PBR is calculated to be 83 whales (7,891 × 0.035 × 0.3).

At this time, stock structure of humpback whales is under consideration and revisions may be proposed within the next few years. For informational purposes, PBR calculations are completed here for the feeding area aggregations. For Southeast Alaska and northern British Columbia, the smallest abundance estimates from the SPLASH study were used with an assumed worst-case CV of 0.300 to calculate PBRs for feeding areas. Using the suggested guidelines presented in Taylor et al. (2003), it would be appropriate to use a recovery factor of 0.3 for the Southeast Alaska/northern British Columbia feeding aggregation because this aggregation has an N_{MIN} greater than 1,500 and less than 5,000 and has an increasing population trend. A recovery factor of 0.1 is appropriate for the Aleutian Islands and Bering Sea feeding aggregation and the Gulf of Alaska feeding aggregation because the N_{MIN} is greater than 1,500 and less than 5,000 and has an unknown population trend. If we calculated a PBR for the Southeast Alaska/northern British Columbia feeding aggregation it would be 24 (2,252 × 0.035 × 0.3). If we calculated a PBR for the Aleutian Islands and Bering Sea, it would be 7.9 (2,256 × 0.035 × 0.1). If we calculated a PBR for the Gulf of Alaska, it would be 7.8 (2,222 × 0.035 × 0.1). However, note that the actual PBR for the Central North Pacific stock is 83 based on the breeding population size in Hawaii, as calculated above.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2014 and 2018 is listed, by marine mammal stock, in Young et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports. Injury events lacking detailed injury information are assigned prorated values following injury determination guidelines described in NMFS (2012). A summary of information used to determine whether an injury was serious or non-serious, as well as a table of prorate values used for large whale reports with incomplete information, is reported in Young et al. (2020). The minimum estimated mean annual level of human-caused mortality and serious injury for Central North Pacific humpback whales between 2014 and 2018 is 26 whales: 9.8 in U.S. commercial fisheries, 0.6 in recreational fisheries, 0.4 in subsistence fisheries, 7.9 in unknown (commercial, recreational, or subsistence) fisheries, 2.3 in marine debris, and 4.5 due to other causes (ship strikes and entanglement in an Alaska Department of Fish and Game (ADF&G) salmon net pen and in mooring gear); however, this estimate is considered a minimum because no observers have been assigned to several fisheries that are known to interact with this stock and, due to limited Canadian observer program data, mortality and serious injury incidental to Canadian commercial fisheries (i.e., those similar to U.S. fisheries known to interact with humpback whales) is uncertain. Assignment of mortality and serious injury to both the Central North Pacific and Western North Pacific stocks of humpback whales, when the stock is unknown and events occur within the area where the stocks are known to overlap, may result in overestimating stock specific mortality and serious injury. Potential threats most likely to result in direct human-caused mortality or serious injury of this stock include ship strikes and entanglement in fishing gear.

Fisheries Information

Information for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is available in Appendix 3 of the Alaska Stock Assessment Reports (observer coverage) and in the NMFS List of Fisheries (LOF) and the fact sheets linked to fishery names in the LOF (observer coverage and reported incidental takes of marine mammals: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-list-fisheries, accessed December 2020).

In 2018, one humpback whale mortality occurred in the Bering Sea/Aleutian Islands pollock trawl fishery, resulting in a minimum estimated mean annual mortality and serious injury rate of 0.2 humpback whales between 2014 and 2018 (Table 1; Breiwick 2013; MML, unpubl. data). Because the stock is unknown, and the event occurred within the area where the Central North Pacific and Western North Pacific stocks are known to overlap, the mortality in this fishery was assigned to both stocks of humpback whales. One Central North Pacific humpback whale was seriously injured in the Hawaii deep-set longline fishery in 2014, resulting in a mean annual mortality and serious injury rate of 0.9 whales in this fishery between 2014 and 2018 (Table 1; Bradford and Forney 2017; Bradford 2018a, 2018b; NMFS-PIFSC, unpubl. data).

In 2012 and 2013, the Alaska Marine Mammal Observer Program placed observers on independent vessels in the state-managed Southeast Alaska salmon drift gillnet fishery to assess mortality and serious injury of marine mammals. Areas around and adjacent to Wrangell and Zarembo Islands (ADF&G Districts 6, 7, and 8) were observed during the 2012 and 2013 programs (Manly 2015). In 2013, one humpback whale was seriously injured. Based on the one observed serious injury, 11 serious injuries were estimated for Districts 6, 7, and 8 in 2013, resulting in an estimated mean annual mortality and serious injury rate of 5.5 Central North Pacific humpback whales in 2012 and 2013 (Table 1). Because these three districts represent only a portion of the overall fishing effort in this fishery, we expect this to be a minimum estimate of mortality and serious injury for the fishery.

Mortality and serious injury reported to the NMFS Alaska Region marine mammal stranding network and through Marine Mammal Authorization Program (MMAP) fisherman self-reports, for fisheries in which observer data are not available, resulted in a minimum mean annual mortality and serious injury rate of 3.2 humpback whales in U.S. commercial fisheries in Alaska waters between 2014 and 2018 (Table 2; Young et al. 2020). Mortality and serious injury in events that occurred in the area where the two stocks overlap was assigned to both the Central North Pacific and Western North Pacific stocks of humpback whales (as noted in Table 2). These mortality and serious injury estimates result from an actual count of verified human-caused deaths and serious injuries and are minimums because not all entangled animals strand or are self-reported nor are all stranded animals found, reported, or have the cause of death determined.

The minimum estimate of the mean annual mortality and serious injury rate incidental to U.S. commercial fisheries for the Central North Pacific stock between 2014 and 2018 (or the most recent data available) is 9.8 humpback whales, based on observer data from Alaska (Table 1: 0.2 in the federally-managed Bering Sea/Aleutian Islands pollock trawl fishery and 5.5 in the state-managed Southeast Alaska salmon drift gillnet fishery), observer data from Hawaii (Table 1: 0.9), and MMAP fishermen self-reports and reports, in which the commercial fishery is confirmed, to the NMFS Alaska Region stranding network (Table 2: 3.2).

Table 1. Summary of incidental mortality and serious injury of Central North Pacific humpback whales due to U.S. commercial fisheries between 2014 and 2018 (or the most recent data available) and calculation of the mean annual mortality and serious injury rate for Alaska fisheries (Breiwick 2013; Manly 2015; MML, unpubl. data) and Hawaii fisheries (Bradford and Forney 2017; Bradford 2018a, 2018b; NMFS-PIFSC, unpubl. data). Methods for calculating percent observer coverage for Alaska fisheries are described in Appendix 3 of the Alaska Stock Assessment Reports.

Fishery name	**Years**	**Data type**	**Percent observer coverage**	**Observed mortality**	**Estimated mortality (CV)**	**Mean estimated annual mortality**
Bering Sea/Aleutian Is. pollock trawl*	2014 2015 2016 2017 2018	obs data	98 99 99 99 99	0 0 0 0 1	0 0 0 0 1.0 (0.11)	0.2 (CV = 0.11)
Southeast Alaska salmon drift gillnet (Districts 6, 7, 8)	2012 2013	obs data	6.4 6.6	0 1	0 11	5.5 (CV = 1.0)
Hawaii deep-set longline	2014 2015 2016 2017 2018	obs data	20 20 20 20 20	1 0 0 0 0	5 0 0 0 0	0.9 (CV = 2.1)
Minimum total estimated annual mortality					Bering Sea/Aleutian Islands	0.2
					Southeast Alaska:	5.5
					Hawaii:	0.9
					Total:	6.6 (CV = 0.88)

*Mortality and serious injury in this fishery was assigned to both the Central North Pacific and Western North Pacific stocks of humpback whales, because the stock is unknown and the two stocks overlap within the area of operation of the fishery.

Reports to the NMFS Alaska Region marine mammal stranding network of swimming, floating, or beachcast humpback whales entangled in fishing gear or with injuries caused by interactions with gear, which may be from commercial, recreational, or subsistence fisheries, are another source of information on fishery-related mortality and serious injury. Mortality and serious injury in events that occurred in the area where the two stocks overlap was assigned to both the Central North Pacific and Western North Pacific stocks (as noted in Table 2). These mortality and serious injury estimates result from an actual count of verified human-caused deaths and serious injuries and are minimums because not all entangled animals strand or are self-reported nor are all stranded animals found, reported, or have the cause of death determined. Between 2014 and 2018, three humpback whales (each with a serious injury prorated at 0.75) entangled in recreational pot fisheries gear, resulting in a minimum mean annual mortality and serious injury rate of 0.6 whales in recreational gear in Alaska waters (Table 2; Young et al. 2020). Humpback whales that entangled in Southeast Alaska subsistence halibut longline gear and in unidentified subsistence gillnet resulted in a minimum mean annual mortality and serious injury rate of 0.4 humpback whales in subsistence fisheries between 2014 and 2018 (Table 2; Young et al. 2020). Additional entanglements in unknown (commercial, recreational, or subsistence) fishing gear between 2014 and 2018 resulted in a minimum mean annual mortality and serious injury rate of 7.9 humpback whales: 1.5 reported to the NMFS Alaska Region stranding network (Table 2; Young et al. 2020) and 6.4 reported to the NMFS Pacific Islands Region stranding network (Table 3; Bradford and Lyman 2018, 2019, 2020).

The minimum mean annual mortality and serious injury rate due to interactions with all fisheries between 2014 and 2018 is 19 Central North Pacific humpback whales (9.8 in commercial fisheries + 0.6 in recreational fisheries + 0.4 in subsistence fisheries + 7.9 in unknown fisheries).

Table 2. Summary of mortality and serious injury of Central North Pacific humpback whales, by year and type, reported to the NMFS Alaska Region marine mammal stranding network and by Marine Mammal Authorization Program (MMAP) fisherman self-reports between 2014 and 2018 (Young et al. 2020). Injury events lacking detailed injury information are assigned prorated values following injury determination guidelines described in NMFS (2012). A summary of information used to determine whether an injury was serious or non-serious, as well as a table of prorate values used for large whale reports with incomplete information, is reported in Young et al. (2020).

Cause of injury	**2014**	**2015**	**2016**	**2017**	**2018**	**Mean annual mortality**
Entangled in Southeast Alaska commercial salmon purse seine gear	0	0.75	0	0	0	0.2
Entangled in Kodiak Island commercial salmon set gillnet	0	0.75[a]	0	0	0	0.2
Entangled in Prince William Sound commercial salmon drift gillnet	0	1.5	0	0	0	0.3
Entangled in Southeast Alaska commercial salmon drift gillnet (in ADF&G Districts that were not observed in 2012 and 2013)	2.5 + 0.75[b]	0.75	2.25	0	1.5	1.6
Entangled in Bering Sea/Aleutian Is. commercial pot gear	0	1[a]	0	0	0	0.2
Entangled in Southeast Alaska commercial pot gear	0	0.75	0	0	0	0.2
Dependent calf of animal seriously injured in Southeast Alaska commercial pot gear	0	0.75	0	0	0	0.2
Entangled in Alaska State-managed commercial cod pot gear (parallel fishery)	0	0	0	1[a]	0	0.2
Ship strike by AK/WA/OR/CA commercial passenger fishing vessel	0	0	0	0.2 + 0.52[a]	0	0.1
Entangled in Gulf of Alaska recreational Dungeness crab pot gear	0	0.75[a]	0	0	0	0.2

Cause of injury	2014	2015	2016	2017	2018	Mean annual mortality
Entangled in Gulf of Alaska recreational shrimp pot gear	0	0.75[a]	0	0	0	0.2
Entangled in Southeast Alaska recreational shrimp pot gear	0	0	0	0	0.75	0.2
Entangled in Southeast Alaska subsistence halibut longline gear	0	0.75	0	0	0	0.2
Entangled in unidentified subsistence gillnet	0	0	0.75	0	0	0.2
Entangled in Prince William Sound shrimp pot gear*	1[a]	0	0	0	0	0.2
Entangled in Southeast Alaska unidentified fishing gear*	0	1.5	0	0	0	0.3
Dependent calf of animal seriously injured in Southeast Alaska unidentified fishing gear*	0	0.75	0	0	0	0.2
Entangled in Southeast Alaska unidentified net*	0	1.5	0	0	0	0.3
Entangled in unidentified fishing gear*	0	0	0	1	0.75 + 0.75[a]	0.5
Entangled in marine debris	3.75 + 0.75[a]	1.75	2.25 + 2[a]	0.75	0	2.3
Entangled in ADF&G salmon net pen	0	0	0.75	0	0	0.2
Entangled in mooring gear	0	0	0.75	0	0	0.2
Ship strike	3.52 + 1.2[a]	2.8	1 + 0.2[a]	1.34	3	2.6
Total in commercial fisheries						3.2
Total in recreational fisheries						0.6
Total in subsistence fisheries						0.4
*Total in unknown (commercial, recreational, or subsistence) fisheries						1.5
Total in marine debris						2.3
Total due to other causes (entangled in salmon net pen, entangled in mooring gear, ship strike)						3

[a]Mortality and serious injury assigned to both the Central North Pacific (CNP) and Western North Pacific (WNP) stocks.
[b]MMAP fisherman self-report.

Table 3. Summary of mortality and serious injury of Central North Pacific humpback whales reported to the NMFS Pacific Islands Region stranding network between 2014 and 2018 (Bradford and Lyman 2018, 2019, 2020).

Cause of injury	2014	2015	2016	2017	2018	Mean annual mortality
Entangled in Alaska shrimp pot gear*	1	0	0	0	0	0.2
Entangled in Alaska king crab, tanner crab, or finfish pot gear*	0.75	0	0	0	0	0.2
Entangled in British Columbia pot gear*	0	0	0	0	2	0.4
Entangled in longline gear*	1	0	0	0	0	0.2
Entangled in unidentified gillnet*	0	0	0	0	1	0.2
Entangled in unidentified fishing gear*	6.5	7.75	2.5	5.25	4	5.2
Ship strike	1	1.2	0.2	1.2	4	1.5
*Total in unknown (commercial, recreational, or subsistence) fisheries						6.4
Total due to other causes (ship strike)						1.5

However, these estimates of mortality and serious injury levels should be considered minimums. No observers have been assigned to several fisheries that are known to interact with this stock, making the estimated mortality and serious injury rate an underestimate of actual mortality and serious injury. Further, due to limited Canadian observer program data, mortality and serious injury incidental to Canadian commercial fisheries (i.e., those similar to U.S. fisheries known to interact with humpback whales) is uncertain. Though interactions are thought to be minimal, data regarding the level of humpback whale mortality and serious injury related to commercial fisheries in northern British Columbia are not available, again indicating that the estimated mortality and serious injury incidental to commercial fisheries is underestimated for this stock.

Alaska Native Subsistence/Harvest Information

Subsistence hunters in Alaska are not authorized to take humpback whales from this stock, and no takes were reported between 2014 and 2018.

Other Mortality

In 2015, increased mortality of large whales was observed along the western Gulf of Alaska (including the areas around Kodiak Island, Afognak Island, Chirikof Island, the Semidi Islands, and the southern shoreline of the Alaska Peninsula) and along the central British Columbia coast (from the northern tip of Haida Gwaii to southern Vancouver Island). NMFS declared an Unusual Mortality Event (UME) for large whales that occurred from 22 May to 31 December 2015 in the western Gulf of Alaska and from 23 April 2015 to 16 April 2016 in British Columbia (https://www.fisheries.noaa.gov/national/marine-life-distress/active-and-closed-unusual-mortality-events, accessed December 2020). Forty-six large whale deaths attributed to the UME included 12 fin whales and 22 humpback whales in Alaska and 5 fin whales and 7 humpback whales in British Columbia. Based on the findings from the investigation, the UME was likely caused by ecological factors (i.e., the 2015 El Niño, Warm Water Blob, and Pacific Coast Domoic Acid Bloom).

Entanglements in marine debris, an ADF&G salmon net pen, and mooring gear reported to the NMFS Alaska Region marine mammal stranding network resulted in minimum mean annual mortality and serious injury rates of 2.3, 0.2, and 0.2 Central North Pacific humpback whales, respectively, between 2014 and 2018 (Table 2; Young et al. 2020). Ship strikes and other interactions with vessels unrelated to fisheries occur frequently with humpback whales (Tables 2 and 3). The minimum mean annual mortality and serious injury rate due to ship strikes in Alaska (Table 2: 2.6) and ship strikes reported in Hawaii (Table 3: 1.5) between 2014 and 2018 is 4.1 humpback whales. These mortality and serious injury estimates result from an actual count of verified human-caused deaths and serious injuries and are minimums because not all animals strand nor are all stranded animals found, reported, or have the cause of death determined. Neilson et al. (2012) summarized 108 large whale ship-strike events in Alaska from 1978 to 2011, 25 of which are known to have resulted in the whale's death. Eighty-six percent of these reports involved humpback whales. Most ship strikes of humpback whales are reported from Southeast Alaska; however, there are also reports from the southcentral, Kodiak Island, and Prince William Sound areas of Alaska (Young et al. 2020). Many of the ship strikes occurring off Hawaii are reported from waters near Maui (Bradford and Lyman 2018, 2019). It is not known whether the difference in ship-strike rates between Southeast Alaska and the northern portion of this stock is due to differences in reporting, amount of vessel traffic, densities of animals, or other factors.

HISTORICAL WHALING

Rice (1978) estimated that the number of humpback whales in the North Pacific may have been approximately 15,000 individuals prior to exploitation; however, this was based upon incomplete data and, given the level of known catches (legal and illegal) since World War II, may be an underestimate. Intensive commercial whaling removed more than 28,000 animals from the North Pacific during the 20th century. Humpback whales in the North Pacific were theoretically fully protected in 1965, but illegal catches by the U.S.S.R. continued until 1972 (Ivashchenko et al. 2013). From 1948 to 1971, 7,334 humpback whales were killed by the U.S.S.R., and 2,654 of these were illegally taken and not reported to the IWC (Ivashchenko et al. 2013). Many animals during this period were taken from the Gulf of Alaska and Bering Sea (Doroshenko 2000); additional illegal catches were made across the North Pacific, from the Kuril Islands to Haida Gwaii, and other takes may have gone unrecorded. The Soviet factory ship *Aleut* is known to have taken 535 humpback whales from 1933 to 1947 (Ivashchenko et al. 2013).

On the feeding grounds of the Central North Pacific stock after World War II, the highest densities of catches occurred around the western Aleutian Islands, in the eastern Aleutian Islands (and adjacent Bering Sea to the north and Pacific Ocean to the south), and British Columbia (Springer et al. 2006). Lower but still relatively high densities of catches occurred south of the Commander Islands, along the south side of the Alaska Peninsula, and around Kodiak Island. Lower densities of catches also occurred in the Gulf of Anadyr, in the central Aleutian

Islands, in much of the offshore Gulf of Alaska, and in Southeast Alaska. No catches were reported in the winter grounds of the Central North Pacific stock in Hawaii nor in Mexican winter areas.

STATUS OF STOCK

NMFS recently concluded a global humpback whale Status Review (Bettridge et al. 2015). Although the estimated mean annual level of human-caused mortality and serious injury for the entire Central North Pacific stock (26 whales) is considered a minimum, it is unlikely that the total mean annual level of human-caused mortality and serious injury exceeds the PBR level (83) for the entire stock. The minimum estimate of the mean annual U.S. commercial fishery-related mortality and serious injury rate for this stock (9.8 whales) is more than 10% of the calculated PBR for the entire stock (10% of PBR = 8.3) and, therefore, cannot be considered insignificant and approaching a zero mortality and serious injury rate. The humpback whale ESA listing final rule (81 FR 62259, 8 September 2016) established 14 Distinct Population Segments (DPSs) with different listing statuses. The DPSs that occur in waters under the jurisdiction of the United States do not equate to the existing MMPA stocks. Some of the listed DPSs partially coincide with the currently defined Central North Pacific stock. Because we cannot manage one portion of an MMPA stock as ESA-listed and another portion of a stock as not ESA-listed, until such time as the MMPA stock delineations are reviewed in light of the DPS designations and Bettridge et al. (2015), NMFS continues to use the existing MMPA stock structure and considers this stock to be endangered and depleted for MMPA management purposes (e.g., selection of a recovery factor, stock status). As a result, the Central North Pacific stock of humpback whales is classified as a strategic stock. Humpback whale mortality and serious injury in Hawaii-based fisheries involves whales from the Hawaii DPS; this DPS is not listed as threatened or endangered under the ESA.

There are key uncertainties in the assessment of the Central North Pacific stock of humpback whales. New DPSs were identified under the ESA; however, stocks have not yet been revised. No estimate of variance is available for the abundance estimate. The feeding areas of the Central North Pacific stock and the Western North Pacific stock overlap in waters from British Columbia to the Bering Sea, so human-related mortality and serious injury estimates must be assigned to or prorated to multiple stocks. The current abundance estimate is calculated using data collected from 2004 to 2006; however, the N_{MIN} is still considered a valid minimum population estimate because the population is increasing (NMFS 2016). There is considerable site fidelity of humpback whales to particular feeding areas; human-related mortality and serious injury could have a disproportionate impact on a local feeding population even if the impacts to the DPS as currently described are low relative to the PBR level. Estimates of human-caused mortality and serious injury from stranding data and fisherman self-reports are underestimates because not all animals strand or are self-reported nor are all stranded animals found, reported, or have the cause of death determined.

HABITAT CONCERNS

This stock is the focus of a large whale-watching industry in its wintering grounds (Hawaii) and summering grounds (Alaska). Regulations concerning the minimum distance to keep from whales and how to operate vessels when in the vicinity of whales have been developed for Hawaii and Alaska waters in an attempt to minimize the effect of whale watching. In land-based studies in both Hawaii and Southeast Alaska, the presence of vessels was shown to induce energetically demanding avoidance behaviors in humpback whales. These include changes such as increases in swim speed and changes in swimming direction as well as several other changes in respiration metrics such as decreases in dive times, increased respiration rate, and decreased inter-breath intervals (Schuler et al. 2019, Currie et al. 2021). Additional concerns have been raised in Hawaii about the effect of jet skis and similar fast waterborne tourist-related traffic, notably in nearshore areas inhabited by mothers and calves. In Alaska, NMFS issued regulations in 2001 to prohibit approaches to humpback whales within 100 yards (91.4 m: 66 FR 29502, 31 May 2001). In 2015, NMFS introduced a voluntary responsible viewing program called Whale SENSE to Juneau area whale-watch operators to provide additional protections for whales in Alaska (https://whalesense.org, accessed December 2020). The growth of the whale-watching industry is an ongoing concern as preferred habitats may be abandoned if disturbance levels are too high.

Other potential concerns for this stock include elevated levels of sound from anthropogenic sources (e.g., shipping, military sonars), harmful algal blooms (Geraci et al. 1989), possible changes in prey distribution with climate change, entanglement in fishing gear, ship strikes due to increased vessel traffic (e.g., from increased shipping in higher latitudes), oil and gas activities, and an overlap between humpback whales and high concentrations of marine debris. In a study that quantified the amount and type of marine debris accumulation in Hawaii coastal waters from 2013 to 2016, the degree of overlap between marine debris and cetacean distribution was greatest for humpback whales (Currie et al. 2017).

The overall trend for most humpback whale populations found in U.S. waters is positive and points toward recovery (81 FR 62259, 8 September 2016); however, this may not be uniform for all breeding areas. A sharp decline in observed reproduction and encounter rates of humpback whales from the Central North Pacific stock between 2013 and 2018 has been related to oceanographic anomalies and consequent impacts on prey resources (Cartwright et al. 2019), suggesting that humpback whales are vulnerable to major environmental changes.

CITATIONS

Baker, C. S., L. M. Herman, A. Perry, W. S. Lawton, J. M. Straley, A. A. Wolman, G. D. Kaufman, H. E. Winn, J. D. Hall, J. M. Reinke, and J. Ostman. 1986. Migratory movement and population structure of humpback whales (*Megaptera novaeangliae*) in the central and eastern North Pacific. Mar. Ecol. Prog. Ser. 31:105-119.

Baker, C. S., A. Perry, and L. M. Herman. 1987. Reproductive histories of female humpback whales (*Megaptera novaeangliae*) in the North Pacific. Mar. Ecol. Prog. Ser. 41:103-114.

Baker, C. S., S. R. Palumbi, R. H. Lambertsen, M. T. Weinrich, J. Calambokidis, and S. J. O'Brien. 1990. Influence of seasonal migration on geographic distribution of mitochondrial DNA haplotypes in humpback whales. Nature 344:238-240.

Baker, C. S., L. Medrano-Gonzalez, J. Calambokidis, A. Perry, F. Pichler, H. Rosenbaum, J. M. Straley, J. Urban-Ramirez, M. Yamaguchi, and O. von Ziegesar. 1998. Population structure of nuclear and mitochondrial DNA variation among humpback whales in the North Pacific. Mol. Ecol. 7:695-707.

Baker, C. S., D. Steel, J. Calambokidis, J. Barlow, A. M. Burdin, P. J. Clapham, E. A. Falcone, J. K. B. Ford, C. M. Gabriele, U. González-Peral, R. LeDuc, D. Mattila, T. J. Quinn, L. Rojas-Bracho, J. M. Straley, B. L. Taylor, J. Urbán-R., M. Vant, P. Wade, D. Weller, B. H. Witteveen, K. Wynne, and M. Yamaguchi. 2008. *geneSPLASH*: an initial, ocean-wide survey of mitochondrial (mt) DNA diversity and population structure among humpback whales in the North Pacific. Final Report for Contract 2006-0093-008 to the National Fish and Wildlife Foundation.

Baker, C. S., D. Steel, J. Calambokidis, E. Falcone, U. González-Peral, J. Barlow, A. M. Burdin, P. J. Clapham, J. K. B. Ford, C. M. Gabriele, D. Mattila, L. Rojas-Bracho, J. M. Straley, B. L. Taylor, J. Urbán, P. R. Wade, D. Weller, B. H. Witteveen, and M. Yamaguchi. 2013. Strong maternal fidelity and natal philopatry shape genetic structure in North Pacific humpback whales. Mar. Ecol. Prog. Ser. 494:291-306. DOI: dx.doi.org/10.3354/meps10508 .

Barlow, J., and P. J. Clapham. 1997. A new birth-interval approach to estimating demographic parameters of humpback whales. Ecology 78(2):535-546.

Barlow, J., J. Calambokidis, E. A. Falcone, C. S. Baker, A. M. Burdin, P. J. Clapham, J. K. B. Ford, C. M. Gabriele, R. LeDuc, D. K. Mattila, T. J. Quinn, L. Rojas-Bracho, J. M. Straley, B. L. Taylor, J. Urbán R., P. Wade, D. Weller, B. H. Witteveen, and M. Yamaguchi. 2011. Humpback whale abundance in the North Pacific estimated by photographic capture-recapture with bias correction from simulation studies. Mar. Mammal Sci. 27:793-818.

Bettridge, S., C. S. Baker, J. Barlow, P. J. Clapham, M. Ford, D. Gouveia, D. K. Mattila, R. M. Pace III, P. E. Rosel, G. K. Silber, and P. R. Wade. 2015. Status review of the humpback whale (*Megaptera novaeangliae*) under the Endangered Species Act. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-SWFSC-540, 240 p.

Bradford, A. L. 2018a. Injury determinations for marine mammals observed interacting with Hawaii and American Samoa longline fisheries during 2015-2016. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-PIFSC-70, 27 p. DOI: dx.doi.org/10.7289/V5/TM-PIFSC-70 .

Bradford, A. L. 2018b. Injury determinations for marine mammals observed interacting with Hawaii and American Samoa longline fisheries during 2017. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-PIFSC-76, 14 p. DOI: dx.doi.org/10.25923/fzad-4784 .

Bradford, A. L., and K. A. Forney. 2017. Injury determinations for marine mammals observed interacting with Hawaii and American Samoa longline fisheries during 2010-2014. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-PIFSC-62, 28 p. DOI: dx.doi.org/10.7289/V5/TM-PIFSC-62 .

Bradford, A. L., and E. Lyman. 2018. Injury determinations for humpback whales and other cetaceans reported to NOAA Response Networks in the Hawaiian Islands during 2013-2016. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-PIFSC-75, 24 p.

Bradford, A. L., and E. G. Lyman. 2019. Injury determinations for humpback whales and other cetaceans reported to NOAA response networks in the Hawaiian Islands during 2017. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-PIFSC-81, 18 p. DOI: dx.doi.org/10.25923/7csm-h961 .

Bradford, A. L., and E. G. Lyman. 2020. Injury determinations for humpback whales and other cetaceans reported to NOAA response networks in the Hawaiian Islands during 2018. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-PIFSC-103, 18 p. DOI: dx.doi.org/10.25923/mtcd-f441 .

Breiwick, J. M. 2013. North Pacific marine mammal bycatch estimation methodology and results, 2007-2011. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-260, 40 p.

Calambokidis, J., G. H. Steiger, J. C. Cubbage, K. C. Balcomb III, and P. Bloedel. 1989. Biology of humpback whales in the Gulf of the Farallones. Report to Gulf of the Farallones National Marine Sanctuary, San Francisco, CA, by Cascadia Research Collective, 218½ West Fourth Avenue, Olympia, WA. 93 p.

Calambokidis, J., G. H. Steiger, and J. R. Evenson. 1993. Photographic identification and abundance estimates of humpback and blue whales off California in 1991-92. Final Contract Report 50ABNF100137 to Southwest Fisheries Science Center, La Jolla, CA 92037. 67 p.

Calambokidis, J., G. H. Steiger, J. M. Straley, T. Quinn, L. M. Herman, S. Cerchio, D. R. Salden, M. Yamaguchi, F. Sato, J. Urban R., J. Jacobson, O. von Ziegesar, K. C. Balcomb, C. M. Gabriele, M. E. Dahlheim, N. Higashi, S. Uchida, J. K. B. Ford, Y. Miyamura, P. Ladrón de Guevara, S. A. Mizroch, L. Schlender, and K. Rasmussen. 1997. Abundance and population structure of humpback whales in the North Pacific basin. Final Contract Report 50ABNF500113 to Southwest Fisheries Science Center, 8901 La Jolla Shores Drive, La Jolla, CA 92037. 72 p.

Calambokidis, J., G. H. Steiger, J. M. Straley, L. M. Herman, S. Cerchio, D. R. Salden, J. Urban R., J. K. Jacobsen, O. von Ziegesar, K. C. Balcomb, C. M. Gabriele, M. E. Dahlheim, S. Uchida, G. Ellis, Y. Miyamura, P. Ladrón de Guevara P., M. Yamaguchi, F. Sato, S. A. Mizroch, L. Schlender, K. Rasmussen, J. Barlow, and T. J. Quinn II. 2001. Movements and population structure of humpback whales in the North Pacific. Mar. Mammal Sci. 17(4):769-794.

Calambokidis, J., E. A. Falcone, T. J. Quinn, A. M. Burdin, P. J. Clapham, J. K. B. Ford, C. M. Gabriele, R. LeDuc, D. Mattila, L. Rojas-Bracho, J. M. Straley, B. L. Taylor, J. Urbán R., D. Weller, B. H. Witteveen, M. Yamaguchi, A. Bendlin, D. Camacho, K. Flynn, A. Havron, J. Huggins, and N. Maloney. 2008. SPLASH: Structure of Populations, Levels of Abundance and Status of Humpback Whales in the North Pacific. Final Report for Contract AB133F-03-RP-00078. 58 p. Available online: http://www.cascadiaresearch.org/files/publications/SPLASH-contract-Report-May08.pdf . Accessed December 2020.

Cartwright, R., A. Venema, V. Hernandez, C. Wyels, J. Cesere, and D. Cesere. 2019. Fluctuating reproductive rates in Hawaii's humpback whales, *Megaptera novaeangliae*, reflect recent climate anomalies in the North Pacific. Roy. Soc. Open Sci. 6:181463. DOI: dx.doi.org/10.1098/rsos.181463 .

Clapham, P. J., and J. G. Mead. 1999. *Megaptera novaeangliae*. Mamm. Species 604:1-9.

Clapham, P. J., J. Barlow, M. Bessinger, T. Cole, D. Mattila, R. Pace, D. Palka, J. Robbins, and R. Seton. 2003. Abundance and demographic parameters of humpback whales from the Gulf of Maine, and stock definition relative to the Scotian Shelf. J. Cetacean Res. Manage. 5:13-22.

Clarke, J., K. Stafford, S. E. Moore, B. Rone, L. Aerts, and J. Crance. 2013. Subarctic cetaceans in the southern Chukchi Sea: evidence of recovery or response to a changing ecosystem. Oceanography 26(4):136-149.

Currie, J. J., S. H. Stack, J. A. McCordic, and G. D. Kaufman. 2017. Quantifying the risk that marine debris poses to cetaceans in coastal waters of the 4-island region of Maui. Marine Pollution Bulletin 121:69-77. DOI: dx.doi.org/10.1016/j.marpolbul.2017.05.031 .

Currie, J. J., J. A. McCordic, G. L. Olson, A. F. Machernis, and S. H. Stack. 2021. The impact of vessels on humpback whale behavior: the benefit of added whale watching guidelines. Frontiers in Marine Science. DOI: dx.doi.org/10.3389/fmars.2021.601433 .

Darling, J. D. 1991. Humpback whales in Japanese waters. Ogasawara and Okinawa. Fluke identification catalog 1987-1990. Final Contract Report, World Wide Fund for Nature, Japan. 22 p.

Darling, J. D., and S. Cerchio. 1993. Movement of a humpback whale (*Megaptera novaeangliae*) between Japan and Hawaii. Mar. Mammal Sci. 1:84-89.

Doroshenko, N. V. 2000. Soviet catches of humpback whales (*Megaptera novaeangliae*) in the North Pacific, p. 48-95. *In* A. V. Yablokov and V. A. Zemsky (eds.), Soviet Whaling Data (1949-1979), Center for Russian Environmental Policy, Marine Mammal Council, Moscow.

Fleming, A., and J. Jackson. 2011. Global review of humpback whales (*Megaptera novaeangliae*). U.S. Dep. Commer., NOAA Tech. Memo. NMFS-SWFSC-474, 206 p.

Ford J. K. B., A. L. Rambeau, R. M. Abernethy, M. D. Boogaards, L. M. Nichol, and L. D. Spaven. 2009. An assessment of the potential for recovery of humpback whales off the Pacific coast of Canada. DFO Can. Sci. Advis. Sec. Res. Doc. 2009/015. iv + 33 p.

Geraci, J. R., D. M. Anderson, R. J. Timperi, D. St. Aubin, G. A. Early, J. H. Prescott, and C. A. Mayo. 1989. Humpback whales (*Megaptera novaeangliae*) fatally poisoned by dinoflagellate toxin. Can. J. Fish. Aquat. Sci. 46(11):1895-1898. DOI: dx.doi.org/10.1139/f89-238 .

Hashagen, K. A., G. A. Green, and B. Adams. 2009. Observations of humpback whales, *Megaptera novaeangliae*, in the Beaufort Sea, Alaska. Northwest. Nat. 90:160-162.

Hill, M. C., E. M. Oleson, S. Baumann-Pickering, A. M. Van Cise, A. D. Ligon, A. R. Bendlin, A. C. Ü, J. S. Trickey, and A. L. Bradford. 2016. Cetacean monitoring in the Mariana Islands Range Complex, 2015. Prepared for the U.S. Pacific Fleet Environmental Readiness Office. PIFSC Data Report DR-16-01. 36 p. + appendix.

Ivashchenko, Y. V., R. L. Brownell, Jr., and P. J. Clapham. 2013. Soviet whaling in the North Pacific: revised catch totals. J. Cetacean Res. Manage. 13:59-71.

Johnson, J. H., and A. A. Wolman. 1984. The humpback whale, *Megaptera novaeangliae*. Mar. Fish. Rev. 46:30-37.

Johnston, D. W., M. E. Chapla, L. E. Williams, and D. K. Mattila. 2007. Identification of humpback whale *Megaptera novaeangliae* wintering habitat in the Northwestern Hawaiian Islands using spatial habitat modeling. Endang. Species Res. 3:249-257.

Manly, B. F. J. 2015. Incidental takes and interactions of marine mammals and birds in districts 6, 7 and 8 of the Southeast Alaska salmon drift gillnet fishery, 2012 and 2013. Final Report to NMFS Alaska Region. 52 p.

Mizroch, S. A., L. M. Herman, J. M. Straley, D. Glockner-Ferrari, C. Jurasz, J. Darling, S. Cerchio, C. Gabriele, D. Salden, and O. von Ziegesar. 2004. Estimating the adult survival rate of central North Pacific humpback whales. J. Mammal. 85(5):963-972.

Mobley, J. M., S. Spitz, R. Grotefendt, P. Forestell, A. Frankel, and G. Bauer. 2001. Abundance of humpback whales in Hawaiian waters: results of 1993-2000 aerial surveys. Report to the Hawaiian Islands Humpback Whale National Marine Sanctuary. 16 p.

Moore, S. E., J. M. Waite, N. A. Friday, and T. Honkalehto. 2002. Distribution and comparative estimates of cetacean abundance on the central and south-eastern Bering Sea shelf with observations on bathymetric and prey associations. Prog. Oceanogr. 55(1-2):249-262.

National Marine Fisheries Service (NMFS). 2012. Process for distinguishing serious from non-serious injury of marine mammals. 42 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-policies-guidance-and-regulations . Accessed December 2020.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2020.

Neilson, J. L., C. M. Gabriele, A. S. Jensen, K. Jackson, and J. M. Straley. 2012. Summary of reported whale-vessel collisions in Alaskan waters. J. Mar. Biol. 2012: Article ID 106282. 18 p. DOI: dx.doi.org/10.1155/2012/106282 .

Nemoto, T. 1957. Foods of baleen whales in the northern Pacific. Sci. Rep. Whales Res. Inst. Tokyo 12:33-89.

Nemoto, T. 1959. Food of baleen whales with reference to whale movements. Sci. Rep. Whales Res. Inst. 14:149-290.

Perry, A., C. S. Baker, and L. M. Herman. 1990. Population characteristics of individually identified humpback whales in the central and eastern North Pacific: a summary and critique. Rep. Int. Whal. Comm. (Special Issue 12):307-317.

Rice, D. W. 1978. The humpback whale in the North Pacific: distribution, exploitation and numbers, Appendix 4, p. 29-44. *In* K. S. Norris and R. R. Reeves (eds.), Report on a workshop on problems related to humpback whales (*Megaptera novaeangliae*) in Hawaii. U.S. Dep. Commer., Natl. Tech. Info. Serv. PB-280794. Springfield, VA.

Rice, D. W. 1998. Marine Mammals of the World: Systematics and Distribution. Soc. Mar. Mammal. Spec. Publ. No. 4.

Schuler, A. R., S. Piwetz, J. Di Clemente, D. Steckler, F. Mueter, and H. C. Pearson. 2019. Humpback whale movements and behavior in response to whale-watching vessels in Juneau, AK. Frontiers in Marine Science 6: Article 710. DOI: dx.doi.org/10.3389/fmars.2019.00710 .

Springer, A. M., G. B. van Vliet, J. F. Piatt, and E. M. Danner. 2006. Whales and whaling in the North Pacific Ocean and Bering Sea: oceanographic insights and ecosystem impacts, p. 245-261. *In* J. A. Estes, R. L. Brownell, Jr., D. P. DeMaster, D. F. Doak, and T. M. Williams (eds.), Whales, Whaling and Ocean Ecosystems. University of California Press. 418 p.

Steiger, G. H., J. Calambokidis, R. Sears, K. C. Balcomb, and J. C. Cubbage. 1991. Movement of humpback whales between California and Costa Rica. Mar. Mammal Sci. 7:306-310.

Straley, J. M., C. M. Gabriele, and T. J. Quinn II. 2009. Assessment of mark recapture models to estimate the abundance of a humpback whale feeding aggregation in Southeast Alaska. J. Biogeogr. 36:427-438.

Taylor, B. L., M. Scott, J. Heyning, and J. Barlow. 2003. Suggested guidelines for recovery factors for endangered marine mammals. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-SWFSC-354, 6 p.

Teerlink, S. F., O. von Ziegesar, J. M. Straley, T. J. Quinn II, C. O. Matkin, and E. L. Saulitis. 2015. First time series of estimated humpback whale (*Megaptera novaeangliae*) abundance in Prince William Sound. Environ. Ecol. Stat. 22:345. DOI: dx.doi.org/10.1007/s10651-014-0301-8 .

Tomlin, A. G. 1967. Mammals of the USSR and Adjacent Countries. Vol. 9. Cetacea. Israel Program for Scientific Translations No. 1124, Natl. Tech. Info. Serv. TT 65-50086. Springfield, VA. 717 p. (Translation of Russian text published in 1957.)

Waite, J. M., M. E. Dahlheim, R. C. Hobbs, S. A. Mizroch, O. von Ziegesar-Matkin, J. M. Straley, L. M. Herman, and J. Jacobsen. 1999. Evidence of a feeding aggregation of humpback whales (*Megaptera novaeangliae*) around Kodiak Island, Alaska. Mar. Mammal Sci. 15:210-220.

Witteveen, B. H., J. M. Straley, O. von Ziegesar, D. Steel, and C. S. Baker. 2004. Abundance and mtDNA differentiation of humpback whales (*Megaptera novaeangliae*) in the Shumagin Islands, Alaska. Can. J. Zool. 82:1352-1359.

Young, N. C., B. J. Delean, V. T. Helker, J. C. Freed, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2014-2018. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-413, 142 p.

Zenkovich, B. A. 1954. Vokrug sveta za kitami, Vol. Gosudarstvennoe Izdatel'stvo Geograficheskoi Literatury, Moscow.

Zerbini, A. N., J. M. Waite, J. L. Laake, and P. R. Wade. 2006. Abundance, trends and distribution of baleen whales off western Alaska and the central Aleutian Islands. Deep-Sea Res. I 53(11):1772-1790.

FIN WHALE (*Balaenoptera physalus*): Northeast Pacific Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Within the U.S. waters in the Pacific Ocean, fin whales are found seasonally off the coast of North America and in the Bering Sea during the summer (Fig. 1). Information on seasonal fin whale distribution has been gleaned from the detection of fin whale calls using bottom-mounted, offshore hydrophone arrays along the U.S. Pacific coast, in the central North Pacific, and in the western Aleutian Islands (Moore et al. 1998, 2006; Watkins et al. 2000; Stafford et al. 2007; Širović et al. 2013; Soule and Wilcock 2013; Archer et al. 2019). Moore et al. (1998, 2006), Watkins et al. (2000), and Stafford et al. (2007) documented fin whale calling along the U.S. Pacific coast where rates were highest from August/September through February, suggesting that these may be important feeding areas during the winter. Širović et al. (2013) speculated that both resident and migratory fin whales may occur off southern California based on shifts in peaks in fin whale calling data. Širović et al. (2015) noted that fin whales were detected in the Southern California Bight year-round and found an overall increase in the fin whale call index from 2006 to 2012. Soule and Wilcock (2013) documented fin whale call rates in a presumed feeding area along the Juan de Fuca Ridge, offshore of northern Washington State, and found that some whales appear to transit northwest from August to October. They speculate that some fin whales migrate northward from the Juan de Fuca Ridge in fall and southward in winter. While peaks in call rates occurred during late summer, fall, and winter in the central North Pacific and the Aleutian Islands, fin whale calls were seldom detected during summer months even though fin whales are regularly seen in summer months in the Gulf of Alaska (Stafford et al. 2007). Fin whale calls have been detected in the southeast Bering Sea by a moored hydrophone. During April 2006 through April 2007, peaks in fin whale call detections were found from September through November 2006 and also in February and March 2007 (Stafford et al. 2010). In addition, fin whale calls were detected in the northeastern Chukchi Sea using instruments moored there from July through October between 2007 and 2010 (Delarue et al. 2013). Call data collected from the Bering Sea suggest that several putative fin whale stocks may feed in the Bering Sea; however, only one of these likely migrates into the Chukchi Sea to feed (Delarue et al. 2013). Some fin whale calls have also been recorded in the Hawaiian portion of the U.S. Exclusive Economic Zone in all months except June and July (Thompson and Friedl 1982, McDonald and Fox 1999). Sightings of fin whales in Hawaii are extremely rare: there was a sighting in 1976 (Shallenberger 1981), a sighting in 1979 (Mizroch et al. 2009), a sighting during an aerial survey in 1994 (Mobley et al. 1996), and five sightings during a survey in 2002 (Barlow 2006).

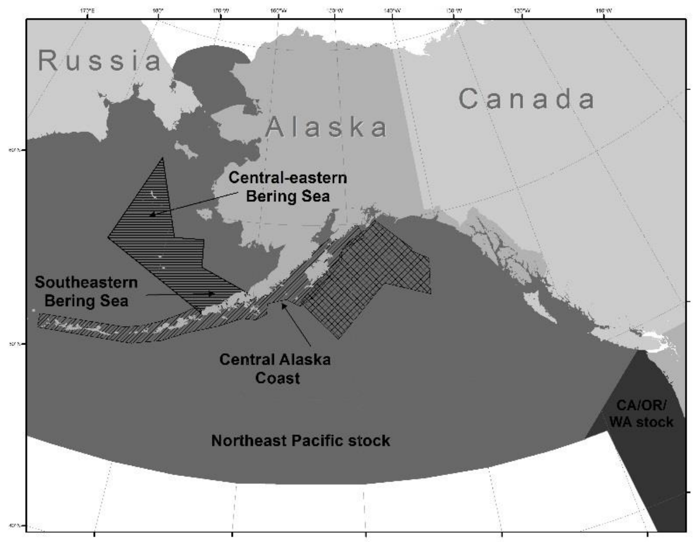

Figure 1. Approximate distribution of fin whales in the eastern North Pacific. Striped areas indicate where vessel surveys occurred in 1999-2010 (horizontal stripes - Bering Sea: Moore et al. 2002; Friday et al. 2012, 2013); 2001-2003 (diagonal stripes - Central Alaska coast and Aleutian Islands: Zerbini et al. 2006); and 2009, 2013, and 2015 (crosshatch - Gulf of Alaska: Rone et al. 2017).

Surveys on the Bering Sea shelf in 1997, 1999, 2000, 2002, 2004, 2008, and 2010 and in coastal waters of the Aleutian Islands and the Alaska Peninsula from 2001 to 2003 provided information about the distribution and abundance of fin whales in these areas (Moore et al. 2000, 2002; Zerbini et al. 2006; Friday et al. 2012, 2013). Fin whales were the most common large whale sighted during the Bering Sea shelf surveys in all years except for 1997 and 2004 (Friday et al. 2012, 2013). Fin whales were consistently distributed both in the "green belt," an area of high productivity along the edge of the eastern Bering Sea continental shelf (Springer et al. 1996), and, at a lower frequency, in the middle shelf. Abundance estimates for fin whales in the Bering Sea were consistently higher in cold years than in warm years (Friday et al. 2012, 2013) indicating a shift in distribution. This is consistent with a fine-scale comparison of fin whale occurrence on the middle shelf between a cold year (1999) and a warm year (2002), which found that the group and individual encounter rates were 7 to 12 times higher in the cold year (Stabeno et al. 2012). Cold years are known to be more favorable for large copepods and euphausiids over the Bering Sea shelf (Stabeno et al. 2012) and fin whale distributions are likely driven by availability of preferred prey.

Based on whaling data, the historical range of fin whales extended into the southern Sea of Okhotsk and Chukchi Sea. It was assumed that they passed through the Bering Strait into the southwestern Chukchi Sea during August and September. Many fin whales were taken as far west as Mys (Cape) Shmidta (68°55'N, 179°24'E) and as far north as 69°04'N, 171°06'W (Mizroch et al. 2009). Fin whale sightings have been increasing during surveys conducted in the U.S. portion of the northern Chukchi Sea from July to October (Funk et al. 2010, Aerts et al. 2012, Clarke et al. 2013, Brower et al. 2018) and fin whale calls were recorded each year from 2007 to 2010 in August and September in the northeastern Chukchi Sea (Delarue et al. 2013) and August to October just north of the Bering Strait (Tsujii et al. 2016), suggesting they may be re-occupying habitat used prior to large-scale commercial whaling. A comparison of data from aerial surveys that covered the same general areas between 1982 and 1991 and between 2008 and 2016 found no fin whale sightings in the earlier time period as compared to regular sightings of fin whales in the latter (Brower et al 2018). In part, this could be due to increased effort from 2008 to 2016; however, the combination of acoustic and visual data seem to support increasing numbers and extended seasonal residency of fin whales in the Alaska Arctic.

The following information was considered in classifying stock structure based on the Dizon et al. (1992) phylogeographic approach: 1) Distributional data: geographic distribution continuous in winter, possibly isolated in summer; 2) Population response data: unknown; 3) Phenotypic data: unknown; and 4) Genotypic data: unknown. Based on this limited information, the International Whaling Commission (IWC) considers fin whales in the North Pacific to all belong to the same stock (Mizroch et al. 1984), although Mizroch et al. (1984) cited additional evidence that supported the establishment of subpopulations in the North Pacific. Further, Fujino (1960) described eastern and western groups, which are mostly isolated with the exception of potential intermingling around the Aleutian Islands. Recoveries of Discovery tags (Rice 1974, Mizroch et al. 2009) indicate that animals wintering off the coast of southern California range from central California to the Gulf of Alaska during the summer months.

Mizroch et al. (2009) provided a comprehensive summary of whaling catch data, recovery of Discovery tags, and opportunistic sightings data and found evidence to suggest there may be at least six populations of fin whales: two that are migratory (eastern and western North Pacific) and two to four more that are resident year-round in peripheral seas such as the Gulf of California, East China Sea, Sanriku-Hokkaido, and possibly the Sea of Japan. It appears likely that the two migratory stocks mingle in the Bering Sea in July and August, rather than in the Aleutian Islands as Fujino (1960) previously concluded (Mizroch et al. 2009). During winter months, fin whales have been seen over a wide geographic area from 23°N to 60°N, but winter distribution and location of primary wintering areas (if any) are poorly known and need further study. As a result, stock structure of fin whales remains uncertain.

For management purposes, three stocks of fin whales are currently recognized in U.S. Pacific waters: 1) Alaska (Northeast Pacific) (Fig. 1), 2) California/Washington/Oregon, and 3) Hawaii. Mizroch et al. (2009) suggest that this structure should be reviewed and updated, if appropriate, to reflect recent analyses, but the absence of any substantial new data on stock structure makes this difficult. The California/Oregon/Washington and Hawaii fin whale stocks are reported in the Stock Assessment Reports for the U.S. Pacific Region.

POPULATION SIZE

There are no reliable estimates of current and historical abundances for the entire Northeast Pacific fin whale stock. Several studies provide information on the distribution and occurrence of fin whales in the Northeast Pacific, as well as estimates of abundance in certain areas within the range of the stock, however, many of these are over a decade or more old.

Visual shipboard surveys for cetaceans were conducted on the eastern Bering Sea shelf during summer in 1997, 1999, 2000, 2002, 2004, 2008, and 2010 (Moore et al. 2000, 2002; Friday et al. 2012, 2013). These surveys were conducted in conjunction with the Alaska Fisheries Science Center (AFSC) echo-integrated trawl surveys for walleye pollock. The surveys covered 789 to 3,752 km of tracklines and observation effort for marine mammals varied according to the availability of observers during each cruise. Results of the surveys in 2002, 2008, and 2010, years when the entire AFSC pollock survey sampling area was surveyed (see Fig. 1), provided estimates of 419 (coefficient of variation (CV) = 0.33), 1,368 (CV = 0.34), and 1,061 (CV = 0.38) fin whales (Friday et al. 2013).

Dedicated line-transect cruises were conducted in coastal waters (as far as 85 km offshore) of western Alaska and the eastern and central Aleutian Islands in July and August from 2001 to 2003 (Zerbini et al. 2006). Over 9,053 km of tracklines were surveyed between the Kenai Peninsula (150°W) and Amchitka Pass (178°W). Fin whales (n = 276) were observed from east of Kodiak Island to Samalga Pass, with high aggregations recorded near the Semidi Islands. Zerbini et al. (2006) estimated that 1,652 fin whales (95% CI: 1,142-2,389) occurred in these areas between 2001 and 2003.

In 2013 and 2015, dedicated line-transect surveys of the offshore waters of the Gulf of Alaska recorded, respectively, 171 and 38 sightings of fin whales (Rone et al. 2017). These surveys provided fin whale abundance estimates of 3,168 fin whales (CV = 0.26) in 2013 and 916 (CV = 0.39) in 2015. The marked differences in these estimates can be partially explained by differences in sampling coverage across the two cruises (Rone et al. 2017).

Estimates of fin whale abundance in the eastern Bering Sea and in the Gulf of Alaska in any given year cannot be considered representative of the entire Northeast Pacific stock because the geographic coverage of surveys was limited relative to the range of the stock. In addition, these estimates have not been corrected for animals missed on the trackline, animals submerged when the ship passed, and responsive movement away from or towards the survey vessel. However, even though no data are available to compute correction factors, it is expected that these estimates are robust because previous studies have shown that these sources of bias are small for this species (Barlow 1995).

Minimum Population Estimate

Although the full range of the Northeast Pacific stock of fin whales in Alaska waters has not been surveyed, a rough estimate of the size of the population west of the Kenai Peninsula has been calculated in previous Stock Assessment Reports by summing the estimates from Moore et al. (2002) and Zerbini et al. (2006) (n = 5,700). However, based on analyses presented in Mizroch et al. (2009), whales surveyed in the Aleutians (Zerbini et al. 2006) could migrate northward and be counted during the Bering Sea surveys. There are also indications that fin whale distribution in the Bering Sea is related to oceanographic conditions and prey density (Stabeno et al. 2012, Friday et al. 2013, Zerbini et al. 2016), making it possible that whales could be double counted when estimates from different years are summed (Moore et al. 2002). Until recently, the best provisional estimate of the fin whale population west and north of the Kenai Peninsula in U.S. waters was 1,368 whales, the greater of the minimum estimates from the 2008 and 2010 surveys (Friday et al. 2013). However, the Gulf of Alaska surveys (Rone et al. 2017) are more recent. The higher of the two abundances computed for fin whales in this region, 3,168 whales (CV = 0.26), better represents a minimum abundance for the Northeast Pacific stock because it is more precise and because it represents a broader survey coverage. A minimum population estimate (N_{MIN}) for this stock can be calculated according to Equation 1 from the potential biological removal (PBR) guidelines (NMFS 2016): $N_{MIN} = N/\exp(0.842\times[\ln(1+[CV(N)]^2)]^{½})$. Using the best provisional estimate (N) of 3,168 from the 2013 survey and the associated CV(N) of 0.26 results in an N_{MIN} of 2,554 whales. However, this is an underestimate for the entire stock because it is based on surveys which covered only a small portion of the stock's range.

Current Population Trend

Zerbini et al. (2006) estimated rates of increase of fin whales in coastal waters south of the Alaska Peninsula (Kodiak and Shumagin Islands). An annual increase of 4.8% (95% CI: 4.1-5.4%) was estimated between 1987 and 2003. This estimate is the first available for North Pacific fin whales and is consistent with other estimates of population growth rates of large whales. It should be used with caution, however, due to uncertainties in the initial population estimate (in 1987) and due to uncertainties about the population structure of fin whales in the area. Also, the study represented only a small fraction of the range of the Northeast Pacific stock and it may not be appropriate to extrapolate this to a broader range.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Zerbini et al. (2006) estimated an annual increase of 4.8% (95% CI: 4.1-5.4%) between 1987 and 2003 for fin whales in coastal waters south of the Alaska Peninsula. However, there are uncertainties in the initial population estimate from 1987, as well as uncertainties regarding fin whale population structure in this area. Therefore, a reliable estimate of the maximum net productivity rate (R_{MAX}) is not available for the Northeast Pacific fin whale stock. Until additional data become available, the default cetacean maximum theoretical net productivity rate of 4% will be used for this stock (NMFS 2016).

POTENTIAL BIOLOGICAL REMOVAL

PBR is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: PBR = $N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 0.1, the recommended value for cetacean stocks that are listed as endangered (NMFS 2016). Using the best provisional estimate of 3,168 (CV = 0.26) from the 2013 survey and the associated N_{MIN} of 2,554, PBR is calculated to be 5.1

fin whales (2,554 × 0.02 × 0.1). However, because the estimate of minimum abundance is for only a small portion of the stock's range, the calculated PBR is likely biased low for the entire Northeast Pacific fin whale stock.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2014 and 2018 is listed, by marine mammal stock, in Young et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports. The minimum estimated mean annual level of human-caused mortality and serious injury for Northeast Pacific fin whales between 2014 and 2018 is 0.6 whales due to ship strikes. Ship strikes are a known threat for this stock and reductions in sea-ice coverage may lead to range extension and increased susceptibility to ship strikes from increased shipping in the Chukchi and Beaufort seas.

Fisheries Information

Information for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is available in Appendix 3 of the Alaska Stock Assessment Reports (observer coverage) and in the NMFS List of Fisheries (LOF) and the fact sheets linked to fishery names in the LOF (observer coverage and reported incidental takes of marine mammals: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-list-fisheries, accessed December 2020).

No incidental mortality or serious injury of Northeast Pacific fin whales due to interactions with fisheries in Alaska waters was reported to the NMFS Alaska Region marine mammal stranding network between 2014 and 2018.

Table 1. Summary of mortality and serious injury of Northeast Pacific fin whales, by year and type, reported to the NMFS Alaska Region marine mammal stranding network between 2014 and 2018 (Young et al. 2020).

Cause of injury	**2014**	**2015**	**2016**	**2017**	**2018**	**Mean annual mortality**
Ship strike	1	0	1	0	1	0.6
Total due to ship strikes						0.6

Alaska Native Subsistence/Harvest Information

Subsistence hunters in Alaska and Russia have not been reported to take fin whales from this stock.

Other Mortality

Between 1900 and 1999, 75,538 fin whales were reportedly killed in commercial whaling operations throughout the North Pacific (Rocha et al. 2014).

In 2015, increased mortality of large whales was observed along the western Gulf of Alaska (including the areas around Kodiak Island, Afognak Island, Chirikof Island, the Semidi Islands, and the southern shoreline of the Alaska Peninsula) and along the central British Columbia coast (from the northern tip of Haida Gwaii to southern Vancouver Island). NMFS declared an Unusual Mortality Event (UME) for large whales that occurred from 22 May to 31 December 2015 in the western Gulf of Alaska and from 23 April 2015 to 16 April 2016 in British Columbia (https://www.fisheries.noaa.gov/national/marine-life-distress/active-and-closed-unusual-mortality-events, accessed December 2020). Forty-six large whale deaths attributed to the UME included 12 fin whales and 22 humpback whales in Alaska and 5 fin whales and 7 humpback whales in British Columbia. Based on the findings from the investigation, the UME was likely caused by ecological factors (i.e., the 2015 El Niño, Warm Water Blob, and Pacific Coast Domoic Acid Bloom).

Fin whale mortality due to ship strikes in Alaska waters was reported to the NMFS Alaska Region marine mammal stranding network in 2014, 2016, and 2018 (Young et al. 2020), resulting in a minimum mean annual mortality and serious injury rate of 0.6 fin whales due to ship strikes between 2014 and 2018 (Table 1).

STATUS OF STOCK

The fin whale is listed as endangered under the Endangered Species Act of 1973, and therefore designated as depleted under the MMPA. As a result, the Northeast Pacific stock is classified as a strategic stock. While estimates of the minimum population size and population trends are available for a portion of this stock, much of the

North Pacific range has not been surveyed. Therefore, the status of the stock relative to its Optimum Sustainable Population is not available. The minimum estimated mean annual level of human-caused mortality and serious injury for Northeast Pacific fin whales (0.6 whales) does not exceed the calculated PBR (5.1 whales). The minimum estimated mean annual rate of U.S. commercial fishery-related mortality and serious injury (0 whales) is less than 10% of the calculated PBR (10% of PBR = 0.5) and, therefore, can be considered insignificant and approaching a zero mortality and serious injury rate.

There are key uncertainties in the assessment of the Northeast Pacific stock of fin whales. While a single stock of fin whales is currently recognized in the Northeast Pacific, fin whale acoustic data suggest that multiple stocks overlap in the Bering Sea. Little is known about the pelagic distribution of fin whales due to the lack of dedicated marine mammal survey effort in the Bering Sea and Gulf of Alaska. The calculated PBR level is likely biased low because only a portion of the range has been surveyed. A reliable estimate of the trend in abundance is not available for this stock.

HABITAT CONCERNS

Changes in ocean conditions that affect the seasonal distribution and quality of prey may affect fin whale movements, distribution, and foraging energetics. Ship strikes are a known source of mortality, and reductions in sea-ice coverage may lead to range extension and concomitant exposure to increased shipping and oil and gas activities in the Bering and Chukchi seas. Ocean warming may increase the frequency of algal blooms that produce biotoxins known to be associated with large whale mortality. However, few data are available to assess the likelihood or extent of such impacts.

CITATIONS

Aerts, L. A. M., A. Kirk, C. Schudel, B. Watts, P. Seiser, A. Mcfarland, and K. Lomac-MacNair. 2012. Marine mammal distribution and abundance in the northeastern Chukchi Sea, July-October 2008-2011. Report prepared by LAMA Ecological for ConocoPhillips Alaska, Inc., Shell Exploration and Production Company and Statoil USA E&P, Inc. 69 p.

Archer, F. I., S. Rankin, K. M. Stafford, M. Castellote, and J. Delarue. 2019. Quantifying spatial and temporal variation of North Pacific fin whale (*Balaenoptera physalus*) acoustic behavior. Mar. Mammal Sci. DOI: dx.doi.org/10.1111/mms.12640 .

Barlow, J. 1995. The abundance of cetaceans in California waters. Part 1: Ship surveys in summer and fall of 1991. Fish. Bull., U.S. 93:1-14.

Barlow, J. 2006. Cetacean abundance in Hawaiian waters estimated from a summer/fall survey in 2002. Mar. Mammal Sci. 22(2):446-464.

Brower, A. A., J. T. Clarke, and M. C. Ferguson. 2018. Increased sightings of subArctic cetaceans in the eastern Chukchi Sea, 2008-2016: population recovery, response to climate change, or increased survey effort? Polar Biol. 41:1033-1039. DOI: dx.doi.org/10.1007/s00300-018-2257-x .

Clarke, J., K. Stafford, S. E. Moore, B. Rone, L. Aerts, and J. Crance. 2013. Subarctic cetaceans in the southern Chukchi Sea: evidence of recovery or response to a changing ecosystem. Oceanography 26(4):136-149. DOI: dx.doi.org/10.5670/oceanog.2013.81 .

Delarue, J., B. Martin, D. Hannay, and C. Berchok. 2013. Acoustic occurrence and affiliation of fin whales detected in the northeastern Chukchi Sea, July to October 2007–2010. Arctic 66(2):159-172.

Dizon, A. E., C. Lockyer, W. F. Perrin, D. P. DeMaster, and J. Sisson. 1992. Rethinking the stock concept: a phylogeographic approach. Conserv. Biol. 6:24-36.

Friday, N. A., A. N. Zerbini, J. M. Waite, and S. E. Moore. 2012. Cetacean distribution and abundance in relation to oceanographic domains on the eastern Bering Sea shelf: 1999-2004. Deep-Sea Res. II 65-70:260-272. DOI: dx.doi.org/10.1016/j.dsr2.2012.02.006 .

Friday, N. A., A. N. Zerbini, J. M. Waite, S. E. Moore, and P. J. Clapham. 2013. Cetacean distribution and abundance in relation to oceanographic domains on the eastern Bering Sea shelf in June and July of 2002, 2008, and 2010. Deep-Sea Res. II 94:244-256. DOI: dx.doi.org/10.1016/j.dsr2.2013.03.011 .

Fujino, K. 1960. Monogenetic and marking approaches to identifying sub-populations of the North Pacific whales. Sci. Rep. Whales Res. Inst. Tokyo 15:84-142.

Funk, D. W., D. S. Ireland, R. Rodrigues, and W. R. Koski (eds.). 2010. Joint monitoring program in the Chukchi and Beaufort seas, open water seasons, 2006–2008. LGL Alaska Report P1050-2, Report from LGL Alaska Research Associates, Inc., LGL Ltd., Greeneridge Sciences, Inc., and JASCO Research, Ltd., for Shell Offshore, Inc. and Other Industry Contributors, and National Marine Fisheries Service, U.S. Fish and Wildlife Service. 506 p. + appendices.

McDonald, M. A., and C. G. Fox. 1999. Passive acoustic methods applied to fin whale population density estimation. J. Acoust. Soc. Am. 105(5):2643-2651.
Mizroch, S. A., D. W. Rice, and J. M. Breiwick. 1984. The fin whale, *Balaenoptera physalus*. Mar. Fish. Rev. 46(4):20-24.
Mizroch, S. A., D. Rice, D. Zwiefelhofer, J. Waite, and W. Perryman. 2009. Distribution and movements of fin whales in the North Pacific Ocean. Mammal Rev. 39(3):193-227.
Mobley, J. R., Jr., M. Smultea, T. Norris, and D. Weller. 1996. Fin whale sighting north of Kaua'i, Hawai'i. Pac. Sci. 50(2):230-233.
Moore, S. E., K. M. Stafford, M. E. Dahlheim, C. G. Fox, H. W. Braham, J. J. Polovina, and D. E. Bain. 1998. Seasonal variation in reception of fin whale calls at five geographic areas in the North Pacific. Mar. Mammal Sci. 14(3):617-627.
Moore, S. E., J. M. Waite, L. L. Mazzuca, and R. C. Hobbs. 2000. Provisional estimates of mysticete whale abundance on the central Bering Sea shelf. J. Cetacean Res. Manage. 2(3):227-234.
Moore, S. E., J. M. Waite, N. A. Friday, and T. Honkalehto. 2002. Distribution and comparative estimates of cetacean abundance on the central and south-eastern Bering Sea shelf with observations on bathymetric and prey associations. Prog. Oceanogr. 55(1-2):249-262.
Moore, S. E., K. M. Stafford, D. K. Mellinger, and C. G. Hildebrand. 2006. Listening for large whales in the offshore waters of Alaska. BioScience 56(1):49-55.
National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2020.
Rice, D. W. 1974. Whales and whale research in the eastern North Pacific, p. 170-195. *In* W. E. Schevill (ed.), The Whale Problem: A Status Report. Harvard Press, Cambridge, MA.
Rocha, R. C., Jr., P. J. Clapham, and Y. V. Ivashchenko. 2014. Emptying the oceans: a summary of industrial whaling catches in the 20th century. Mar. Fish. Rev. 76:37-48. DOI: dx.doi.org/10.7755/MFR.76.4.3 .
Rone, B. K., A. N. Zerbini, A. B. Douglas, D. W. Weller, and P. J. Clapham. 2017. Abundance and distribution of cetaceans in the Gulf of Alaska. Mar. Biol. 164:23. DOI: dx.doi.org/10.1007/s00227-016-3052-2 .
Shallenberger, E. W. 1981. The status of Hawaiian cetaceans. Final Report for MMC Contract MM7AC028. Natl. Tech. Info. Ser. PB82-109398.
Širović, A., L. N. Williams, S. M. Kerosky, S. M. Wiggins, and J. A. Hildebrand. 2013. Temporal separation of two fin whale call types across the eastern North Pacific. Mar. Biol. 160:47-57.
Širović, A., A. Rice, E. Chou, J. A. Hildebrand, S. M. Wiggins, and M. A. Roch. 2015. Seven years of blue and fin whale call abundance in the Southern California Bight. Endang. Species Res. 28:61-76.
Soule, D. C., and W. S. D. Wilcock. 2013. Fin whale tracks recorded by a seismic network on the Juan de Fuca Ridge, Northeast Pacific Ocean. J. Acoust. Soc. Am. 133(3):1751-1761.
Springer, A. M., C. P. McRoy, and M. V. Flint. 1996. The Bering Sea green belt: shelf-edge processes and ecosystem production. Fish. Oceanogr. 5:205-223.
Stabeno, P., S. Moore, J. Napp, M. Sigler, and A. Zerbini. 2012. Comparison of warm and cold years on the southeastern Bering Sea shelf and some implications for the ecosystem. Deep-Sea Res. II 65-70:31-45.
Stafford, K. M., D. K. Mellinger, S. E. Moore, and C. G. Fox. 2007. Seasonal variability and detection range modeling of baleen whale calls in the Gulf of Alaska, 1999-2002. J. Acoust. Soc. Amer. 122(6):3378-3390.
Stafford, K. M., S. E. Moore, P. J. Stabeno, D. V. Holliday, J. M. Napp, and D. K. Mellinger. 2010. Biophysical ocean observation in the southeastern Bering Sea. Geophys. Res. Lett. 37:L02606. DOI: dx.doi.org/10.1029/2009GL040724 .
Thompson, P. O., and W. A. Friedl. 1982. A long term study of low frequency sound from several species of whales off Oahu, Hawaii. Cetology 45:1-19.
Tsujii, K., M. Otsuki, T. Akamatsu, I. Matsuo, K. Amakasu, M. Kitamura, T. Kikuchi, K. Miyashita, and Y. Mitani. 2016. The migration of fin whales into the southern Chukchi Sea as monitored with passive acoustics. ICES Journal of Marine Science 73(8):2085-2092. DOI: dx.doi.org/10.1093/icesjms/fsv271 .
Watkins, W. A., M. A. Daher, G. M. Reppucci, J. E. George, D. L. Martin, N. A. DiMarzio, and D. P. Gannon. 2000. Seasonality and distribution of whale calls in the North Pacific. Oceanography 13(1):62-67.
Young, N. C., B. J. Delean, V. T. Helker, J. C. Freed, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2014-2018. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-413, 142 p.

Zerbini, A. N., J. M. Waite, J. L. Laake, and P. R. Wade. 2006. Abundance, trends and distribution of baleen whales off western Alaska and the central Aleutian Islands. Deep-Sea Res. I 53(11):1772-1790.

Zerbini, A. N., N. A. Friday, D. M. Palacios, J. M. Waite, P. H. Ressler, B. K. Rone, S. E. Moore, and P. J. Clapham. 2016. Baleen whale abundance and distribution in relation to environmental variables and prey density in the eastern Bering Sea. Deep-Sea Res. II 134:312-330. DOI: dx.doi.org/10.1016/j.dsr2.2015.11.002 .

Revised 12/30/2018

MINKE WHALE (*Balaenoptera acutorostrata*): Alaska Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

In the North Pacific Ocean, minke whales occur from the Bering and Chukchi seas south to near the Equator (Leatherwood et al. 1982). The following information was considered in classifying stock structure according to the Dizon et al. (1992) phylogeographic approach: 1) Distributional data: geographic distribution continuous; 2) Population response data: unknown; 3) Phenotypic data: unknown; and 4) Genotypic data: unknown. Based on this limited information, in 1991 the International Whaling Commission (IWC) recognized three stocks of minke whales in the North Pacific: one in the Sea of Japan/East China Sea, one in the rest of the western Pacific west of 180°N, and one in the "remainder" of the Pacific (Donovan 1991). The "remainder" stock designation reflects the lack of exploitation in the eastern Pacific and does not indicate that only one population exists in this area (Donovan 1991). In the "remainder" area, minke whales are relatively common in the Bering and Chukchi seas and in the inshore waters of the Gulf of Alaska (Moore et al 2000, Friday et al. 2012, Clarke et al. 2013) but are not considered abundant in any other part of the eastern Pacific (Leatherwood et al. 1982, Brueggeman et al. 1990). Visual and acoustic data found minke whales in the Chukchi Sea north of Bering Strait in July and August (Clarke et al. 2013), and minke whale "boing" sounds have been detected in the northeast Chukchi Sea in August, October, and November (Delarue 2013). There are two types of geographically distinct boing sounds produced by minke whales in the North Pacific (Rankin and Barlow 2005). Those recorded in the Chukchi Sea matched "central Pacific" boing sounds leading the authors to hypothesize that minke whales from the Chukchi Sea might winter in the central North Pacific, not near Hawaii (Delarue et al. 2013).

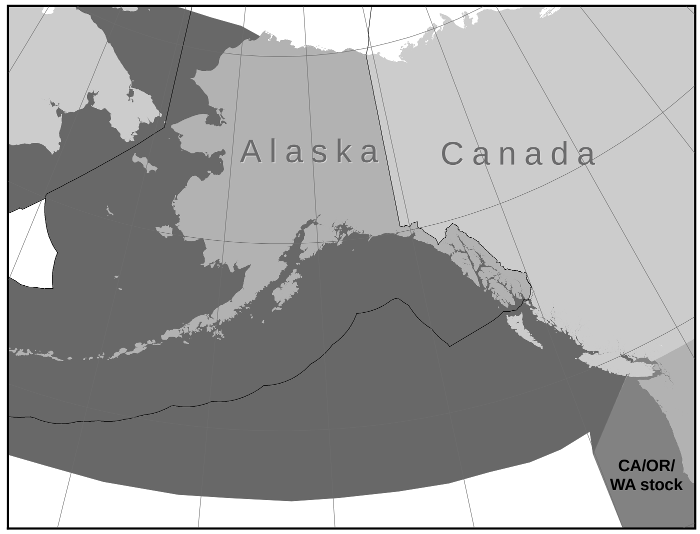

Figure 1. Approximate distribution of minke whales in the eastern North Pacific (dark shaded areas). The U.S. Exclusive Economic Zone is delineated by the solid black line.

Ship surveys on the eastern Bering Sea shelf in 1999, 2000, 2002, 2004, 2008, and 2010 resulted in new information about the distribution and relative abundance of minke whales in this area (Moore et al. 2002; Friday et al. 2012, 2013). When comparing distribution and abundance in years when the entire study area was surveyed (2002, 2008, and 2010), Friday et al. (2013) found that minke whales were scattered throughout the study area in all oceanographic domains (coastal, middle shelf, and outer shelf/slope) in 2002 and 2008 but were concentrated in the outer shelf and slope in 2010. The highest minke whale abundance in the study area occurred in 2010 and abundance was greater in cold years (2008 and 2010) than a warm year (2002); however, changes in abundance were thought to be due at least in part to changes in distribution (Friday et al. 2013).

So few minke whales were seen during three offshore Gulf of Alaska surveys for cetaceans in 2009, 2013, and 2015 that a population estimate for the species in this area could not be determined (Rone et al. 2017).

In the northern part of their range, minke whales are believed to be migratory, whereas, they appear to establish home ranges in the inland waters of Washington and along central California (Dorsey et al. 1990). Because the "resident" minke whales from California to Washington appear behaviorally distinct from migratory whales farther north, minke whales in Alaska are considered a separate stock from minke whales in California, Oregon, and Washington (Dorsey et al. 1990). Accordingly, two stocks of minke whales are recognized in U.S. waters: 1) Alaska, and 2) California/Washington/Oregon (Fig. 1). The California/Oregon/Washington minke whale stock is reported in the Stock Assessment Reports for the U.S. Pacific Region.

POPULATION SIZE

No estimates have been made for the number of minke whales in the entire North Pacific. However, some information is available on the numbers of minke whales in some areas of Alaska. Visual surveys for cetaceans were conducted on the eastern Bering Sea shelf in 2002, 2008, and 2010 in cooperation with research on commercial fisheries (Friday et al. 2013). The surveys included 3,752 km, 3,253 km, and 1,638 km of effort in 2002, 2008, and 2010, respectively. Results of the surveys in 2002, 2008, and 2010 provide provisional abundance estimates of 389 (CV = 0.52), 517 (CV = 0.69), and 2,020 (CV = 0.73) minke whales on the eastern Bering Sea shelf, respectively (Friday et al. 2013). These estimates are considered provisional because they have not been corrected for animals missed on the trackline, animals submerged when the ship passed, or responsive movement. Additionally, line-transect surveys were conducted in shelf and nearshore waters (within 30-45 nautical miles of land) in 2001-2003 from the Kenai Fjords in the Gulf of Alaska to the central Aleutian Islands. Minke whale abundance was estimated to be 1,233 (CV = 0.34) for this area (Zerbini et al. 2006). This estimate has also not been corrected for animals missed on the trackline. The majority of the sightings were in the Aleutian Islands, rather than in the Gulf of Alaska, and in water shallower than 200 m. So few minke whales were seen during three offshore Gulf of Alaska surveys for cetaceans in 2009, 2013, and 2015 that a population estimate for the species in this area could not be determined (Rone et al. 2017). These estimates cannot be used as an estimate of the entire Alaska stock of minke whales because only a portion of the stock's range was surveyed.

Minimum Population

It is not possible to produce a reliable estimate of minimum abundance for this stock, as current estimates of abundance are not available.

Current Population Trend

There are no data on trends in minke whale abundance in Alaska waters.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

There are no estimates of the growth rate of minke whale populations in the North Pacific (Best 1993). Until additional data become available, the cetacean maximum theoretical net productivity rate (R_{MAX}) of 4% will be used for this stock (Wade and Angliss 1997).

POTENTIAL BIOLOGICAL REMOVAL

Potential biological removal (PBR) is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: $PBR = N_{MIN} \times 0.5R_{MAX} \times F_R$. Given the status of this stock is unknown, the appropriate recovery factor (F_R) is 0.5 (Wade and Angliss 1997). However, because an estimate of minimum abundance is not available, the PBR for the Alaska minke whale stock is unknown.

ANNUAL HUMAN-CAUSED MORTALITY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals in 2012-2016 is listed, by marine mammal stock, in Helker et al. (in press); however, only the mortality and serious injury data are included in the Stock Assessment Reports. The total estimated annual level of human-caused mortality and serious injury for Alaska minke whales in 2012-2016 is zero.

Fisheries Information

Information on U.S. commercial fisheries in Alaska waters (including observer programs, observer coverage, and observed incidental takes of marine mammals) is presented in Appendices 3-6 of the Alaska Stock Assessment Reports.

No mortality or serious injury of minke whales was observed in U.S. commercial fisheries in 2012-2016 (Breiwick 2013; MML, unpubl data).

Alaska Native Subsistence/Harvest Information

No minke whales were ever taken by the modern shore-based whale fishery in the eastern North Pacific, which lasted from 1905 to 1971 (Rice 1974). Subsistence takes of minke whales by Alaska Natives are rare but have been known to occur. Only seven minke whales are reported to have been taken for subsistence by Alaska Natives between 1930 and 1987 (C. Allison, International Whaling Commission, UK, pers. comm.). The most recent reported catches (two whales) in Alaska occurred in 1989 (Anonymous 1991), but reporting is likely incomplete. Based on this information, the average annual subsistence take was zero minke whales in 2012-2016.

Other Mortality

From 2012 to 2016, no human-related mortality or serious injury of minke whales was reported to the NMFS Alaska Region stranding network (Helker et al. in press).

STATUS OF STOCK

Minke whales are not designated as depleted under the Marine Mammal Protection Act or listed as threatened or endangered under the Endangered Species Act. The abundance estimate for this stock is unknown and, thus, PBR is unknown. However, because minke whales are considered common in the waters off Alaska and human-caused mortality and serious injury is thought to be minimal, this stock is presumed to be a non-strategic stock. Because the PBR is unknown, the mean annual U.S. commercial fishery-related mortality and serious injury rate that can be considered insignificant and approaching zero mortality and serious injury rate is unknown. Population trends and status of this stock relative to its Optimum Sustainable Population are unknown.

There are key uncertainties in the assessment of the Alaska stock of minke whales. The greatest uncertainty is the stock structure of this species in the eastern North Pacific. Differences in abundance in warm and cold years on the eastern Bering Sea shelf (due at least in part to changes in distribution) are an additional source of uncertainty. Reliable estimates of the minimum population size, population trends, and PBR are not available.

HABITAT CONCERNS

Potential concerns include elevated levels of sound from anthropogenic sources (e.g., shipping, military sonars), possible changes in prey distribution with climate change, entanglement in fishing gear, ship strikes due to increased vessel traffic (e.g., from increased shipping in higher latitudes), and oil and gas activities.

CITATIONS

Anonymous. 1991. International Whaling Commission Report. Rep. Int. Whal. Comm. 41:1-2.

Best, P. B. 1993. Increase rates in severely depleted stocks of baleen whales. ICES J. Mar. Sci. 50:169-186.

Breiwick, J. M. 2013. North Pacific marine mammal bycatch estimation methodology and results, 2007-2011. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-260, 40 p.

Brueggeman, J. J., G. A. Green, K. C. Balcomb, C. E. Bowlby, R. A. Grotefendt, K. T. Briggs, M. L. Bonnell, R. G. Ford, D. H. Varoujean, D. Heinemann, and D. G. Chapman. 1990. Oregon-Washington marine mammal and seabird survey: information synthesis and hypothesis formulation. U.S. Dep. Interior, Outer Continental Shelf Study, Minerals Management Service 89-0030.

Clarke, J., K. Stafford S. E. Moore, B. Rone, L. Aerts, and J. Crance. 2013. Subarctic cetaceans in the southern Chukchi Sea: evidence of recovery or response to a changing ecosystem. Oceanography 26(4):136-149.

Delarue, J., B. Martin, and D. Hannay. 2013. Minke whale boing sound detections in the northeastern Chukchi Sea. Mar. Mammal Sci. 29:E333–E341.

Dizon, A. E., C. Lockyer, W. F. Perrin, D. P. DeMaster, and J. Sisson. 1992. Rethinking the stock concept: a phylogeographic approach. Conserv. Biol. 6:24-36.

Donovan, G. P. 1991. A review of IWC stock boundaries. Rep. Int. Whal. Comm. (Special Issue 13):39-68.

Dorsey, E. M., S. J. Stern, A. R. Hoelzel, and J. Jacobsen. 1990. Minke whales (*Balaenoptera acutorostrata*) from the west coast of North America: individual recognition and small scale site fidelity. Rep. Int. Whal. Comm. (Special Issue 12):357-368.

Friday, N. A., J. M. Waite, A. N. Zerbini, and S. E. Moore. 2012. Cetacean distribution and abundance in relation to oceanographic domains on the eastern Bering Sea shelf: 1999–2004. Deep-Sea Res. II 65-70:260-272.

Friday, N. A., A. N. Zerbini, J. M. Waite, S. E. Moore, and P. J. Clapham. 2013. Cetacean distribution and abundance in relation to oceanographic domains on the eastern Bering Sea shelf in June and July of 2002, 2008, and 2010. Deep-Sea Res. II 94:244-256.

Helker, V. T., M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. In press. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2012-2016. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-XXX, XXX p.

Leatherwood, S., R. R. Reeves, W. F. Perrin, and W. E. Evans. 1982. Whales, dolphins, and porpoises of the eastern North Pacific and adjacent Arctic waters: a guide to their identification. U.S. Dep. Commer., NOAA Tech. Rep. NMFS Circular 444, 245 p.

Moore, S. E., J. M. Waite, L. L. Mazzuca, and R. C. Hobbs. 2000. Provisional estimates of mysticete whale abundance on the central Bering Sea shelf. J. Cetacean Res. Manage. 2(3):227-234.

Moore, S. E., J. M. Waite, N. A. Friday and T. Honkalehto. 2002. Distribution and comparative estimates of cetacean abundance on the central and south-eastern Bering Sea shelf with observations on bathymetric and prey associations. Prog. Oceanogr. 55(1-2):249-262.

Rankin, S., and J. Barlow. 2005. Source of the North Pacific "boing" sound attributed to minke whales. J. Acoust. Soc. Am. 118:3346–3351.

Rice, D. W. 1974. Whales and whale research in the eastern North Pacific, p. 170-195. *In* W. E. Schevill (ed.), The Whale Problem: A Status Report. Harvard Press, Cambridge, MA.

Rone, B. K., A. N. Zerbini, A. B. Douglas, D. W. Weller, and P. J. Clapham. 2017. Abundance and distribution of cetaceans in the Gulf of Alaska. Mar. Biol. 164:23. DOI: dx.doi.org/10.1007/s00227-016-3052-2 .

Wade, P. R., and R. Angliss. 1997. Guidelines for assessing marine mammal stocks: report of the GAMMS Workshop April 3-5, 1996, Seattle, Washington. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-OPR-12, 93 p.

Zerbini, A. N., J. M. Waite, J. L. Laake, and P. R. Wade. 2006. Abundance, trends, and distribution of baleen whales off western Alaska and the central Aleutian Islands. Deep-Sea Res. I 53:1772-1790.

NORTH PACIFIC RIGHT WHALE (*Eubalaena japonica*): Eastern North Pacific Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Once distributed widely across the North Pacific from North America to the Far East, North Pacific right whales (*Eubalaena japonica*) are today among the world's rarest marine mammals (Wade et al. 2011). A distinct geographic distribution, different catch and recovery histories, and recent genetic analysis have led to the generally accepted belief that the species comprises eastern and western populations that are largely or wholly discrete (Brownell et al. 2001, LeDuc et al. 2012). The summer range of the eastern stock includes the Gulf of Alaska and the Bering Sea, while the western stock is believed to feed in the Okhotsk Sea and in pelagic waters of the northwestern North Pacific. The winter calving grounds of both stocks remain unknown.

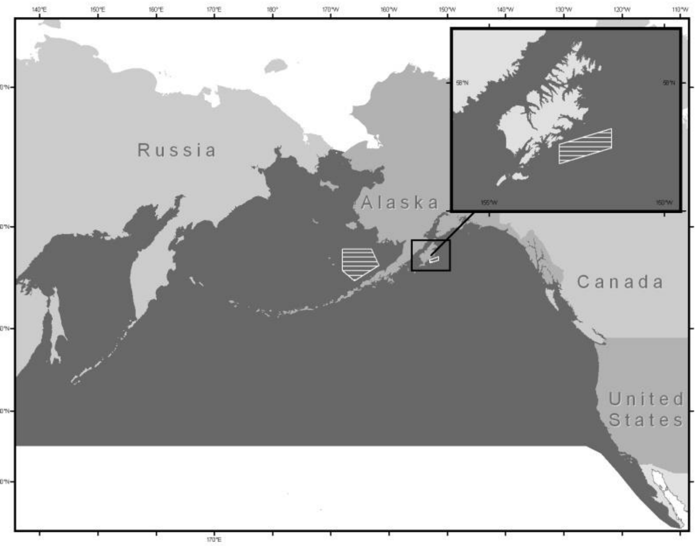

Figure 1. Approximate historical distribution of North Pacific right whales in the North Pacific (dark shaded area). Striped areas indicate North Pacific right whale critical habitat (73 FR 19000, 8 April 2008).

Right whales were the subject of intensive commercial exploitation, beginning in the Gulf of Alaska in 1835, and by 1849 were already seriously depleted in the eastern Pacific (Scarff 1986, 1991; Josephson et al. 2008). Additional hunting in the 1850s reduced the population in the western Pacific, and by 1900 the species was effectively considered commercially extinct throughout its range. Although there were sporadic opportunistic catches in the early 20th century, the stock was likely undergoing a modest recovery by about 1960; however, this was entirely negated by large illegal catches by the U.S.S.R. in the 1960s, which likely wiped out the bulk of the eastern population (Ivashchenko and Clapham 2012, Ivashchenko et al. 2017).

Analysis of whaling records from the 19th century, together with the more recent Soviet catches, has shown that right whales were broadly distributed across the eastern North Pacific (Scarff 1986, Brownell et al. 2001, Ivashchenko and Clapham 2012). There are sporadic records from below 20°N, but the bulk of the data show right whales concentrated north of 35°N. This includes coastal and offshore waters ranging from Washington State and British Columbia through the Gulf of Alaska, Alaska Peninsula, Aleutian Islands, and Bering Sea.

Modern information on the summer and autumn distribution of right whales has been derived from dedicated vessel and aerial surveys, bottom-mounted acoustic recorders, and vessel surveys for fisheries ecology and management that have also included dedicated marine mammal observers. Aerial and vessel surveys for right whales (LeDuc et al. 2001, Wade et al. 2006, Clapham et al. 2013) have occurred in a portion of the southeastern Bering Sea (Fig. 1) where right whales have been observed or acoustically detected in most summers since 1996 (Goddard and Rugh 1998, Munger et al. 2008, Rone et al. 2012, Wright 2017). North Pacific right whales have been observed consistently in this area, although it is clear from historical and Japanese sighting survey data (Fig. 2) that right whales often range outside this area and occur elsewhere in the Bering Sea (Scarff 1986, Moore et al. 2000, 2002; LeDuc et al. 2001; Clapham et al. 2004). Because of the paucity of right whales in the eastern North Pacific, sightings today are relatively rare and are often of single individuals (Fig. 2). In the summer of 2017, however, the International Whaling Commission's (IWC) Pacific Ocean Whale and Ecosystem Research (POWER) survey used a combination of passive acoustic monitoring and visual sightings to find 15 right whales in the southeastern Bering Sea (Matsuoka et al. 2017). The majority of these sightings (10 of 15 animals) were in Bristol Bay approximately 60 nmi east of the North Pacific right whale critical habitat, with others in the critical habitat itself. Three additional right whales were sighted during the 2018 IWC POWER survey (Matsuoka et al. 2018). Two were within the critical habitat, while the third was sighted approximately 5 nmi south of St. Lawrence Island, in the northern Bering Sea.

Bottom-mounted acoustic recorders were deployed in the southeastern Bering Sea (2000-present) and the northern Gulf of Alaska (1999-2001) to document the seasonal distribution of right whale calls. Analysis of the data from those recorders supports the survey data and shows that right whales remain in the southeastern Bering Sea from May through December with peak call detection in September (Mellinger et al. 2004, Munger et al. 2008, Stafford and Mellinger 2009, Stafford et al. 2010, Clapham et al. 2013, Wright 2017, Wright et al. 2019). Recorders deployed by the Alaska Fisheries Science Center's Marine Mammal Laboratory indicated that North Pacific right whales occurred in two passes of the eastern Aleutian Islands (Umnak and Unimak Pass) (Wright 2017, Wright et al. 2018). No North Pacific right whale calls were detected from January to April in the southeastern Bering Sea, which supports the theory that North Pacific right whales migrate out of the Bering Sea during winter months (Wright 2017).

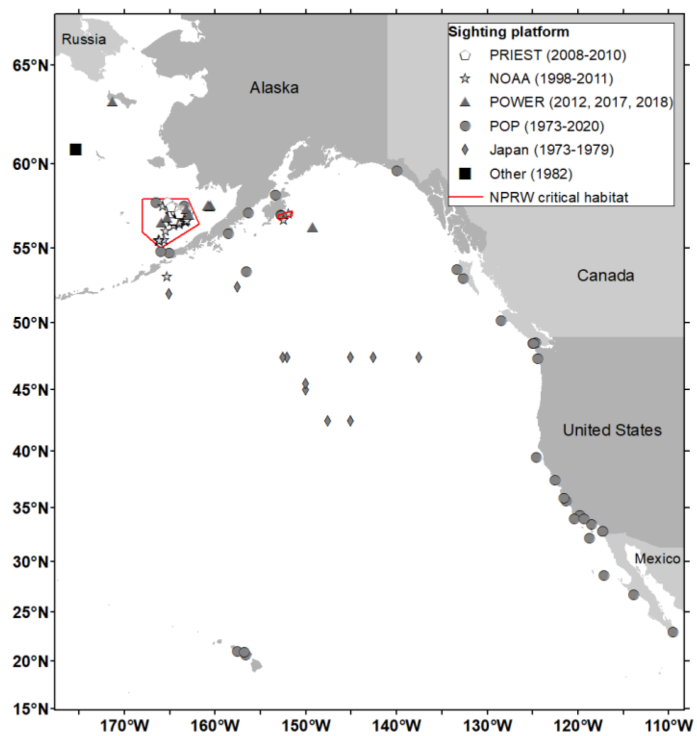

Figure 2. Location of all Eastern North Pacific right whale sightings in the North Pacific by platform since 1970. PRIEST = BOEM-NOAA (Pacific RIght whale Ecology STudy) survey; NOAA = other NOAA surveys; POWER = IWC's Pacific Ocean Whale and Ecosystem Research survey; POP = opportunistic sighting documented in MML's Platforms of Opportunity database; Japan = Japanese sighting survey; Other = Bering Sea (Navarin Basin) survey (Brueggeman et al. 1984).

There continues to be debate regarding the northern extent of the right whale's range, specifically whether they once commonly occurred in the northern Bering Sea and north of the Bering Strait. Records from historical whaling in such areas are often compromised by uncertainty regarding whether these could have been bowhead whales; the extent of overlap between the two species remains unclear. In recent years, there have been a few reliable records of right whales in this region: an individual right whale was visually identified north of St. Lawrence Island in November 2012, an individual was sighted on 26 June 2018 by hunters off of St. Lawrence Island on the northeast side of Sivuqaq mountain (G. Sheffield, University of Alaska Fairbanks, Nome, AK), and the IWC POWER cruise recorded a single right whale just south of St. Lawrence in July 2018 (Matsuoka et al. 2018). This latter individual was subsequently observed and photographed by an ecotourism cruise in Pengkingney Fjord in Russian waters just south of the Bering Strait (D. Brown, Heritage Expeditions). Passive acoustic monitoring from 2008 to 2016 of the northern Bering Sea detected calls matching the North Pacific right whale up-call criterion in late fall through spring only in 2016 (Wright et al. 2019). It remains unknown whether these recent northern detections and sightings represent a reoccupation of their historical distribution or a northward shift in their distribution.

There have been far fewer sightings of right whales in the Gulf of Alaska than in the Bering Sea (Brownell et al. 2001); although, until the summer of 2015, survey effort was lacking in the Gulf, notably in the offshore areas where right whales commonly occurred during whaling days (Ivashchenko and Clapham 2012). Nonetheless, sightings in the Gulf of Alaska since the cessation of whaling are extremely rare (Fig. 2), and there have been only a few acoustic detections (Mellinger et al. 2004, Širović et al. 2015).

Three separate surveys have occurred in the Gulf of Alaska in the summer. In summer 2013, the U.S. Navy-funded Gulf of Alaska Line-Transect Survey (GOALS-II) surveyed for marine mammals within the Temporary Maritime Activities Area (TMAA) using visual line-transect methods and passive acoustic monitoring (Rone et al. 2014). In August 2015, a dedicated vessel survey for right whales was conducted by NMFS using visual and acoustic survey techniques, surveying both the shelf and deeper waters to the south (Rone et al. 2017). And in summer 2019, the IWC POWER cruise systematically surveyed the northern Gulf of Alaska, within the U.S. Exclusive Economic Zone, from Umnak Pass in the Aleutian Islands to the Canadian border in the eastern North

Pacific (Matsuoka et al. 2020). In all three surveys, right whales were acoustically detected in the Barnabus Trough area off Kodiak Island, but were not visually observed.

Most of the illegal Soviet catches of right whales occurred in offshore areas, including a large area to the east and southeast of Kodiak Island (Doroshenko 2000, Ivashchenko and Clapham 2012); the Soviet catch distribution closely parallels that seen in plots of 19th-century American whaling catches by Townsend (1935). Whether this region remains an important habitat for this species is currently unknown. The sightings and acoustic detection of right whales in coastal waters east of Kodiak Island indicate at least occasional use of this area; however, the lack of visual detections of right whales during the GOALS-II cruise in July 2013, the NMFS cruise in August 2015, and the IWC POWER cruise in 2019 adds to the concern that right whales may today be extremely rare in the Gulf of Alaska. To date, there have been no matches of photographically identified individuals between the Gulf of Alaska and the Bering Sea, and there is no information to address the question of whether these regions are connected or whether they form largely separate subpopulations.

As noted above, the location of winter calving grounds for North Pacific right whales has long been a mystery. North Atlantic (*E. glacialis*) and Southern Hemisphere (*E. australis*) right whales calve in coastal waters during the winter months. However, in the eastern North Pacific no such calving grounds have been identified (Scarff 1986). Migratory patterns of North Pacific right whales are unknown, although it is thought they migrate from high-latitude feeding grounds in summer to more temperate waters during the winter, possibly including offshore waters (Braham and Rice 1984, Scarff 1986, Clapham et al. 2004). A right whale sighted off Maui in April 1996 (Salden and Michelsen 1999) was identified 119 days later and 4,111 km north in the Bering Sea (Kennedy et al. 2011); to date this is the only low- to high-latitude match of an individually identified right whale in the eastern North Pacific. There is one other modern record from Hawaii of a right whale, an animal seen twice in March and April 1979 (Herman et al. 1980, Rowntree et al. 1980) (Fig. 2).

Although there were a handful of sightings of right whales in the eastern North Pacific from Japanese sighting surveys in the 1970s (Fig. 2), sightings in that area since then have been extremely rare. Two sightings of individual right whales occurred off British Columbia in 2013, one in June and one in October (Ford et al. 2016). The two different individuals represent the first right whale sightings in Canadian waters since the 1950s. Another right whale sighting was made by the Canadian Coast Guard in the same area in June 2018. Most recently, a right whale was sighted off Vancouver Island in May 2020. The timing of these sightings lends support to the theory that right whales migrate to more temperate waters during the winter.

Occasional sightings of right whales have been made off California and off Baja California, Mexico (Fig. 2); this includes two recent records from California in 2017, off La Jolla and in the Channel Islands (both of which were single whales). While the scarcity of records from this region superficially suggests (as did Brownell et al. 2001) that it lacked historical importance for the species, this ignores the fact that right whales had been severely depleted in their feeding grounds prior to 1854, when the first coastal whaling station was established in California. It remains possible that California and Mexico, and possibly offshore waters of Hawaii, were once the principal calving grounds for right whales from the Gulf of Alaska and Bering Sea.

The following information was considered in classifying stock structure according to the Dizon et al. (1992) phylogeographic approach: 1) Distributional data: distinct geographic distribution; 2) Population response data: unknown; 3) Phenotypic data: unknown; and 4) Genotypic data: evidence for some isolation of populations. Based on this limited information, two stocks of North Pacific right whales are currently recognized: a Western North Pacific stock (feeding primarily in the Sea of Okhotsk) and an Eastern North Pacific stock (feeding primarily in the southeastern Bering Sea) (Rosenbaum et al. 2000, Brownell et al. 2001, LeDuc et al. 2012).

In summary, the range of the right whale in the North Pacific was historically broad, with feeding grounds in the Bering Sea, Gulf of Alaska, Okhotsk Sea, and northwestern North Pacific; all of these areas remain inhabited today from May to December.

POPULATION SIZE

The historical (pre-whaling) population size of the North Pacific right whale is unknown. However, Scarff (1991) estimated that 26,500 to 37,000 animals were killed during the period from 1839 to 1909, with the majority being taken in a single decade (1840 to 1849). The U.S.S.R. illegally killed an estimated 771 right whales in the eastern and western North Pacific, with the majority (662) killed between 1962 and 1968 (Ivashchenko et al. 2017). These takes severely impacted the two populations concerned, notably in the east (Ivashchenko and Clapham 2012, Ivashchenko et al. 2013). Of the 662 right whales killed in the 1960s, 517 were taken in the eastern North Pacific, including 366 in the Gulf of Alaska, 31 in the Aleutian Islands, 116 in the Bering Sea, and 4 in unspecified pelagic waters (Ivashchenko et al. 2013).

Earlier estimates of population size were at best speculative. Based on sighting data, Wada (1973) estimated a total population of 100-200 right whales in the North Pacific in 1970. Rice (1974) stated that only a few individuals remained in the Eastern North Pacific stock and that for all practical purposes the stock was extinct because no sightings of a mature female with a calf had been confirmed since 1900. However, various sightings made since 1996 have invalidated this view (Wade et al. 2006, Zerbini et al. 2015, Ford et al. 2016, Matsuoka et al. 2017). Brownell et al. (2001) suggested from a review of sighting records that the abundance of this species in the western North Pacific was likely in the "low hundreds," including the population in the Sea of Okhotsk.

The North Pacific Right Whale Photo-identification Catalogue currently contains a minimum of 26 individual whales from the eastern North Pacific. From 2008 to 2018, 26 right whales were photographically identified, some repeatedly (Clapham et al. 2013; Ford et al. 2016; Matsuoka et al. 2017, 2018). Including individuals observed more than once across years, this comprises 8 animals photographed in 2008 (all in the Bering Sea), 7 in 2009 (Bering Sea), 3 in 2010 (1 in the Bering Sea, 2 off Kodiak), 2 in 2011 (Bering Sea), 1 in 2012 (Gulf of Alaska), 2 in 2013 (both off British Columbia), 14 in 2017 (12 in the Bering Sea, 1 in Kodiak, 1 in the Channel Islands), and 3 in the Bering Sea in 2018. The number of unique right whales decreased from previous years as a result of obtaining better quality photographs that allowed for additional internal matches in the catalogue.

LeDuc et al. (2012) analyzed 49 biopsy samples from 24 individual right whales, all but one of which were from the eastern North Pacific. The analysis revealed a male-biased sex ratio and a loss of genetic diversity that appeared to be midway between that observed for right whales in the North Atlantic and the Southern Hemisphere. The analysis also suggested a degree of separation between eastern and western populations, a male:female ratio of 2:1, and a low effective population size for the Eastern North Pacific stock, which LeDuc et al. (2012) considered to be at "extreme risk" of extirpation. Six biopsy samples were obtained from right whales in the Bering Sea during the IWC POWER cruises (3 in 2017, 3 in 2018), all from individuals of previously unknown sex. None were obtained during the 2019 cruise. Of the six whales sampled, five were male and only one was female. This suggests that the sex ratio may in fact be more skewed toward males than previously believed, which would put the population at even greater risk. These samples have not yet been integrated into the overall sample for reanalysis; while this may change the male:female ratio, it is unlikely to change the overall conclusions of LeDuc et al. (2012).

The only recent estimate of abundance comes from mark-recapture analyses of photo-identification and genetic data. Photographic (18 identified individuals) and genotype (21 identified individuals) data through 2008 were used to calculate the first mark-recapture estimates of abundance for right whales in the Bering Sea and Aleutian Islands, resulting in separate estimates of 31 (95% CL: 23-54; CV = 0.22) and 28 (95% CL: 24-42), respectively (Wade et al. 2011). The abundance estimates are for the last year of each study, corresponding to 2008 for the photo-identification estimate and 2004 for the genetic identification estimate. Wade et al. (2011) also estimated that the population consisted of 8 females (95% CL: 7-18) and 20 males (95% CL: 17-37).

The Wade et al. (2011) estimates may relate to a subpopulation that uses the Bering Sea; there is no estimate for right whales in the Gulf of Alaska, and to date there have been no photo-identification matches between the two regions. Consequently, the total size of the Eastern North Pacific population may be somewhat higher than the Wade et al. (2011) estimates. However, given the extreme paucity of recent sightings in the Gulf of Alaska, it seems unlikely that the overall abundance is significantly larger.

Minimum Population Estimate

The minimum estimate of abundance (N_{MIN}) of Eastern North Pacific right whales is 26 whales based on the 20th percentile of the photo-identification estimate of 31 whales (CV = 0.226: Wade et al. 2011). This estimate will be 12 years old in 2020, and the 2016 guidelines for preparing Stock Assessment Reports (NMFS 2016) recommend that N_{MIN} be considered unknown if the abundance estimate is more than 8 years old; however, given the extremely low abundance of this stock and the very low calf production, it seems unlikely that the current abundance is significantly different.

Current Population Trend

Due to a low resighting rate and the extremely low population size, no estimate of trend in abundance is available for this stock.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Due to insufficient information, the default cetacean maximum theoretical net productivity rate (R_{MAX}) of 4% is used for this stock (NMFS 2016). However, given the small apparent size, male bias, and very low calf production in this population, this rate is likely to be unrealistically high.

POTENTIAL BIOLOGICAL REMOVAL

Potential biological removal (PBR) is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: PBR = $N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock is 0.1, the recommended value for cetacean stocks which are listed as endangered (NMFS 2016). A reliable estimate of N_{MIN} for this stock is 26 whales based on the mark-recapture estimate of 31 whales (CV = 0.226: Wade et al. 2011). The calculated PBR level for this stock is therefore 0.05 ($26 \times 0.02 \times 0.1$), which would be equivalent to one take every 20 years. However, the male bias likely results in lower than expected calf production and, thus, this PBR could be overestimated.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2014 and 2018 is listed, by marine mammal stock, in Young et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports. No human-caused mortality or serious injury of Eastern North Pacific right whales was reported between 2014 and 2018; although, given the remote nature of the known and likely habitats of North Pacific right whales, it is very unlikely that any mortality or serious injury in this population would be observed. Consequently, it is possible that the current absence of reported mortality or serious injury due to entanglement in fishing gear, ship strikes, or other anthropogenic causes (e.g., oil spills) is not a reflection of the true situation.

Fisheries Information

Information for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is available in Appendix 3 of the Alaska Stock Assessment Reports (observer coverage) and in the NMFS List of Fisheries (LOF) and the fact sheets linked to fishery names in the LOF (observer coverage and reported incidental takes of marine mammals: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-list-fisheries, accessed December 2020).

There are no historical reports of fisheries-caused mortality or serious injury of Eastern North Pacific right whales. However, given what we know about susceptibility of other large whales to fisheries-caused mortality and serious injury, we assume that the potential exists for North Pacific right whales. Mortality and serious injury of humpback whales and fin whales in trawl gear, gray whales in gillnet gear, and bowhead whales in pot gear (George et al. 2017) has been documented. While much of the trawl fleet has observer coverage, several gillnet fisheries and pot fisheries in the range of Eastern North Pacific right whales do not. Therefore, the potential for fisheries-caused mortality and serious injury may be greater than is reflected in existing observer data.

Right whales, presumably from the Western North Pacific population, have suffered fisheries-caused mortality or serious injury. Gillnets were implicated in the death of a right whale off the Kamchatka Peninsula (Russia) in October of 1989 (Kornev 1994). The Marine Mammal Commission reported that in February 2015, a young right whale was found entangled in aquaculture gear in South Korea; much of the gear was cut off, but the whale’s fate is unknown. In October 2016, an entangled right whale was reported to have died while being disentangled in Volcano Bay, Hokkaido, Japan. And in July 2018, fishermen in the Sea of Okhotsk took video of a right whale that was entangled in the rope of a crab pot but later freed itself. No other incidental takes of right whales are known to have occurred in the North Pacific, although two photographs from the North Pacific Right Whale Photo-identification Catalogue show possible fishing gear entanglement (A. Kennedy, NMFS-AFSC-MML, pers. comm., 21 September 2011; Ford et al. 2016). The right whale photographed on 25 October 2013 off British Columbia and northern Washington State showed evidence of probable fishing gear entanglement (Ford et al. 2016). Given the very small estimate of abundance, any mortality or serious injury incidental to commercial fisheries would be considered significant. Entanglement in fishing gear, including lobster pot and sink gillnet gear, is a significant source of mortality and serious injury for North Atlantic right whales (Waring et al. 2014).

Alaska Native Subsistence/Harvest Information

Subsistence hunters in Alaska and Russia do not hunt animals from this stock.

Other Mortality

Ship strikes are considered one of the primary sources of human-caused mortality and serious injury of right whales in the North Atlantic (Cole et al. 2005; Henry et al. 2012, 2019; Hayes et al. 2018), and it is possible that right whales in the North Pacific are also vulnerable to this source of mortality. However, due to their rare occurrence and scattered distribution, it is impossible to assess the threat of ship strikes to the Eastern North Pacific stock of right whales. There is concern that increased shipping through Arctic waters and the Bering Sea, with retreating sea ice, may increase the potential risk to right whales from shipping.

Overall, given the remote nature of the known and likely habitats of North Pacific right whales, it is very unlikely that any mortality or serious injury in this population would be observed. Consequently, it is possible that the current absence of reported ship-strike-related or other anthropogenic mortality or serious injury in this stock is not a reflection of the true situation.

STATUS OF STOCK

The right whale is listed as endangered under the Endangered Species Act of 1973, and therefore designated as depleted under the Marine Mammal Protection Act. In 2008, NMFS relisted the North Pacific right whale as endangered as a separate species (*Eubalaena japonica*) from the North Atlantic species, *E. glacialis* (73 FR 12024, 06 March 2008). As a result, the stock is classified as a strategic stock. The abundance of this stock is considered to represent only a small fraction of its pre-commercial whaling abundance, i.e., the stock is well below its Optimum Sustainable Population (OSP). The minimum estimated mean annual level of human-caused mortality and serious injury is unknown for this stock. The reason(s) for the apparent lack of recovery for this stock is (are) unknown. Brownell et al. (2001) and Ivashchenko and Clapham (2012) noted the devastating impact of extensive illegal Soviet catches in the eastern North Pacific in the 1960s, and both suggested that the prognosis for right whales in this area was poor. Biologists working aboard the Soviet factory ships that killed right whales in the eastern North Pacific in the 1960s considered that the fleets had caught close to 100% of the animals they encountered (Ivashchenko and Clapham 2012); accordingly, it is quite possible that the Soviets killed the great majority of the animals in the population at that time. In its review of the status of right whales worldwide, the IWC expressed "considerable concern" over the status of this population (IWC 2001), which is currently the most endangered stock of large whales in the world for which an abundance estimate is available. A genetic analysis of biopsy samples from North Pacific right whales found an apparent loss of genetic diversity, low frequencies of females and calves, extremely low effective population size, and possible isolation from conspecifics in the western Pacific indicating that right whales in the eastern North Pacific are in severe danger of immediate extirpation from the eastern North Pacific (LeDuc et al. 2012).

There are key uncertainties in the assessment of the Eastern North Pacific stock of North Pacific right whales. The abundance of this stock is critically low and migration patterns, calving grounds, and breeding grounds are not well known. There appear to be considerably more males than females in the population and calf production is very low. 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, and that some surplus growth could be removed while still allowing recovery. However, the Eastern North Pacific right whale population is far below historical levels and at a very small population size, and small populations can have different dynamics than larger populations from Allee effects and stochastic dynamics. Although there is currently no known direct human-caused mortality, given the small number of animals estimated to be in the population, any human-caused mortality or serious injury from ship strikes or commercial fisheries is likely to have a serious population-level impact.

HABITAT CONCERNS

NMFS conducted an analysis of right whale distribution in historical times and in more recent years and stated that principal habitat requirements for right whales are dense concentrations of prey (Clapham et al. 2006) and, on this basis, proposed two areas of critical habitat: one in the southeastern Bering Sea and another south of Kodiak Island (70 FR 66332, 2 November 2005). In 2006, NMFS issued a final rule designating these two areas as northern right whale critical habitat, one in the Gulf of Alaska and one in the Bering Sea (71 FR 38277, 6 July 2006; Fig. 1). In 2008, NMFS redesignated the same two areas as Eastern North Pacific right whale critical habitat under the newly recognized species name, *E. japonica* (73 FR 19000, 8 April 2008; Fig. 1).

Potential threats to the habitat of this population derive primarily from commercial shipping and fishing vessel activity. There is considerable fishing activity within portions of the critical habitat of this species, increasing the risk of entanglement. However, photographs of right whales in the eastern North Pacific to date have shown little evidence of entanglement scars; the sole exception is the animal photographed in the Strait of Juan de Fuca in October 2013 (Ford et al. 2016). Unimak Pass is a choke-point for shipping traffic between North America and Asia, with shipping density and risk of an accidental spill highest in the summer (Renner and Kuletz 2015), a time when right whales are believed to be present (Wright et al. 2018). The high volume of large vessels transiting Unimak Pass (e.g., 1,961 making 4,615 transits in 2012: Nuka Research and Planning Group, LLC 2014a, 2014b), a subset of which continue north through the Bering Sea, increases both the risk of ship strikes and the risk of a large or very large oil spill in areas in which right whales may occur. The risk of accidents in Unimak Pass, specifically, is predicted to increase in the coming decades, and studies indicate that more accidents are likely to involve container vessels (Wolniakowski et al. 2011).

Past offshore oil and gas leasing has occurred in the Gulf of Alaska and Bering Sea in the northern areas of known right whale habitat. The Bureau of Ocean Energy Management (BOEM) proposed an Outer Continental Shelf leasing plan for 2007-2012 that prioritized lease sales for the North Aleutian Basin in 2010 and 2012 (Aplin and Elliott 2007), but it was later withdrawn by Presidential Executive Order. Therefore, the North Aleutian Basin was not included in the 2017-2022 national lease schedule by BOEM, and there are no residual active leases from past sales. However, BOEM has announced plans to replace the 2017-2022 OCS plan (with a new 2019-2024 leasing plan) and to reconsider all current moratoria on offshore oil and gas exploration and extraction (82 FR 30886, 3 July 2017). It is noteworthy that two tagged right whales were observed to briefly visit the North Aleutian Basin area, one in 2004 and one in 2009 (Zerbini et al. 2015). The development of oil fields off Sakhalin Island in Russia is occurring within habitat of the western North Pacific population of right whales (NMFS 2006). However, no oil exploration or production is currently underway in offshore areas of the Bering Sea or Gulf of Alaska, and no lease sales are currently scheduled to occur in those areas. The possibility remains that there will be lease sales in these areas in the future, even though no discoveries have yet been announced and most leases have not contained commercially viable deposits (NMFS 2006). However, in Cook Inlet, lease sales are planned (the next federal sale under the existing 2017-2022 leasing plan will occur in 2021 and state sales currently occur annually) and exploration activity is occurring in both state and federal waters. BOEM (2016) conducted an oil spill model for lower Cook Inlet that suggested if a very large oil spill occurs in offshore waters it will impact right whale habitat around Kodiak Island and along the Alaska Peninsula. Although there is currently no oil and gas activity in the Alaska Chukchi Sea, oil exploration and production is ongoing in the Beaufort Sea, and this will likely include an increased level of associated vessel traffic through the Bering Sea en route to and from the Arctic, which could increase risks to right whales from ship strikes.

CITATIONS

Aplin, D., and W. Elliott. 2007. Conservation concerns for cetaceans in the Bering Sea and adjacent waters: offshore oil development and other threats. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/59/E9). 14 p.

Braham, H. W., and D. W. Rice. 1984. The right whale, *Balaena glacialis*. Mar. Fish. Rev. 46(4):38-44.

Brownell, R. L., P. J. Clapham, T. Miyashita, and T. Kasuya. 2001. Conservation status of North Pacific right whales. J. Cetacean Res. Manage. (Special Issue 2):269-286.

Brueggeman, J. J., R. A. Grotefendt, and A. W. Erickson. 1984. Endangered whale abundance and distribution in the Navarin Basin of the Bering Sea during the ice-free period, p. 201-236. *In* B. R. Melteff and D. H. Rosenberg (eds.), Proceedings of the Workshop on Biological Interactions Among Marine Mammals and Commercial Fisheries in the Southeastern Bering Sea. University of Alaska Sea Grant Report 84-1.

Bureau of Ocean Energy Management (BOEM). 2016. Cook Inlet Planning Area Oil and Gas Lease Sale 244, in the Cook Inlet, Alaska - Final Environmental Impact Statement. Appendix A.

Clapham, P. J., C. Good, S. E. Quinn, R. R. Reeves, J. E. Scarff, and R. L. Brownell, Jr. 2004. Distribution of North Pacific right whales (*Eubalaena japonica*) as shown by 19th and 20th century whaling catch and sighting records. J. Cetacean Res. Manage. 6(1):1-6.

Clapham, P. J., K. E. W. Shelden, and P. R. Wade. 2006. Review of information relating to possible critical habitat for Eastern North Pacific right whales, p. 1-27. *In* P. J. Clapham, K. E. W. Shelden, and P. R. Wade (eds.), Habitat requirements and extinction risks of Eastern North Pacific right whales. AFSC Processed Report 2006-06. Available from Alaska Fisheries Science Center, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Clapham, P. J., A. S. Kennedy, B. K. Rone, A. N. Zerbini, J. L. Crance, and C. L. Berchok. 2013. North Pacific right whales in the southeastern Bering Sea: Final Report. U.S. Dep. Commer., OCS Study BOEM 2012-074. 175 p. Available from Marine Mammal Laboratory, AFSC, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Cole, T. V. N., D. L. Hartley, and R. M. Merrick. 2005. Mortality and serious injury determinations for large whale stocks along the eastern seaboard of the United States, 1999-2003. U.S. Dep. Commer., NEFSC Ref. Doc. 05-08, 20 p. Available from National Marine Fisheries Service, 166 Water Street, Woods Hole, MA 02543-1026.

Dizon, A. E., C. Lockyer, W. F. Perrin, D. P. DeMaster, and J. Sisson. 1992. Rethinking the stock concept: a phylogeographic approach. Conserv. Biol. 6:24-36.

Doroshenko, N. V. 2000. Soviet whaling for blue, gray, bowhead and right whales in the North Pacific Ocean, 1961-1979, p. 96-103. *In* A. V. Yablokov and V. A. Zemsky (eds.), Soviet Whaling Data (1949-1979). Center for Russian Environmental Policy, Marine Mammal Council, Moscow.

Ford, J. K. B., J. F. Pilkington, B. Gisborne, T. R. Frasier, R. M. Abernethy, and G. M. Ellis. 2016. Recent observations of critically endangered North Pacific right whales (*Eubalaena japonica*) off the west coast of Canada. Marine Biodiversity Records 9:50. DOI: dx.doi.org/10.1186/s41200-016-0036-3 .

George, J. C., G. Sheffield, D. J. Reed, B. Tudor, R. Stimmelmayr, B. T. Person, T. Sformo, and R. Suydam. 2017. Frequency of injuries from line entanglements, killer whales, and ship strikes on Bering-Chukchi-Beaufort seas bowhead whales. Arctic 70(1):37-46.

Goddard, P. C., and D. J. Rugh. 1998. A group of right whales seen in the Bering Sea in July 1996. Mar. Mammal Sci. 14(2):344-349.

Hayes, S. A., S. Gardner, L. Garrison, A. Henry, and L. Leandro. 2018. North Atlantic right whales – evaluating their recovery challenges in 2018. U.S. Dep. Commer., NOAA Tech Memo NMFS-NE-247, 24 p.

Henry, A. G., T. V. N. Cole, M. Garron, L. Hall, W. Ledwell, and A. Reid. 2012. Mortality and serious injury determinations for baleen whale stocks along the Gulf of Mexico, United States east coast and Atlantic Canadian provinces, 2006-2010. U.S. Dep. Commer., NEFSC Reference Document 12-11, 24 p. Available from National Marine Fisheries Service, 166 Water Street, Woods Hole, MA 02543-1026.

Henry, A., M. Garron, A. Reid, D. Morin, W. Ledwell, and T. V. N. Cole. 2019. Serious injury and mortality determinations for baleen whale stocks along the Gulf of Mexico, United States east coast, and Atlantic Canadian Provinces, 2012-2016. U.S. Dep. Commer, Northeast Fisheries Science Center Reference Document 19-13, 54 p. Available online: https://repository.library.noaa.gov/view/noaa/21249 . Accessed December 2020.

Herman, L. M., C. S. Baker, P. H. Forestell, and R. C. Antinoja. 1980. Right whale, *Balaena glacialis*, sightings near Hawaii: a clue to the wintering grounds? Mar. Ecol. Prog. Ser. 2:271-275.

International Whaling Commission (IWC). 2001. Report of the workshop on the comprehensive assessment of right whales: a worldwide comparison. J. Cetacean Res. Manage. (Special Issue 2):1-60.

Ivashchenko, Y. V., and P. J. Clapham. 2012. Soviet catches of right whales *Eubalaena japonica* and bowhead whales *Balaena mysticetus* in the North Pacific Ocean and the Okhotsk Sea. Endang. Species Res. 18:201-217.

Ivashchenko, Y. V., R. L. Brownell, Jr., and P. J. Clapham. 2013. Soviet whaling in the North Pacific: revised catch totals. J. Cetacean Res. Manage. 13:59-71.

Ivashchenko, Y. V., P. J. Clapham, and R. L. Brownell, Jr. 2017. New data on Soviet catches of blue (*Balaenoptera musculus*) and right whales (*Eubalaena japonica*) in the North Pacific. J. Cetacean Res. Manage. 17:15-22.

Josephson, E. A., T. D. Smith, and R. R. Reeves. 2008. Depletion within a decade: the American 19th-century North Pacific right whale fishery, p. 133-147. *In* D. J. Starkey, P. Holm, and M. Barnard (eds.), Oceans Past: Management Insights from the History of Marine Animal Populations. Earthscan, London.

Kennedy, A. S., D. R. Salden, and P. J. Clapham. 2011. First high- to low-latitude match of an Eastern North Pacific right whale (*Eubalaena japonica*). Mar. Mammal Sci. 28(4):E539-E544. DOI: dx.doi.org/10.1111/j.1748-7692.2011.00539.x .

Kornev, S. I. 1994. A note on the death of a right whale (*Eubalaena glacialis*) off Cape Lopakta (Kamchatka). Rep. Int. Whal. Comm. (Special Issue 15):443-444.

LeDuc, R. G., W. L. Perryman, J. W. Gilpatrick, Jr., J. Hyde, C. Stinchcomb, J. V. Carretta, and R. L. Brownell, Jr. 2001. A note on recent surveys for right whales in the southeastern Bering Sea. J. Cetacean Res. Manage. (Special Issue 2):287-289.

LeDuc, R. G., B. L. Taylor, K. Martien, K. M. Robertson, R. L. Pitman, J. C. Salinas, A. M. Burdin, A. S. Kennedy, P. R. Wade, P. J. Clapham, and R. L. Brownell, Jr. 2012. Genetic analysis of right whales in the eastern North Pacific confirms severe extinction risk. Endang. Species Res. 18:163-167.

Matsuoka, K., J. Taylor, I. Yoshimua, J. Crance, and H. Kasai. 2017. Cruise report of the 2017 IWC-Pacific Ocean Whale and Ecosystem Research (IWC-POWER). Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/67b/ASI/12). 46 p. Available from International Whaling Commission, Cambridge, UK, at https://iwc.int/power .

Matsuoka, K., J. Crance, A. James, I. Yoshimura, and H. Kasai. 2018. Cruise report of the 2018 IWC-Pacific Ocean Whale and Ecosystem Research (IWC-POWER). Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee SC/68a/ASI/04. Available from International Whaling Commission, Cambridge, UK, at https://iwc.int/power .

Matsuoka, K., J. Crance, J. W. Gilpatrick, Jr., I. Yoshimura, and C. Okoshi. 2020. Cruise report of the 2019 IWC-Pacific Ocean Whale and Ecosystem Research (IWC-POWER). Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee. Available from International Whaling Commission, Cambridge, UK, at https://iwc.int/power .

Mellinger, D. K., K. M. Stafford, S. E. Moore, L. Munger, and C. G. Fox. 2004. Detection of North Pacific right whale (*Eubalaena japonica*) calls in the Gulf of Alaska. Mar. Mammal Sci. 20:872-879.

Moore, S. E., J. M. Waite, L. L. Mazzuca, and R. C. Hobbs. 2000. Provisional estimates of mysticete whale abundance on the central Bering Sea shelf. J. Cetacean Res. Manage. 2(3):227-234.

Moore, S. E., J. M. Waite, N. A. Friday, and T. Honkalehto. 2002. Distribution and comparative estimates of cetacean abundance on the central and south-eastern Bering Sea shelf with observations on bathymetric and prey associations. Prog. Oceanogr. 55(1-2):249-262.

Munger, L. M., S. M. Wiggins, S. E. Moore, and J. A. Hildebrand. 2008. North Pacific right whale (*Eubalaena japonica*) seasonal and diel calling patterns from long-term acoustic recordings in the southeastern Bering Sea, 2000–2006. Mar. Mammal Sci. 24(4):795-814.

National Marine Fisheries Service (NMFS). 2006. Review of the status of the right whales in the North Atlantic and North Pacific Oceans. v + 62 p.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2020.

Nuka Research and Planning Group, LLC. 2014a. Aleutian Islands risk assessment project; recommending an optimal response system for the Aleutian Islands: summary report. 51 p.

Nuka Research and Planning Group, LLC. 2014b. Summary of large vessel transits of Unimak Pass in 2012. Aleutian Islands Risk Assessment Phase B.

Renner, M., and K. J. Kuletz. 2015. A spatial-seasonal analysis of the oiling risk from shipping traffic to seabirds in the Aleutian Archipelago. Mar. Pollut. Bull. 101(1):127-136.

Rice, D. W. 1974. Whales and whale research in the eastern North Pacific, p. 170-195. *In* W. E. Schevill (ed.), The Whale Problem: A Status Report. Harvard Press, Cambridge, MA.

Rone, B. K., C. L. Berchok, J. L. Crance, and P. J. Clapham. 2012. Using air-deployed passive sonobuoys to detect and locate critically endangered North Pacific right whales. Mar. Mammal Sci. 28:E528-E538.

Rone, B. K., A. B. Douglas, T. M. Yack, A. N. Zerbini, T. N. Norris, E. Ferguson, and J. Calambokidis. 2014. Report for the Gulf of Alaska Line-Transect Survey (GOALS) II: marine mammal occurrence in the Temporary Maritime Activities Area (TMAA). Submitted to Naval Facilities Engineering Command (NAVFAC) Pacific, Honolulu, Hawaii under Contract No. N62470-10-D-3011, Task Order 0022, issued to HDR Inc., San Diego, California. Prepared by Cascadia Research Collective, Olympia, WA; Alaska Fisheries Science Center, Seattle, WA; and Bio-Waves, Inc., Encinitas, CA. April 2014.

Rone, B. K., A. N. Zerbini, A. B. Douglas, D. W. Weller, and P. J. Clapham. 2017. Abundance and distribution of cetaceans in the Gulf of Alaska. Mar. Biol. 164:23. DOI: dx.doi.org/10.1007/s00227-016-3052-2 .

Rosenbaum, H. C., R. L. Brownell, M. W. Brown, C. Schaeff, V. Portway, B. N. White, S. Malik, L. A. Pastene, N. J. Patenaude, C. S. Baker, M. Goto, P. B. Best, P. J. Clapham, P. Hamilton, M. Moore, R. Payne, V. Rowntree, C. T. Tynan, J. L. Bannister, and R. DeSalle. 2000. World-wide genetic differentiation of *Eubalaena*: questioning the number of right whale species. Mol. Ecol. 9(11):1793-1802.

Rowntree, V., J. Darling, G. Silber, and M. Ferrari. 1980. Rare sighting of a right whale (*Eubalaena glacialis*) in Hawaii. Can. J. Zool. 58:308-312.

Salden, D. R., and J. Mickelsen. 1999. Rare sightings of a North Pacific right whale (*Eubalaena glacialis*) in Hawaii. Pac. Sci. 53:341-345.

Scarff, J. E. 1986. Historic and present distribution of the right whale (*Eubalaena glacialis*) in the eastern North Pacific south of 50°N and east of 180°W. Rep. Int. Whal. Comm. (Special Issue 10):43-63.

Scarff, J. E. 1991. Historic distribution and abundance of the right whale, *Eubalaena glacialis*, in the North Pacific, Bering Sea, Sea of Okhotsk and Sea of Japan from the Maury Whale Charts. Rep. Int. Whal. Comm. 41:467-487.

Širović, A., S. C. Johnson, L. K. Roche, L. M. Varga, S. M. Wiggins, and J. A. Hildebrand. 2015. North Pacific right whales (*Eubalaena japonica*) recorded in the northeastern Pacific Ocean in 2013. Mar. Mammal Sci. 31(2):800-807. DOI: dx.doi.org/10.1111/mms.12189 .

Stafford, K. M., and D. K. Mellinger. 2009. Analysis of acoustic and oceanographic data from the Bering Sea, May 2006 – April 2007. North Pacific Research Board Final Report, NPRB Project No. 719. 24 p.

Stafford, K. M., S. E. Moore, P. J. Stabeno, D. V. Holliday, J. M. Napp, and D. K. Mellinger. 2010. Biophysical ocean observation in the southeastern Bering Sea. Geophys. Res. Lett. 37(2). DOI: dx.doi.org/10.1029/2009GL040724 .

Townsend, C. H. 1935. The distribution of certain whales as shown by logbook records of American whaleships. Zoologica NY 19:1-50.

Wada, S. 1973. The ninth memorandum on the stock assessment of whales in the North Pacific. Rep. Int. Whal. Comm. 23:164-169.

Wade, P. R. 1998. Calculating limits to the allowable human-caused mortality of cetaceans and pinnipeds. Mar. Mammal Sci. 14:1-37. DOI: dx.doi.org/10.1111/j.1748-7692.1998.tb00688.x .

Wade, P. R., M. P. Heide-Jørgensen, K. Shelden, J. Barlow, J. Carretta, J. Durban, R. LeDuc, L. Munger, S. Rankin, A. Sauter, and C. Stinchcomb. 2006. Acoustic detection and satellite tracking leads to discovery of rare concentration of endangered North Pacific right whales. Biol. Lett. 2:417-419.

Wade, P. R., A. Kennedy, R. LeDuc, J. Barlow, J. Carretta, K. Shelden, W. Perryman, R. Pitman, K. Robertson, B. Rone, J. C. Salinas, A. Zerbini, R. L. Brownell, Jr., and P. Clapham. 2011. The world's smallest whale population. Biol. Lett. 7:83-85.

Waring, G. T., E. Josephson, K. Maze-Foley, and P. E. Rosel (eds.). 2014. U.S. Atlantic and Gulf of Mexico marine mammal stock assessments - 2013. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-NE-228, 464 p.

Wolniakowski, K. U., J. Wright, G. Folley, and M. R. Franklin. 2011. Aleutian Islands Risk Assessment Project. Phase A Summary Report. 58 p.

Wright, D. L. 2017. Passive acoustic monitoring of the critically endangered Eastern North Pacific right whale (*Eubalaena japonica*). Final Report to Marine Mammal Commission, 4340 East-West Highway, Suite 700, Bethesda, MD 20814. 58 p.

Wright, D. L., M. Castellote, C. L. Berchok, D. Ponirakis, J. L. Crance, and P. J. Clapham. 2018. Acoustic detection of North Pacific right whales in a high-traffic Aleutian Pass, 2009-2015. Endang. Species Res. 37:77-90. DOI: dx.doi.org/10.3354/esr00915 .

Wright, D. L., C. L. Berchok, J. L. Crance, and P. J. Clapham. 2019. Acoustic detection of the critically endangered North Pacific right whale in the northern Bering Sea. Mar. Mammal Sci. 35:311-326. DOI: dx.doi.org/10.1111/mms.12521 .

Young, N. C., B. J. Delean, V. T. Helker, J. C. Freed, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2014-2018. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-413, 142 p.

Zerbini, A. N., M. F. Baumgartner, A. S. Kennedy, B. K. Rone, P. R. Wade, and P. J. Clapham. 2015. Space use patterns of the endangered North Pacific right whale (*Eubalaena japonica*) in the Bering Sea. Mar. Ecol. Prog. Ser. 532:269-281. DOI: dx.doi.org/10.3354/meps11366 .

BOWHEAD WHALE (*Balaena mysticetus*): Western Arctic Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Western Arctic bowhead whales are distributed in seasonally ice-covered waters of the Arctic and near-Arctic, generally north of 60°N and south of 75°N in the western Arctic Basin (Braham 1984, Moore and Reeves 1993). For management purposes, four stocks of bowhead whales are recognized worldwide by the International Whaling Commission (IWC 2010). Small stocks, comprising only a few hundred individuals, occur in the Sea of Okhotsk and the offshore waters of Spitsbergen (Zeh et al. 1993, Shelden and Rugh 1995, Wiig et al. 2009, Shpak et al. 2014, Boertmann et al. 2015). Bowhead whales occur in western Greenland (Hudson Bay and Foxe Basin) and eastern Canada (Baffin Bay and Davis Strait), and evidence suggests that these should be considered one stock based on genetics (Postma et al. 2006, Bachmann et al. 2010, Heide-Jørgensen et al. 2010, Wiig et al. 2010), aerial surveys (Cosens et al. 2006), and tagging data (Dueck et al. 2006; Heide-Jørgensen et al. 2006; IWC 2010, 2011). This stock, previously thought to include only a few hundred animals, may number over a thousand (Heide-Jørgensen et al. 2006, Wiig et al. 2011), and perhaps over 6,000 (IWC 2008, Doniol-Valcroze et al. 2015, Frasier et al. 2015). The only stock found within U.S. waters is the Western Arctic stock (Fig. 1), also known as the Bering-Chukchi-Beaufort stock (Rugh et al. 2003) or Bering Sea stock (Burns et al. 1993). The IWC Scientific Committee concluded, in several reviews of the extensive genetic and satellite telemetry data, that the weight-of-evidence is most consistent with one bowhead whale stock that migrates throughout waters of northern and western Alaska and northeastern Russia (IWC 2008, 2018).

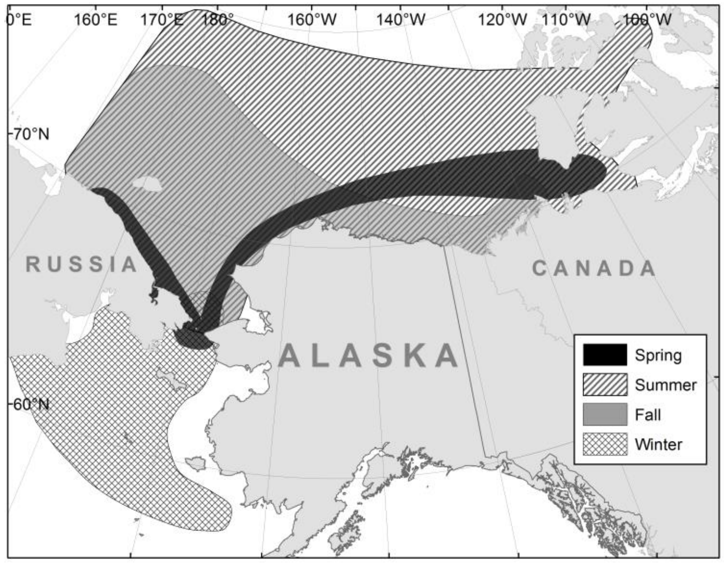

Figure 1. Annual range of the Western Arctic stock of bowhead whales by season from satellite tracking data, 2006-2017 (map based on Quakenbush et al. (2018): Fig. 2).

The majority of the Western Arctic stock migrates annually from wintering areas in the northern Bering and southern Chukchi seas (December to April), through the Chukchi Sea and Beaufort Sea in the spring (April through May), to the eastern Beaufort Sea (Fig. 1) where they spend much of the late spring and summer (May through September). During late summer and autumn (September through December), this stock migrates back to the Chukchi Sea and then to the Bering Sea (Fig. 1) to overwinter (Braham et al. 1980; Moore and Reeves 1993; Quakenbush et al. 2010a, 2018; Citta et al. 2015). During winter and spring, bowhead whales are closely associated with sea ice (Moore and Reeves 1993, Quakenbush et al. 2010a, Citta et al. 2015, Druckenmiller et al. 2018). The bowhead whale spring migration follows fractures in the sea ice along the coast to Point Barrow, generally in the shear zone between the shorefast ice and the mobile pack ice, then continues offshore on a direct path to the Cape Bathurst polynya (Citta et al. 2015). In most years, during summer, a large proportion of the population is in the relatively ice-free waters of Amundsen Gulf in the eastern Beaufort Sea (Citta et al. 2015), an area often exposed to industrial activity related to petroleum exploration (e.g., Richardson et al. 1987, Davies 1997). However, summer aerial surveys conducted in the western Beaufort Sea during July and August of 2012-2017 have had relatively high sighting rates of bowhead whales, including cows with calves and feeding animals (Clarke et al. 2018a, 2018b), suggesting interannual variability in bowhead whale summer distribution. Additionally, data from a satellite tagging study conducted between 2006 and 2018 indicated that, although most tagged whales began to leave the Canadian Beaufort Sea in September, the timing of their westward migration across the Beaufort Sea was highly variable; furthermore, all tagged whales observed in summer and fall in Beaufort and Chukchi waters near Point Barrow were known to have returned from Canada (Quakenbush and Citta 2019). Timing of the onset of the westward migration across the Beaufort Sea is associated with oceanographic conditions in the eastern Beaufort Sea (Citta et al. 2018, Clarke et al. 2018b). During the autumn migration, bowhead whales generally inhabit shelf waters across the Beaufort Sea (Citta et al. 2015). The autumn migration across the Chukchi Sea is more dispersed (Clarke et al. 2016); here, bowhead whales generally prefer cold, saline waters that are mostly of Bering Sea origin (Citta et al. 2018). During winter in the Bering Sea, bowhead whales often use areas covered by nearly 100% sea ice, even when polynyas are available (Quakenbush et al. 2010a, Citta et al. 2015).

Evidence from stomach contents and habitat associations suggests that Western Arctic bowhead whales feed on concentrations of zooplankton throughout their range. Likely or confirmed feeding areas include Amundsen Gulf and the eastern Beaufort Sea; the central and western Beaufort Sea; the Chukchi shelf break, especially Herald Valley and the Central Channel; and the coast of Chukotka between Wrangel Island and Bering Strait (Lowry et al. 2004; Ashjian et al. 2010; Clarke and Ferguson 2010; Quakenbush et al. 2010a, 2010b; Okkonen et al. 2011; Fish et al. 2013; Citta et al. 2015, 2018; Clarke et al. 2017; Harwood et al. 2017). Citta et al. (2015) identified six core use areas for Western Arctic bowhead whales based on bowhead whale satellite telemetry, oceanography, sea ice, and winds. During spring in the Cape Bathurst polyna, whales are found in water <75 m deep where calanoid copepods are ascending after diapause. In summer and into fall, bowhead whales inhabit shelf waters in the Beaufort Sea, including the Tuktoyaktuk shelf and areas farther west, where episodic wind-driven upwelling and high river discharge results in high densities of zooplankton (Citta et al. 2015, Harwood et al. 2017, Okkonen et al. 2018, Clarke et al. 2018b). During summer and fall, Western Arctic bowhead whales may congregate on the shallow shelf east of Point Barrow, where variable wind dynamics promote large aggregations of zooplankton onto the shelf (Ashjian et al. 2010, Okkonen et al. 2011, Citta et al. 2015). In winter, dive behavior suggests that bowhead whales feed in shelf waters of the Bering Sea, from Bering Strait south through Anadyr Strait, and near the seafloor in the Gulf of Anadyr (Citta et al. 2012, 2015). Of four bowhead whales harvested in November (two in 2012) and December (two in 2010) near St. Lawrence Island, in the northern Bering Sea, three had been feeding (Sheffield and George 2013). Results from mercury and stable isotope analysis are consistent with year-round foraging and seasonal migration of bowhead whales (Pomerleau et al. 2018).

Clarke et al. (2015) evaluated biologically important areas (BIAs) for bowhead whales in the U.S. Arctic region and identified nine BIAs based on satellite telemetry and aerial survey data. The four reproductive BIAs encompass areas where the majority of bowhead whales identified as calves were observed each season. The three feeding BIAs were located in the western Beaufort Sea. In most years, the krill trap area (Ashjian et al. 2010) from Smith Bay to Point Barrow is the most consistent feeding area for bowhead whales from August to October (Clarke et al. 2015). In other areas of the western Beaufort Sea, bowhead whales may feed in ephemeral prey patches on the continental shelf, out to approximately the 50 m isobath, in September and October. These ephemeral foraging areas are also evident in satellite telemetry data (Quakenbush and Citta 2019).

POPULATION SIZE

All stocks of bowhead whales were severely depleted during intense commercial whaling, starting in the early 16th century near Labrador, Canada (Ross 1993), and spreading to the Bering Sea in the mid-19th century (Braham 1984, Bockstoce and Burns 1993, Bockstoce et al. 2007). Woodby and Botkin (1993) summarized previous efforts to estimate bowhead whale population size prior to the onset of commercial whaling. They reported a minimum worldwide population estimate of 50,000, with 10,400 to 23,000 in the Western Arctic stock (dropping to less than 3,000 at the end of commercial whaling). Brandon and Wade (2006) used Bayesian model averaging to estimate that the Western Arctic stock consisted of 10,960 bowhead whales (9,190 to 13,950; 5th and 95th percentiles, respectively) in 1848 at the start of commercial whaling.

The recently adopted Aboriginal Whaling Scheme (IWC 2018) requires that abundance estimates be conducted every 10 years as input into the Strike Limit Algorithm (SLA) that the IWC approved for estimating a safe strike limit for aboriginal subsistence hunting. Ice-based visual and acoustic counts have been conducted since 1978 (Krogman et al. 1989; Table 1). These counts have been corrected for whales missed due to distance offshore since the mid-1980s, using acoustic methods described in (Clark et al. 1994). Correction factors were estimated for whales missed during a watch (due to visibility, number of observers, and offshore distance) and when no watch was in effect (through interpolations from sampled periods) (Zeh et al. 1993, Givens et al. 2016). The spring ice-based estimates of abundance have not been corrected for a small portion of the population that may not migrate past Point Barrow during the period when counts are made. According to Melnikov and Zeh (2007), 470 bowhead whales (95% CI: 332-665) likely migrated to Chukotka instead of Barrow in spring 2000 and 2001.

Table 1. Summary of abundance estimates for the Western Arctic stock of bowhead whales. The historical estimates were made by back-projecting using a simple recruitment model. All other estimates were developed by corrected ice-based census counts. Historical estimates are from Woodby and Botkin (1993); 1978-2001 estimates are from George et al. (2004) and Zeh and Punt (2005). The 2011 estimate is reported in Givens et al. (2016).

Year	Abundance range or estimate (CV)	Year	Abundance estimate (CV)
Historical	10,400-23,000	1985	5,762 (0.253)
End of commercial whaling	1,000-3,000	1986	8,917 (0.215)
1978	4,765 (0.305)	1987	5,298 (0.327)
1980	3,885 (0.343)	1988	6,928 (0.120)
1981	4,467 (0.273)	1993	8,167 (0.017)
1982	7,395 (0.281)	2001	10,545 (0.128)
1983	6,573 (0.345)	2011	16,820 (0.052)

Bowhead whales were identified from aerial photographs taken in 1985 and 1986, and again in 2003 and 2004, and the results were used in a sight-resight analysis (Table 2). These population estimates and their associated error are comparable to the estimates obtained from the combined ice-based visual and acoustic counts (Raftery and Zeh 1998, Schweder et al. 2009, Koski et al. 2010). An aerial photographic survey was conducted near Point Barrow concurrently with the ice-based spring census in 2011, which, in addition to an abundance estimate based on sight-resight data, also provided a revised survival estimate for the population (Givens et al. 2018) (Table 2). However, because the 2011 ice-based estimate had a lower coefficient of variation (CV), the

Table 2. Summary of abundance estimates for the Western Arctic stock of bowhead whales from aerial sight-resight surveys. Estimates are reported in da Silva et al. 2000, 2007 (1986 estimate), Koski et al. 2010 (2004 estimate), and Givens et al. 2018 (2011 estimate). LB = lower bound of 95% confidence interval.

Year	Abundance range or estimate (CV)	Survival estimate (LB)
1986	4,719 - 7,331	0.985 (0.958)
2004	12,631 (0.2442)	
2011	27,133 (0.217)	0.996 (0.976)

IWC Scientific Committee considered this estimate the most appropriate for management and use in the SLA (IWC 2018). This estimate is more than 8 years old and is outdated for use in stock assessments; however, because this population is increasing, this is still considered a valid minimum population estimate (NMFS 2016).

Minimum Population Estimate

The minimum population estimate (N_{MIN}) for the Western Arctic stock is calculated from Equation 1 from the potential biological removal (PBR) guidelines (NMFS 2016): $N_{MIN} = N/\exp(0.842\times[\ln(1+[CV(N)]^2)]^{½})$. Using the 2011 population estimate (N) from the ice-based survey of 16,820 and its associated CV(N) of 0.052 (Table 1), N_{MIN} for this stock of bowhead whales is 16,100 whales. The 2016 guidelines for preparing Stock Assessment Reports (NMFS 2016) recommend that N_{MIN} be considered unknown if the abundance estimate is more than 8 years old, unless there is compelling evidence that the stock has not declined since the last estimate. Because this population is increasing, this is still considered a valid minimum population estimate.

Current Population Trend

Based on concurrent passive acoustic and ice-based visual surveys, Givens et al. (2013) reported that the Western Arctic stock of bowhead whales increased at a rate of 3.7% (95% CI = 2.9-4.6%) from 1978 to 2011, during which time abundance tripled from approximately 5,000 to approximately 16,820 whales (Givens et al. 2016) (Fig. 2). Schweder et al. (2009) estimated the yearly growth rate to be 3.2% (95% CI = 0.5-4.8%) between 1984 and 2003 using a sight-resight analysis of aerial photographs.

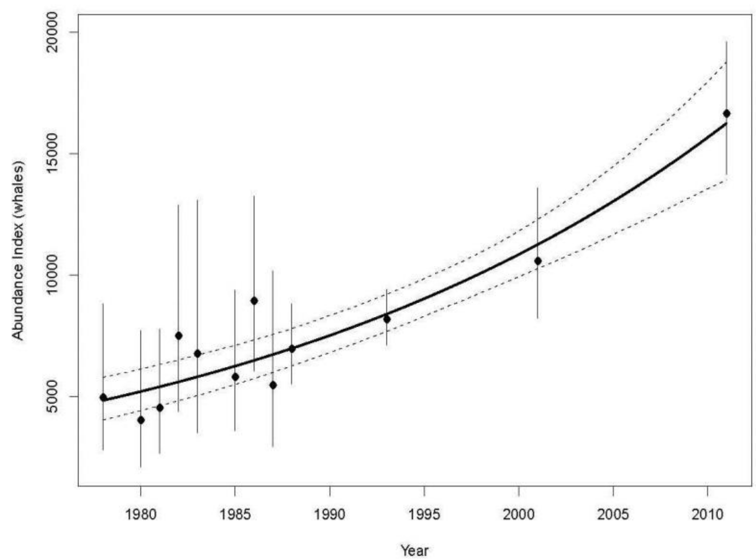

Figure 2. Abundance estimates (points with confidence interval lines) and trend (black line with confidence range) for the Western Arctic stock of bowhead whales, 1978-2011 (Givens et al. 2013), as computed from ice-based counts and acoustic data collected during bowhead whale spring migrations past Point Barrow, Alaska.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

The current estimate for the rate of increase for the Western Arctic stock of bowhead whales (3.7%: 95% CI = 2.9-4.6%) should not be used as an estimate of the maximum net productivity rate (R_{MAX}) because the population is currently being harvested and the population has been estimated to be at a substantial fraction of its carrying capacity (Brandon and Wade 2006); therefore, this stock may not be growing at its maximum rate. Thus, the cetacean maximum theoretical net productivity rate of 4% will be used for the Western Arctic stock of bowhead whales (NMFS 2016).

POTENTIAL BIOLOGICAL REMOVAL

PBR is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: $PBR = N_{MIN} \times 0.5R_{MAX} \times F_R$. The recovery factor ($F_R$) for this stock has been set at 0.5 rather than the default value of 0.1 for endangered species because population levels are increasing in the presence of a known take (NMFS 2016). Thus, PBR is 161 whales (16,100 × 0.02 × 0.5). The calculation of a PBR level for the Western Arctic bowhead whale stock is required by the MMPA even though the subsistence harvest quota is established under the authority of the IWC based on an extensively tested SLA (IWC 2003). The quota is based on subsistence need or the ability of the bowhead whale population to sustain a harvest, whichever is smaller. The IWC bowhead whale quota takes precedence over the PBR estimate for the purpose of managing the Alaska Native subsistence harvest from this stock. From 2013 to 2018, the IWC established a block quota of 336 landed bowhead whales. Because some whales are struck and lost, the IWC set a strike limit of 67 (plus up to 15 previously unused strikes) per year. In recent years, an arrangement between the United States and the Russian Federation ensures that the total quota of bowhead whales struck will not exceed the limits set by the IWC. Under this arrangement, the Chukotka Natives in Russia may use no more than seven strikes, and Alaska Natives may use no more than 75 strikes. The total block quota for 2019 to 2025 is 392 whales, with no more than 67 strikes per year, except that any unused portion of a strike quota from the three prior quota blocks can be carried forward and added to the strike quotas of subsequent years, provided that no more than 50% of the annual strike limit is added to the strike quota for any one year.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals between 2014 and 2018 is listed, by marine mammal stock, in Young et al. (2020); however, only the mortality and serious injury data are included in the Stock Assessment Reports. The minimum estimated mean annual level of human-caused mortality and serious injury for Western Arctic bowhead whales between 2014 and 2018 is 56 whales: 0.2 in U.S. commercial fisheries (Table 3) and 56 in subsistence takes by Natives of Alaska (number landed + struck and lost mortality) and Russia (number landed, struck and lost not reported). Potential threats most likely to result in direct human-caused mortality or serious injury of individuals in this stock include entanglement in fishing gear and ship strikes due to increased vessel traffic (from increased commercial shipping in the Chukchi and Beaufort seas) (Smith and Stephenson 2013).

Fisheries Information

Information for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is available in Appendix 3 of the Alaska Stock Assessment Reports (observer coverage) and in the NMFS List of Fisheries (LOF) and the fact sheets linked to fishery names in the LOF (observer coverage and reported incidental takes of marine mammals: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-list-fisheries, accessed December 2020).

While there are no observer program records of bowhead whale mortality or serious injury incidental to U.S. commercial fisheries in Alaska, Citta et al. (2014) found that the distribution of satellite-tagged bowhead whales in the Bering Sea spatially, but not temporally, overlapped areas where commercial pot fisheries occurred and noted the potential risk of entanglement in lost gear. Approximately 12% of the bowhead whales taken in the subsistence hunt between 1990 and 2012 showed evidence of entanglement in line or net (Philo et al. 1993). George et al. (2017) examined 904 records of bowhead whales harvested between 1990 and 2012. Of these, 514 records were examined for at least one of the three types of scars indicating injuries from line entanglement wounds (514 records examined), attacks by killer whales (377 records examined), or ship strikes (and/or propeller injuries) (504 records examined). Their best estimate of the occurrence of entanglement scars was approximately 12.2% (59/485; 29 records with possible entanglement scars were excluded from the analysis) with the cause most likely from fishing/crab pot gear in the Bering Sea. Most entanglement injuries occurred on the peduncle and were rarely observed on smaller subadult and juvenile whales (<10 m), possibly because young whales are less likely to survive entanglements (George et al. 2017) and have presumably had fewer years during which to acquire entanglement scars. A review of the photo-identification catalogue from 1985 to 2011 found the probability of scarring due to entanglement at about 2.2% per year (95% CI: 1.1-3.3%), with 12.4% of living bowhead whales photographed in 2011 showing evidence of entanglement (George et al. 2019).

One dead bowhead whale was found floating in Kotzebue Sound in early July 2010, entangled in crab pot gear similar to that used by commercial crabbers in the Bering Sea (Suydam et al. 2011), and one entangled bowhead whale was photographed during the 2011 spring aerial photographic survey of bowhead whales near Point Barrow (Mocklin et al. 2012) but it was not considered to be seriously injured. In July 2015, a dead adult female bowhead whale drifting near Saint Lawrence Island in the Bering Strait was entangled in commercial fishing gear (Suydam et al. 2016), which included lines, two floats, and an attached color coded/numbered permit tag for the 2012/2013 winter commercial blue king crab fishery located in Saint Matthew Island waters of the northern Bering Sea (Sheffield and Savoonga Whaling Captains Association 2015) (Table 3). Two of the bowhead whales taken in the Alaska Native subsistence hunt in 2017 were seriously injured due to entanglement in pot gear suspected (but not confirmed) to be from Bering Sea commercial pot fisheries (Young et al. 2020) and a third whale taken during the subsistence hunt on 5 May 2017 was reported as "lethargic" and was later found to have 84 m of 19-mm rope attached to the baleen rack, left pectoral flipper, and peduncle, penetrating up to 10 cm through the epidermis (Rolland et al. 2019); however, because these whales are included in the Alaska Native subsistence harvest for 2017 (Table 4), they are not listed in Table 3. Thus, the minimum estimated average annual mortality and serious injury rate in U.S. commercial fisheries between 2014 and 2018 is 0.2 bowhead whales (Table 3; Young et al. 2020), although, the actual rate is currently unknown. This mortality and serious injury estimate results from an actual count of verified human-caused deaths and serious injuries and is a minimum because not all entangled animals are found, reported, or have the cause of death determined.

Table 3. Summary of mortality and serious injury of Western Arctic bowhead whales, by year and type, reported to the NMFS Alaska Region marine mammal stranding network between 2014 and 2018 (Young et al. 2020).

Cause of injury	2014	2015	2016	2017	2018	Mean annual mortality
Entangled in Bering Sea/Aleutian Is. commercial blue king crab pot gear	0	1	0	0	0	0.2
Total in commercial fisheries						0.2

Alaska Native Subsistence/Harvest Information

NMFS signed an agreement with the Alaska Eskimo Whaling Commission (in 1998, as last amended in 2019) to protect the bowhead whale and the Eskimo culture. This co-management agreement promotes full and equal participation by Alaska Natives in decisions affecting the subsistence management of marine mammals (to the maximum extent allowed by law) as a tool for conserving marine mammal populations in Alaska (https://www.fisheries.noaa.gov/alaska/marine-mammal-protection/co-management-marine-mammals-alaska, accessed December 2020).

Alaska Natives have been taking bowhead whales for subsistence purposes for at least 2,000 years (Marquette and Bockstoce 1980, Stoker and Krupnik 1993). Subsistence takes have been regulated by a quota system under the authority of the IWC since 1977. Alaska Native subsistence hunters, primarily from 11 Alaska communities, take approximately 0.1-0.5% of the Western Arctic bowhead whale stock per annum (Philo et al. 1993, Suydam et al. 2011). Under this quota, the number of bowhead whales landed by Alaska Natives between 1974 and 2018 ranged from 8 to 55 whales per year (Suydam and George 2012; Suydam et al. 2012, 2013, 2014, 2015, 2016, 2017, 2018, 2019; George and Suydam 2014). The maximum number of strikes per year is set by a quota which is determined by subsistence needs and bowhead whale abundance and trend estimates (Stoker and Krupnik 1993). Suydam and George (2012) summarized Alaska subsistence harvests of bowhead whales from 1974 to 2011 and reported a total of 1,149 whales landed by hunters from 12 villages, with Utqiaġvik (formerly Barrow) landing the most whales (n = 590) and Shaktoolik landing only one. Alaska Natives landed 221 bowhead whales between 2014 and 2018 and 52 of the 65 whales that were struck and lost were determined to have died or had a poor chance of survival, resulting in an average annual take of 55 whales (Table 4). Unlike the NMFS process for determining serious injuries (described in NMFS 2012), these estimates of struck and lost mortality are based on the Whaling Captains' assessment of the likelihood of survival (see criteria described in Suydam et al. 1995). The number of whales landed at each village varies greatly from year to year, as success is influenced by village size and ice and weather conditions. The efficiency of the hunt (the percent of whales struck that are retrieved) has increased since the implementation of the bowhead whale quota in 1978. In 1978, the efficiency was about 50%. In 2018, 47 of 68 whales struck were landed, resulting in an efficiency of 69% and the mean efficiency for 2008 to 2017 was 77% (Suydam et al. 2019).

Canadian and Russian Natives also take whales from this stock. No catches of Western Arctic bowhead whales were reported by Canadian hunters between 2014 and 2018; however, two bowhead whales were landed in Russia in 2016 (Ilyashenko and Zharikov 2017), one in 2017 (Zharikov 2018), and none in 2018 (Zharikov et al. 2019), resulting in an average annual take of 0.6 (landed) whales.

The total average annual subsistence take for 2014 to 2018 is 56 bowhead whales, which includes the number landed (plus the struck and lost mortality) by Alaska Natives and the number landed (struck and lost not reported) by Russian Natives.

Table 4. Summary of the Alaska Native subsistence harvest of Western Arctic bowhead whales between 2014 and 2018.

Year	Landed	Struck and lost	Struck and lost mortality	Total (landed + struck and lost mortality)
2014[a]	38	15	12	50
2015[b]	39	10	6	45
2016[c]	47	12	12	59
2017[d]	50	7	5	55
2018[e]	47	21	17	64
Mean annual number taken (landed + struck and lost mortality)				55

[a]Suydam et al. (2015); [b]Suydam et al. (2016); [c]Suydam et al. (2017); [d]Suydam et al. (2018); [e]Suydam et al. (2019).

Other Mortality

Pelagic commercial whaling for bowhead whales was conducted from 1849 to 1914 in the Bering, Chukchi, and Beaufort seas (Bockstoce et al. 2007). During the first two decades of the fishery (1850-1870), over 60% of the estimated pre-whaling population was killed, and effort remained high into the 20th century (Braham 1984). Woodby and Botkin (1993) estimated that the pelagic whaling industry harvested 18,684 whales from this stock. From 1848 to 1919, shore-based whaling operations (including landings as well as struck and lost estimates from the U.S., Canada, and Russia) took an additional 1,527 whales (Woodby and Botkin 1993). An unknown percentage of the whales taken by the shore-based operations were harvested for subsistence purposes. Historical harvest estimates likely underestimate the actual harvest as a result of under-reporting of the Soviet catches (Yablokov 1994) and incomplete reporting of struck and lost whales.

Transient killer whales are known to prey on bowhead whales. In a study of marks on bowhead whales taken in the subsistence harvest between spring 1976 and fall 1992, 4.1% to 7.9% had scars indicating that they had survived attacks by killer whales (George et al. 1994). Of 377 complete records for killer whale scars collected from 1990 to 2012, 29 whales (7.9%) had scarring "rake marks" consistent with killer whale injuries and another 10 had possible injuries (George et al. 2017). A higher rate of killer whale rake mark scars occurred from 2002 to 2012 than in the previous decade. George et al. (2017) noted this may be due to better reporting and/or sampling bias, an increase in killer whale population size, an increase in occurrence of killer whales at high latitudes (Clarke et al. 2013), or a longer open water period offering more opportunities to attack bowhead whales. The Aerial Surveys of Arctic Marine Mammals (ASAMM) project photo-documented bowhead whale carcasses that had injuries consistent with killer whale predation in 2012 (two carcasses), 2013 (two), 2015 (two), 2016 (three), and 2017 (one) and three of these carcasses (one each in 2013, 2015, and 2017) were likely calves or yearlings (Willoughby et al. 2018).

With increasing ship traffic and oil and gas exploration and development activities in the Chukchi and Beaufort seas, ship strikes may pose a greater risk to bowhead whales. Currently, ship-strike injuries on bowhead whales in Alaska are thought to be uncommon (George et al. 2017, 2019). Only 10 whales harvested between 1990 and 2012 (approximately 2% of the records examined) showed clear evidence of scarring from ship propellers, while only seven whales from the photo-identification catalogue from 1985 to 2011 (1% of the sample) had evidence of ship-inflicted scars.

STATUS OF STOCK

Based on currently available data, the minimum estimated mean annual mortality and serious injury rate incidental to U.S. commercial fisheries (0.2 whales) is not known to exceed 10% of the PBR (10% of PBR = 16) and, therefore, can be considered insignificant and approaching a zero mortality and serious injury rate. The minimum estimated mean annual level of human-caused mortality and serious injury (56 whales) is not known to exceed the PBR (161) nor the IWC annual maximum strike limit (67 + up to 15 previously unused strikes). The Western Arctic bowhead whale stock has been increasing; the estimate of 16,820 whales from 2011 is between 31% and 168% of the pre-exploitation abundance of 10,000 to 55,000 whales estimated by Brandon and Wade (2004, 2006). However, the stock is classified as a strategic stock because the bowhead whale is listed as endangered under the U.S. Endangered Species Act and is, therefore, also designated as depleted under the MMPA.

There are key uncertainties in the assessment of the Western Arctic stock of bowhead whales. The current abundance estimate is calculated using data from 2011; however, the N_{MIN} is still considered a valid minimum population estimate because the population is increasing (NMFS 2016). Although there are few records of bowhead whales being killed or seriously injured incidental to commercial fishing, about 12.2% of harvested bowhead whales examined for scarring (59/485 records) had scars indicating line entanglement wounds (George et al. 2017) and the southern range of the population overlaps with commercial pot fisheries (Citta et al. 2014). The stock may be particularly sensitive to anthropogenic sound; under some circumstances, the stock changes either distribution or calling behavior in response to levels of anthropogenic sounds that are slightly above ambient (Blackwell et al. 2015). The reduction in sea ice may lead to increased predation of bowhead whales by killer whales.

HABITAT CONCERNS

Vessel traffic in arctic waters is increasing, largely due to an increase in commercial shipping facilitated by the lack of sea ice. This increase in vessel traffic could result in an increased number of vessel collisions with bowhead whales (Huntington et al. 2015). Oil and gas development in the Arctic imposes risks of various forms of pollution, including oil spills, in bowhead whale habitat, and the technology for effectively recovering spilled oil in icy conditions is lacking (Wilkinson et al. 2017).

Also of concern is noise produced by seismic surveys and vessel traffic resulting from shipping and offshore energy exploration, development, and production operations. Evidence indicates that bowhead whales are sensitive to noise from offshore drilling platforms and seismic survey operations (Richardson and Malme 1993, Richardson 1995, Davies 1997, Robertson et al. 2013, Blackwell et al. 2017). Bowhead whales often avoid sound sources associated with active drilling (Schick and Urban 2000) and seismic operations (Miller et al. 1999). Exposure to seismic operations resulted in subtle changes to dive, surfacing, and respiration behaviors (Robertson et al. 2013). Source levels, time of year, and whale behavior (migrating, feeding, etc.) all affect the extent of displacement or changes in behavior (e.g., Richardson et al. 1986, 1999; Ljungblad et al. 1988; Miller et al. 2005; Harris et al. 2007; MMS 2008; Funk et al. 2010) and impacts on bowhead calling rates (Greene et al. 1998; Blackwell et al. 2013, 2015).

Global climate model projections for the next 50 to 100 years consistently show pronounced warming over the Arctic, accelerated sea-ice loss, and continued permafrost degradation (USGS 2011, IPCC 2013, Jeffries et al. 2015). Within the Arctic, some of the largest changes are projected to occur in the Bering, Beaufort, and Chukchi seas (Chapman and Walsh 2007, Walsh 2008). Ice-associated animals, including the bowhead whale, may be sensitive to changes in arctic weather, sea surface temperatures, sea-ice extent, and the concomitant effect on prey availability. Based on an analysis of various life-history features, Laidre et al. (2008) concluded that, on a worldwide basis, bowhead whales were likely to be moderately sensitive to climate change. Using statistical models, Chambault et al. (2018) found that bowhead whales in Baffin Bay, Greenland, targeted a narrow range of temperatures (-0.5 to 2°C) and may be exposed to thermal stress as a result of warming temperatures. However, thermal stress resulting from increased sea surface temperatures has not been observed in the Western Arctic stock of bowhead whales. On the contrary, landed Western Arctic bowhead whales had better body condition during years of light ice cover (George et al. 2006). In addition, a positive correlation between body condition of Western Arctic bowhead whales and summer sea-ice loss has been observed over the last 2.5 decades in the Pacific Arctic (George et al. 2015). Ice-free areas along the shelf break are thought to create increased upwelling and likely more feeding opportunities for foraging whales. The movement and foraging behavior of bowhead whales is becoming more variable as feeding areas are altered in response to retreating sea ice. Additionally, Hannay et al. (2013) found that a large fraction of bowhead whale acoustic detections in the northeast Chukchi Sea occurred just in advance of the progression of sea ice formation during the fall migration, suggesting that an increase in ice-free days may lead to a delayed migration out of the Chukchi Sea during fall. Sheffield and George (2013) presented evidence that the occurrence of fish has become more prevalent in the diets of Western Arctic bowhead whales near Utqiaġvik in the autumn. However, there are insufficient data to make reliable projections about whether arctic climate change will result in negative (thermal stress, habitat loss) or positive (prey abundance) effects on this population.

Ocean acidification, driven primarily by the production of carbon dioxide (CO_2) emissions into the atmosphere, is also a concern due to potential effects on prey. Because their primary prey are small crustaceans (especially calanoid copepods, euphausiids, gammarid and hyperid amphipods, and mysids that have exoskeletons composed of chitin and calcium carbonate), bowhead whale survival and recruitment may be impacted by increased ocean acidification (Lowry et al. 2004). The nature and timing of impacts to bowhead whales from ocean acidification are extremely uncertain and will depend partially on the whales' ability to switch to alternate prey species. Ecosystem responses may have very long lags as they propagate through trophic webs.

CITATIONS

Ashjian, C. J., S. R. Braund, R. G. Campbell, J. C. George, J. Kruse, W. Maslowski, S. E. Moore, C. R. Nicolson, S. R. Okkonen, B. F. Sherr, E. B. Sherr, and Y. H. Spitz. 2010. Climate variability, oceanography, bowhead whale distribution, and Iñupiat subsistence whaling near Barrow, Alaska. Arctic 63(2):179-194.

Bachmann, L., Ø. Wiig, M. P. Heide-Jørgensen, K. L. Laidre, L. D. Postma, L. Dueck, and P. J. Palsbøl. 2010. Genetic diversity in Eastern Canadian and Western Greenland bowhead whales (*Balaena mysticetus*). Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/62/BRG26). 6 p.

Blackwell, S. B., C. S. Nations, T. L. McDonald, C. R. Greene, A. M. Thode, M. Guerra, and A. M. Macrander. 2013. Effects of airgun sounds on bowhead whale calling rates in the Alaskan Beaufort Sea. Mar. Mammal Sci. 29(4):E342-E365. DOI: dx.doi.org/10.1111/mms.12001 .

Blackwell, S. B., C. S. Nations, T. L. McDonald, A. M. Thode, D. Mathias, K. H. Kim, C. R. Greene, Jr., and A. M. Macrander. 2015. Effects of airgun sounds on bowhead whale calling rates: evidence for two behavioral thresholds. PLoS ONE 10(6):e0125720. DOI: dx.doi.org/10.1371/journal.pone.0125720 .

Blackwell, S. B., C. S. Nations, A. M. Thode, M. E. Kauffman, A. S. Conrad, R. G. Norman, and K. H. Kim. 2017. Effects of tones associated with drilling activities on bowhead whales calling rates. PLoS ONE 12(11):e0188459. DOI: dx.doi.org/10.1371/journal.pone.0188459 .

Bockstoce, J. R., and J. J. Burns. 1993. Commercial whaling in the North Pacific sector, p. 563-577. *In* J. J. Burns, J. J. Montague, and C. J. Cowles (eds.), The Bowhead Whale. Soc. Mar. Mammal., Spec. Publ. No. 2.

Bockstoce, J. R., D. B. Botkin, A. Philp, B. W. Collins, and J. C. George. 2007. The geographic distribution of bowhead whales (*Balaena mysticetus*) in the Bering, Chukchi, and Beaufort seas: evidence from whaleship records, 1849-1914. Mar. Fish. Rev. 67(3):1-43.

Boertmann, D., L. A. Kyhn, L. Witting, and M. P. Heide-Jorgensen. 2015. A hidden getaway for bowhead whales in the Greenland Sea. Polar Biol. 38(8):1315-1319. DOI: dx.doi.org/10.1007/s00300-015-1695-y .

Braham, H. W. 1984. The bowhead whale, *Balaena mysticetus*. Mar. Fish. Rev. 46(4):45-53.

Braham, H. W., M. A. Fraker, and B. D. Krogman. 1980. Spring migration of the Western Arctic population of bowhead whales. Mar. Fish. Rev. 42(9-10):36-46.

Brandon, J., and P. R. Wade. 2004. Assessment of the Bering-Chukchi-Beaufort Seas stock of bowhead whales. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/56/BRG20). 32 p.

Brandon, J., and P. R. Wade. 2006. Assessment of the Bering-Chukchi-Beaufort Seas stock of bowhead whales using Bayesian model averaging. J. Cetacean Res. Manage. 8(3):225-239.

Burns, J. J., J. J. Montague, and C. J. Cowles (eds.). 1993. The Bowhead Whale. Soc. Mar. Mammal., Spec. Publ. No. 2. 787 p.

Chambault, P., C. M. Albertsen, T. A. Patterson, R. G. Hansen, O. Tervo, K. L. Laidre, and M. P. Heide-Jørgensen. 2018. Sea surface temperature predicts the movements of an Arctic cetacean: the bowhead whale. Scientific Reports 8(1):9658. DOI: dx.doi.org/10.1038/s41598-018-27966-1 .

Chapman, W. L., and J. E. Walsh. 2007. Simulations of arctic temperature and pressure by global coupled models. J. Climate 20:609-632.

Citta, J. J., L. T. Quakenbush, J. C. George, R. J. Small, M P. Heide-Jørgensen, H. Brower, B. Adams, and L. Brower. 2012. Winter movements of bowhead whales (*Balaena mysticetus*) in the Bering Sea. Arctic 65(1):13-34.

Citta, J. J., J. J. Burns, L. T. Quakenbush, V. Vanek, J. C. George, R. J. Small, M. P. Heide-Jørgensen, and H. Brower. 2014. Potential for bowhead whale entanglement in cod and crab pot gear in the Bering Sea. Mar. Mammal Sci. 30(2):445-459. DOI: dx.doi.org/10.1111/mms.12047 .

Citta, J. J., L. T. Quakenbush, S. R. Okkonen, M. L. Druckenmiller, W. Maslowski, J. Clement-Kinney, J. C. George, H. Brower, R. J. Small, C. J. Ashjian, L. A. Harwood, and M. P. Heide-Jørgensen. 2015. Ecological characteristics of core-use areas used by Bering-Chukchi-Beaufort (BCB) bowhead whales, 2006-2012. Prog. Oceanogr. 136:201-222. DOI: dx.doi.org/10.1016/j.pocean.2014.08.012 .

Citta, J. J., S. R. Okkonen, L. T. Quakenbush, W. Maslowski, R. Osinski, J. C. George, R. J. Small, H. Brower, Jr., M. P. Heide-Jørgensen, and L. A. Harwood. 2018. Oceanographic characteristics associated with autumn movements of bowhead whales in the Chukchi Sea. Deep Sea Research II 152:121-131. DOI: dx.doi.org/10.1016/j.dsr2.2017.03.009 .

Clark, C. W., S. Mitchell, and R. Charif. 1994. Distribution and behavior of the bowhead whale, *Balaena mysticetus*, based on preliminary analysis of acoustic data collected during the 1993 spring migration off Point Barrow, Alaska. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/46/AS19). 24 p.

Clarke, J. T., and M. C. Ferguson. 2010. Aerial surveys for bowhead whales in the Alaskan Beaufort Sea: BWASP update 2000-2009 with comparisons to historical data. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/62/BRG14). 11 p.

Clarke, J., K. Stafford, S. E. Moore, B. Rone, L. Aerts, and J. Crance. 2013. Subarctic cetaceans in the southern Chukchi Sea: evidence of recovery or response to a changing ecosystem. Oceanography 26(4):136-149. DOI: dx.doi.org/10.5670/oceanog.2013.81 .

Clarke, J. T., M. C. Ferguson, C. Curtice, and J. Harrison. 2015. Biologically important areas for cetaceans within US waters - Arctic region. Aquat. Mamm. (Special Issue) 41(1):94-103.

Clarke, J. T., A. S. Kennedy, and M. C. Ferguson. 2016. Bowhead and gray whale distributions, sighting rates, and habitat associations in the eastern Chukchi Sea, summer and fall 2009-15, with a retrospective comparison to 1982-91. Arctic 69(4):359-377.

Clarke, J. T., A. A. Brower, M. C. Ferguson, and A. L. Willoughby. 2017. Distribution and relative abundance of marine mammals in the eastern Chukchi and western Beaufort seas, 2016. Annual Report, OCS Study BOEM 2017-078. Marine Mammal Laboratory, AFSC, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115.

Clarke, J. T., M. C. Ferguson, A. A. Brower, and A. L. Willoughby. 2018a. Bowhead whale calves in the western Beaufort Sea, 2012-2017. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/67b/AWMP3). 11 p.

Clarke, J. T., M. C. Ferguson, A. L. Willoughby, and A. A. Brower. 2018b. Bowhead and beluga whale distributions, sighting rates, and habitat associations in the western Beaufort Sea in summer and fall 2009-16, with comparison to 1982-91. Arctic 71(2):115-138.

Cosens, S. E., H. Cleator, and P. Richard. 2006. Numbers of bowhead whales (*Balaena mysticetus*) in the eastern Canadian Arctic, based on aerial surveys in August 2002, 2003 and 2004. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/58/BRG7). 19 p.

da Silva, C. Q., J. Zeh, D. Madigan, J. Laake, D. Rugh, L. Baraff, W. Koski, and G. Miller. 2000. Capture-recapture estimation of bowhead whale population size using photo-identification data. J. Cetacean Res. Manage. 2(1):45-61.

da Silva, C. Q., P. V. S. Gomes, and M. A. Stradioto. 2007. Bayesian estimation of survival and capture probabilities using logit link and photoidentification data. Comput. Stat. Data Anal. 51:6521-6534.

Davies, J. R. 1997. The impact of an offshore drilling platform on the fall migration path of bowhead whales: a GIS-based assessment. Unpubl. MS Thesis, Western Washington University, Bellingham, WA. 51 p.

Doniol-Valcroze, T., J. -F. Gosselin, D. Pike, J. Lawson, N. Asselin, K. Hedges, and S. Ferguson. 2015. Abundance estimate of the Eastern Canada – West Greenland bowhead whale population based on the 2013 High Arctic Cetacean Survey. DFO Can. Sci. Advis. Sec. Res. Doc. 2015/058. v + 27 p.

Dueck, L. P., M. P. Heide-Jørgensen, M. V. Jensen, and L. D. Postma. 2006. Update on investigations of bowhead whale (*Balaena mysticetus*) movements in the eastern Arctic, 2003-2005, based on satellite-linked telemetry. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/58/BRG5). 17 p.

Druckenmiller, M. L., J. J. Citta, M. C. Ferguson, J. T. Clarke, J. C. George, and L. Quakenbush. 2018. Trends in sea-ice cover within bowhead whale habitats in the Pacific Arctic. Deep Sea Research II 152:95-107. DOI: dx.doi.org/10.1016/j.dsr2.2017.10.017 .

Fish, F. E., K. T. Goetz, D. J. Rugh, and L. Vate Brattström. 2013. Hydrodynamic patterns associated with echelon formation swimming by feeding bowhead whales (*Balaena mysticetus*). Mar. Mammal Sci. 29(4):E498-E507. DOI: dx.doi.org/10.1111/mms.12004 .

Frasier, T. R., S. D. Petersen, L. Postma, L. Johnson, M P. Heide-Jørgensen, and S. H. Ferguson. 2015. Abundance estimates of the Eastern Canada-West Greenland bowhead whale (*Balaena mysticetus*) population based on genetic capture-mark-recapture analyses. DFO Can. Sci. Advis. Sec. Res. Doc. 2015/008. iv + 21 p.

Funk, D. W., R. Rodrigues, D. S. Ireland, and W. R. Koski. 2010. Summary and assessment of potential effects on marine mammals, Chapter 11. *In* D. W. Funk, D. S. Ireland, R. Rodrigues, and W. R. Koski (eds.), Joint monitoring program in the Chukchi and Beaufort seas, open water seasons, 2006–2008. LGL Alaska Report P1050-2, Report from LGL Alaska Research Associates, Inc., LGL, Ltd., Greeneridge Sciences, Inc., and JASCO Research, Ltd., for Shell Offshore, Inc., and Other Industry Contributors, and National Marine Fisheries Service, U.S. Fish and Wildlife Service. 506 p. + appendices.

George, J. C., and R. S. Suydam. 2014. Update on characteristics of bowhead whale (*Balaena mysticetus*) calves. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/65b/BRG20Rev). 7 p.

George, J. C., L. Philo, K. Hazard, D. Withrow, G. Carroll, and R. Suydam. 1994. Frequency of killer whale (*Orcinus orca*) attacks and ship collisions based on scarring on bowhead whales (*Balaena mysticetus*) of the Bering-Chukchi-Beaufort Seas stock. Arctic 47(3):247-55.

George, J. C., J. Zeh, R. Suydam, and C. Clark. 2004. Abundance and population trend (1978-2001) of Western Arctic bowhead whales surveyed near Barrow, Alaska. Mar. Mammal Sci. 20:755-773.

George, J. C., C. Nicolson, S. Drobot, J. Maslanik, and R. Suydam. 2006. Sea ice density and bowhead whale body condition preliminary findings. Poster presented to the Society for Marine Mammalogy, San Diego, CA.

George, J. C., M. L. Druckenmiller, K. L. Laidre, R. Suydam, and B. Person. 2015. Bowhead whale body condition and links to summer sea ice and upwelling in the Beaufort Sea. Prog. Oceanogr. 136:250-262.

George, J. C., G. Sheffield, D. J. Reed, B. Tudor, R. Stimmelmayr, B. T. Person, T. Sformo, and R. Suydam. 2017. Frequency of injuries from line entanglements, killer whales, and ship strikes on Bering-Chukchi-Beaufort Sea bowhead whales. Arctic 70(1):37-46. DOI: dx.doi.org/10.14430/arctic4631 .

George, J. C., B. Tudor, G. H. Givens, J. Mocklin, and L. Vate Brattström. 2019. Entanglement-scar acquisition rates and scar frequency for Bering-Chukchi-Beaufort Seas bowhead whales using aerial photography. Mar. Mammal Sci. 35(4):1304-1321. DOI: dx.doi.org/10.1111/mms.12597 .

Givens, G. H., S. L. Edmondson, J. C. George, R. Suydam, R. A. Charif, A. Rahaman, D. Hawthorne, B. Tudor, R. A. DeLong, and C. W. Clark. 2013. Estimate of 2011 abundance of the Bering-Chukchi-Beaufort Seas bowhead whale population. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/65a/BRG1). 30 p.

Givens, G. H., S. L. Edmondson, J. C. George, R. Suydam, R. A. Charif, A. Rahaman, D. Hawthorne, B. Tudor, R. A. DeLong, and C. W. Clark. 2016. Horvitz-Thompson whale abundance estimation adjusting for uncertain recapture, temporal availability variation, and intermittent effort. Environmetrics 27:134-146. DOI: dx.doi.org/10.1002/env.2379 .

Givens, G. H., J. A. Mocklin, L. Vate Brattström, B. J. Tudor, W. R. Koski, J. E. Zeh, R. Suydam, and J. C. George. 2018. Survival rate and 2011 abundance of Bering-Chukchi-Beaufort Seas bowhead whales from photo-identification data over three decades. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/67b/AWMP/01 Rev1). 24 p.

Greene, C. R., Jr., W. J. Richardson, and N. S. Altman. 1998. Bowhead whale call detection rates versus distance from airguns operating in the Alaskan Beaufort Sea during fall migration, 1996. J. Acoust. Soc. Am. 104:1826. DOI: dx.doi.org/10.1121/1.423473 .

Hannay, D. E., J. Delarue, X. Mouy, B. S. Martin, D. Leary, J. N. Oswald, and J. Vallarta. 2013. Marine mammal acoustic detections in the northeastern Chukchi Sea, September 2007–July 2011. Continental Shelf Research 67:127-46. DOI: dx.doi.org/10.1016/j.csr.2013.07.009 .

Harris, R. E., T. Elliot, and R. A. Davis. 2007. Results of mitigation and monitoring program, Beaufort Span 2-D marine seismic program, open water season 2006. LGL, Ltd., LGL Report TA4319-1, Report from LGL, Ltd., King City, Ont., for GX Technol., Houston, TX. 48 p.

Harwood, L. A., L. T. Quakenbush, R. J. Small, C. J. George, J. Pokiak, C. Pokiak, M. P. Heide-Jørgensen, E. V. Lea, and H. Brower. 2017. Movements and inferred foraging by bowhead whales in the Canadian Beaufort Sea during August and September, 2006–12. Arctic 70(2):161-76. DOI: dx.doi.org/10.14430/arctic4648 .

Heide-Jørgensen, M. P., K. L. Laidre, M. V. Jensen, L. Dueck, and L. D. Postma. 2006. Dissolving stock discreteness with satellite tracking: bowhead whales in Baffin Bay. Mar. Mammal Sci. 22:34-45.

Heide-Jørgensen, M. P., K. L. Laidre, Ø. Wiig, and L. Dueck. 2010. Large scale sexual segregation of bowhead whales. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/62/BRG23). 13 p.

Huntington, H. P., R. Daniel, A. Hartsig, K. Harun, M. Heiman, R. Meehan, G. Noongwook, L. Pearson, M. Prior-Parks, M. Robards, and G. Stetson. 2015. Vessels, risks, and rules: planning for safe shipping in Bering Strait. Marine Policy 51:119-127.

Ilyashenko, V., and K. Zharikov. 2017. Aboriginal subsistence whaling in the Russian federation in 2016. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/67a/AWMP03). 2 p.

Intergovernmental Panel on Climate Change (IPCC). 2013. Summary for policymakers. *In* T. Stocker, D. Qin, G. Plattner, M. Tignor, S. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P. Midgley (eds.), Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the IPCC. Cambridge University Press, Cambridge, UK, and New York, NY. Available online: https://www.ipcc.ch/site/assets/uploads/2018/02/WG1AR5_SPM_FINAL.pdf . Accessed December 2020.

International Whaling Commission (IWC). 2003. Report of the fourth workshop on the development of an Aboriginal Subsistence Whaling Management Procedure (AWMP). J. Cetacean Res. Manage. 5(Suppl.):489-497.

International Whaling Commission (IWC). 2008. Annex F: Report of the sub-committee on bowhead, right and gray whales. J. Cetacean Res. Manage. 10(Suppl.):150-166.

International Whaling Commission (IWC). 2010. Annex F: Report of the sub-committee on bowhead, right and gray whales. J. Cetacean Res. Manage. 11(Suppl. 2):154-179.

International Whaling Commission (IWC). 2011. Annex F: Report of the sub-committee on bowhead, right and gray whales. J. Cetacean Res. Manage. 12(Suppl.):168-184.

International Whaling Commission (IWC). 2018. Report of the Scientific Committee. Unpubl. doc. IWC/67/Rep01. Available online: www.iwc.int . Accessed December 2020.

Jeffries, M. O., J. Richter-Menge, and J. E. Overland (eds.). 2015. Arctic report card 2015. Available online: http://www.arctic.noaa.gov/reportcard . Accessed December 2020.

Koski, W., J. Zeh, J. Mocklin, A. R. Davis, D. J. Rugh, J. C. George, and R. Suydam. 2010. Abundance of Bering-Chukchi-Beaufort bowhead whales (*Balaena mysticetus*) in 2004 estimated from photo-identification data. J. Cetacean Res. Manage. 11(2):89-99.

Krogman, B., D. Rugh, R. Sonntag, J. Zeh, and D. Ko. 1989. Ice-based census of bowhead whales migrating past Point Barrow, Alaska, 1978-1983. Mar. Mammal Sci. 5:116-138.

Laidre, K. L., I. Stirling, L. Lowry, Ø. Wiig, M. P. Heide-Jørgensen, and S. Ferguson. 2008. Quantifying the sensitivity of arctic marine mammals to climate-induced habitat change. Ecol. Appl. 18(2)Suppl.:S97-S125.

Ljungblad, D. K., B. Würsig, S. L. Swartz, and J. M. Keene. 1988. Observations on the behavioral responses of bowhead whales (*Balaena mysticetus*) to active geophysical vessels in the Alaskan Beaufort Sea. Arctic 41(3):183-194.

Lowry, L. F., G. Sheffield, and J. C. George. 2004. Bowhead whale feeding in the Alaskan Beaufort Sea, based on stomach contents analyses. J. Cetacean Res. Manage. 6(3):215-223.

Marquette, W. M., and J. R. Bockstoce. 1980. Historical shore-based catch of bowhead whales in the Bering, Chukchi, and Beaufort seas. Mar. Fish. Rev. 42(9-10):5-19.

Melnikov, V. V., and J. E. Zeh. 2007. Chukotka Peninsula counts and estimates of the number of migrating bowhead whales (*Balaena mysticetus*). J. Cetacean Res. Manage. 9(1):29-35.

Miller, G. W., R. E. Elliott, W. R. Koski, V. D. Moulton, and W. J. Richardson. 1999. Whales, p. 5-1 to 5-109. *In* W. J. Richardson (ed.), Marine mammal and acoustical monitoring of Western Geophysical's open-water seismic program in the Alaskan Beaufort Sea, 1998. LGL Report TA2230–3. Report from LGL Ltd., King City, Ont., and Greeneridge Sciences Inc., Santa Barbara, CA, for Western Geophysical, Houston, TX, and National Marine Fisheries Service, Anchorage, AK, and Silver Spring, MD. 390 p.

Miller, G. W., V. D. Moulton, R. A. Davis, M. Holst, P. Millman, A. MacGillivray, and D. Hannay. 2005. Monitoring seismic effects on marine mammals—southeastern Beaufort Sea, 2001-2002, p. 511-542. *In* S. L. Armsworthy, P. J. Cranford, and K. Lee (eds.), Offshore Oil and Gas Development Effects Monitoring/Approaches and Technologies. Battelle Press, Columbus, OH.

Minerals Management Service (MMS). 2008. Beaufort Sea and Chukchi Sea Planning Areas, Oil and Gas Lease Sales 209, 212, 217, and 221, Draft Environmental Impact Statement: U.S. Department of the Interior, Minerals Management Service, Alaska OCS Region, MMS 2008-0055, November 2008. Available online: https://www.boem.gov/oil-gas-energy/leasing/beaufort-and-chukchi-sea-planning-areas-oil-and-gas-lease-sales-209-212-217 . Accessed December 2020.

Mocklin, J., J. C. George, M. Ferguson, L. Vate Brattström, V. Beaver, B. Rone, C. Christman, A. Brower, B. Shea, and C. Accardo. 2012. Aerial photography of bowhead whales near Barrow, Alaska, during the 2011 spring migration. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/64/BRG3). 9 p.

Moore, S. E., and R. R. Reeves. 1993. Distribution and movement, p. 313-386. *In* J. J. Burns, J. J. Montague, and C. J. Cowles (eds.), The Bowhead Whale. Soc. Mar. Mammal., Spec. Publ. No. 2.

National Marine Fisheries Service (NMFS). 2012. Process for distinguishing serious from non-serious injury of marine mammals. 42 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-protection-act-policies-guidance-and-regulations . Accessed December 2020.

National Marine Fisheries Service (NMFS). 2016. Guidelines for preparing stock assessment reports pursuant to the 1994 amendments to the Marine Mammal Protection Act. 23 p. Available online: https://www.fisheries.noaa.gov/national/marine-mammal-protection/guidelines-assessing-marine-mammal-stocks . Accessed December 2020.

Okkonen, S. R., C. J. Ashjian, R. G. Campbell, J. Clarke, S. E. Moore, and K. D. Taylor. 2011. Satellite observations of circulation features associated with the Barrow area bowhead whale feeding hotspot. Remote Sens. Environ. 115:2168-2174.

Okkonen, S. R., J. T. Clarke, and R. A. Potter. 2018. Relationships among high river discharges, upwelling events, and bowhead whale (*Balaena mysticetus*) occurrence in the central Alaskan Beaufort Sea. Deep Sea Research II 152:195-202. DOI: dx.doi.org/10.1016/j.dsr2.2016.11.015 .

Philo, L. M., E. B. Shotts, and J. C. George. 1993. Morbidity and mortality, p. 275-312. *In* J. J. Burns, J. J. Montague, and C. J. Cowles (eds.), The Bowhead Whale. Soc. Mar. Mammal., Spec. Publ. No. 2.

Pomerleau, C., C. J. Matthews, C. Gobeil, G. A. Stern, S. H. Ferguson, and R. W. Macdonald. 2018. Mercury and stable isotope cycles in baleen plates are consistent with year-round feeding in two bowhead whale (*Balaena mysticetus*) populations. Polar Biol. 41(9):1881-93. DOI: dx.doi.org/10.1007/s00300-018-2329-y .

Postma, L. D., L. P. Dueck, M. P. Heide-Jørgensen, and S. E. Cosens. 2006. Molecular genetic support of a single population of bowhead whales (*Balaena mysticetus*) in Eastern Canadian Arctic and Western Greenland waters. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/58/BRG4). 15 p.

Quakenbush, L. T., and J. J. Citta. 2019. Satellite tracking of bowhead whales: habitat use, passive acoustics and environmental monitoring. Final Report. U.S. Department of the Interior, Bureau of Ocean Energy Management, Alaska Outer Continental Shelf Region, Anchorage, AK. OCS Study BOEM 2019-076. 60 p. + appendices. Available online: https://www.boem.gov/sites/default/files/documents/regions/alaska-ocs-region/environment/BOEM%202019-076.pdf . Accessed December 2020.

Quakenbush, L. T., R. J. Small, and J. J. Citta. 2010a. Satellite tracking of Western Arctic bowhead whales. Unpubl. report submitted to the Bureau of Ocean Energy Management, Regulation and Enforcement (BOEMRE 2010-033).

Quakenbush, L. T., J. J. Citta, J. C. George, R. J. Small, and M. P. Heide-Jørgensen. 2010b. Fall and winter movements of bowhead whales (*Balaena mysticetus*) in the Chukchi Sea and within a potential petroleum development area. Arctic 63(3):289-307.

Quakenbush, L., J. Citta, J. C. George, M. P. Heide-Jørgensen, H. Brower, L. Harwood, B. Adams, C. Pokiak, J. Pokiak, and E. Lea. 2018. Bering-Chukchi-Beaufort stock of bowhead whales: 2006-2017 satellite telemetry results with some observations on stock sub-structure. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/67B/AWMP/04). 25 p.

Raftery, A., and J. Zeh. 1998. Estimating bowhead whale population size and rate of increase from the 1993 census. J. Am. Stat. Assoc. 93:451-463.

Richardson, W. J. 1995. Documented disturbance reactions, p. 241-324. *In* W. J. Richardson, C. R. Greene, C. I. Malme, and D. H. Thomson (eds.), Marine Mammals and Noise. Academic Press, San Diego, CA.

Richardson, W. J., and C. I. Malme. 1993. Man-made noise and behavioral responses, p. 631-700. *In* J. J. Burns, J. J. Montague, and C. J. Cowles (eds.), The Bowhead Whale. Soc. Mar. Mammal., Spec. Publ. No. 2.

Richardson, W. J., B. Würsig, and C. R. Greene, Jr. 1986. Reactions of bowhead whales, *Balaena mysticetus*, to seismic exploration in the Canadian Beaufort Sea. J. Acoust. Soc. Am. 79(4):1117-1128.

Richardson, W. J., R. A. Davis, C. R. Evans, D. K. Ljungblad, and P. Norton. 1987. Summer distribution of bowhead whales, *Balaena mysticetus*, relative to oil industry activities in the Canadian Beaufort Sea, 1980-84. Arctic 40(2):93-104.

Richardson, W. J., G. W. Miller, and C. R. Greene, Jr. 1999. Displacement of migrating bowhead whales by sounds from seismic surveys in shallow waters of the Beaufort Sea. J. Acoust. Soc. Am. 106(4, Pt. 2):2281.

Robertson, F. C., W. R. Koski, T. A. Thomas, W. J. Richardson, B. Würsig, and A. W. Trites. 2013. Seismic operations have variable effects on dive-cycle behavior of bowhead whales in the Beaufort Sea. Endang. Species Res. 21:143-160. DOI: dx.doi.org/10.3354/esr00515 .

Rolland, R. M., K. M. Graham, R. Stimmelmayr, R. S. Suydam, and J. C. George. 2019. Chronic stress from fishing gear entanglement is recorded in baleen from a bowhead whale (*Balaena mysticetus*). Mar. Mammal Sci. 35(4):1625-1642. DOI: dx.doi.org/10.1111/mms.12596 .

Ross, W. G. 1993. Commercial whaling in the North Atlantic sector, p. 511-561. *In* J. J. Burns, J. J. Montague, and C. J. Cowles (eds.), The Bowhead Whale. Soc. Mar. Mammal., Spec. Publ. No. 2.

Rugh, D., D. DeMaster, A. Rooney, J. Breiwick, K. Shelden, and S. Moore. 2003. A review of bowhead whale (*Balaena mysticetus*) stock identity. J. Cetacean Res. Manage. 5(3):267-279.

Schick, R. S., and D. L. Urban. 2000. Spatial components of bowhead whale (*Balaena mysticetus*) distribution in the Alaskan Beaufort Sea. Can. J. Fish. Aquat. Sci. 57:2193-2200.

Schweder, T., D. Sadykova, D. Rugh, and W. Koski. 2009. Population estimates from aerial photographic surveys of naturally and variably marked bowhead whales. J. Agric. Biol. Environ. Stat. 15(1):1-19.

Sheffield, G., and J. C. George. 2013. Section V – North Slope Borough research: B - diet studies, p. 253-277. *In* K. E. W. Shelden and J. A. Mocklin (eds.), Bowhead Whale Feeding Ecology Study (BOWFEST) in the western Beaufort Sea. Final Report, OCS Study BOEM 2013-0114. Marine Mammal Laboratory, AFSC, NMFS, 7600 Sand Point Way NE, Seattle, WA 98115. Available online: https://www.boem.gov/sites/default/files/uploadedFiles/BOEM/BOEM_Newsroom/Library/Publications/BOEM_2013-0114_BOWFEST_Final_Report.pdf . Accessed December 2020.

Sheffield, G., and Savoonga Whaling Captains Association. 2015. Bowhead whale entangled in commercial crab pot gear recovered near Saint Lawrence Island, Bering Strait. UAF Alaska Sea Grant, Marine Advisory Program, Report to the North Slope Borough Department of Wildlife Management, Barrow, AK. 8 p.

Shelden, K. E. W., and D. J. Rugh. 1995. The bowhead whale (*Balaena mysticetus*): status review. Mar. Fish. Rev. 57(3-4):1-20.

Shpak, O. V., I. G. Meschersky, A. N. Chichkina, D. M. Kuznetsova, A. Y. Paramonov, and V. V. Rozhnov. 2014. New data on the Okhotsk Sea bowhead whales. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/65b/BRG17). 5 p.

Smith, L. C., and S. R. Stephenson. 2013. New trans-Arctic shipping routes navigable by midcentury. Proc. Nat. Acad. Sci. 110(13):4871-4872. DOI: dx.doi.org/10.1073/pnas.1214212110 .

Stoker, S. W., and I. I. Krupnik. 1993. Subsistence whaling, p. 579-629. *In* J. J. Burns, J. J. Montague, and C. J. Cowles (eds.), The Bowhead Whale. Soc. Mar. Mammal., Spec. Publ. No. 2.

Suydam, R., and J. C. George. 2012. Preliminary analysis of subsistence harvest data concerning bowhead whales (*Balaena mysticetus*) taken by Alaskan Natives, 1974 to 2011. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/64/AWMP8). 13 p.

Suydam, R. S., R. P. Angliss, J. C. George, S. R. Braund, and D. P. DeMaster. 1995. Revised data on the subsistence harvest of bowhead whales (*Balaena mysticetus*) by Alaska Eskimos, 1973-1993. Rep. Int. Whal. Comm. 45:335-338.

Suydam, R., J. C. George, B. Person, C. Hanns, and G. Sheffield. 2011. Subsistence harvest of bowhead whales (*Balaena mysticetus*) by Alaskan Eskimos during 2010. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/63/BRG2). 7 p.

Suydam, R., J. C. George, B. Person, C. Hanns, R. Stimmelmayr, L. Pierce, and G. Sheffield. 2012. Subsistence harvest of bowhead whales (*Balaena mysticetus*) by Alaskan Eskimos during 2011. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/64/BRG2). 8 p.

Suydam R., J. C. George, B. Person, C. Hanns, R. Stimmelmayr, L. Pierce, and G. Sheffield. 2013. Subsistence harvest of bowhead whales (*Balaena mysticetus*) by Alaskan Eskimos during 2012. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/65a/BRG19). 7 p.

Suydam, R., J. C. George, B. Person, C. Hanns, R. Stimmelmayr, L. Pierce, and G. Sheffield. 2014. Subsistence harvest of bowhead whales (*Balaena mysticetus*) by Alaskan Eskimos during 2013. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/65b/BRG8). 10 p.

Suydam, R., J. C. George, B. Person, D. Ramey, C. Hanns, R. Stimmelmayr, L. Pierce, and G. Sheffield. 2015. Subsistence harvest of bowhead whales (*Balaena mysticetus*) by Alaskan Eskimos during 2014. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/66a/BRG6). 9 p.

Suydam, R., J. C. George, B. Person, D. Ramey, R. Stimmelmayr, T. Sformo, L. Pierce, and G. Sheffield. 2016. Subsistence harvest of bowhead whales (*Balaena mysticetus*) by Alaskan Eskimos during 2015 and other aspects of bowhead biology and science. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/66b/BRG3). 10 p.

Suydam, R., J. C. George, B. Person, D. Ramey, R. Stimmelmayr, T. Sformo, L. Pierce, and G. Sheffield. 2017. Subsistence harvest of bowhead whales (*Balaena mysticetus*) by Alaskan Eskimos during 2016. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/67a/AWMP02rev). 8 p.

Suydam, R., J. C. George, B. Person, R. Stimmelmayr, T. Sformo, L. Pierce, A. Von Duyke, L. de Sousa, and G. Sheffield. 2018. Subsistence harvest of bowhead whales (*Balaena mysticetus*) by Alaskan Natives during 2017. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/67b/AWMP/05). 8 p.

Suydam, R., J. C. George, B. T. Person, R. Stimmelmayr, T. L. Sformo, L. Pierce, A. Von Duyke, L. de Sousa, R. Acker, G. Sheffield, and A. Baird. 2019. Subsistence harvest of bowhead whales (*Balaena mysticetus*) by Alaskan Natives during 2018. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/68a/ASW02). 9 p.

U.S. Geological Survey (USGS). 2011. An evaluation of the science needs to inform decisions on Outer Continental Shelf energy development in the Chukchi and Beaufort seas. L. Holland-Bartels and B. Pierce (eds.), Alaska: U.S. Geological Survey Circular 1370. 278 p.

Walsh, J. E. 2008. Climate of the arctic marine environment. Ecol. Appl. 18(2)Suppl.:S3-S22. DOI: dx.doi.org/10.1890/06-0503.1 .

Wiig, Ø., L. Bachmann, N. Øien, K. M. Kovacs, and C. Lydersen. 2009. Observations of bowhead whales (*Balaena mysticetus*) in the Svalbard area 1940-2008. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/61/BRG2). 5 p.

Wiig, Ø., L. Bachmann, M. P. Heide-Jørgensen, K. L. Laidre, L. D. Postma, L. Dueck, and P. J. Palsbøl. 2010. Within and between stock re-identification of bowhead whales in Eastern Canada and West Greenland. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/62/BRG25). 7 p.

Wiig, Ø., M. P. Heide-Jørgensen, C. Lindqvist, K. L. Laidre, P. J. Palsbøll, and L. Bachmann. 2011. Population estimates of mark and recaptured genotyped bowhead whales (*Balaena mysticetus*) in Disko Bay, West Greenland. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/63/BRG18). 4 p.

Wilkinson, J., C. J. Beegle-Krause, K.-U. Evers, N. Hughes, A. Lewis, M. Reed, and P. Wadhams. 2017. Oil spill response capabilities and technologies for ice-covered Arctic marine waters: A review of recent developments and established practices. Ambio 46(Suppl. 3):S423–S441. DOI: dx.doi.org/10.1007/s13280-017-0958-y .

Willoughby, A. L., J. T. Clarke, M. C. Ferguson, R. Stimmelmayr, and A. B. Brower. 2018. Bowhead whale carcasses in the eastern Chukchi and western Beaufort seas, 2009-2017. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/67b/AWMP2). 10 p.

Woodby, D. A., and D. B. Botkin. 1993. Stock sizes prior to commercial whaling, p. 387-407. *In* J. J. Burns, J. J. Montague, and C. J. Cowles (eds.), The Bowhead Whale. Soc. Mar. Mammal., Spec. Publ. No. 2.

Yablokov, A. V. 1994. Validity of whaling data. Nature 367:108.

Young, N. C., B. J. Delean, V. T. Helker, J. C. Freed, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. 2020. Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks, 2014-2018. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-413, 142 p.

Zeh, J. E., and A. E. Punt. 2005. Updated 1978-2001 abundance estimates and their correlations for the Bering-Chukchi-Beaufort Seas stock of bowhead whales. J. Cetacean Res. Manage. 7(2):169-175.

Zeh, J. E., C. W. Clark, J. C. George, D. E. Withrow, G. M. Carroll, and W. R. Koski. 1993. Current population size and dynamics, p. 409-489. *In* J. J. Burns, J. J. Montague, and C. J. Cowles (eds.), The Bowhead Whale. Soc. Mar. Mammal., Spec. Publ. No. 2.

Zharikov, K. 2018. Aboriginal subsistence whaling in the Russian Federation in 2017. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/67b/AWMP/WP/02). 2 p.

Zharikov, K. A., D. I. Litovka, and E. V. Vereshagin. 2019. Aboriginal subsistence whaling in the Russian Federation during 2018. Unpubl. doc. submitted to Int. Whal. Comm. Scientific Committee (SC/68a/ASW03). 3 p.

Appendix 1. Summary of substantial changes to the text and/or values in the 2020 stock assessments (last revised 12/30/2020). An 'X' indicates sections where the information presented has been updated since the 2019 stock assessments were released. Stock Assessment Reports for those stocks in boldface were updated in 2020.

Stock	Stock definition	Population size	PBR	Fishery mortality	Subsistence mortality	Status
Steller sea lion (Western U.S.)	X	X	X	X	X	X
Steller sea lion (Eastern U.S.)						
Northern fur seal (Eastern Pacific)		X	X	X	X	X
Harbor seal (Aleutian Islands)						
Harbor seal (Pribilof Islands)						
Harbor seal (Bristol Bay)						
Harbor seal (North Kodiak)						
Harbor seal (South Kodiak)						
Harbor seal (Prince William Sound)						
Harbor seal (Cook Inlet/Shelikof Strait)						
Harbor seal (Glacier Bay/Icy Strait)						
Harbor seal (Lynn Canal/Stephens Passage)						
Harbor seal (Sitka/Chatham Strait)						
Harbor seal (Dixon/Cape Decision)						
Harbor seal (Clarence Strait)						
Spotted seal (Bering)	X	X	X	X	X	X
Bearded seal (Beringia)	X	X	X	X	X	X
Ringed seal (Arctic)	X	X	X	X	X	X
Ribbon seal	X	X	X	X	X	X
Beluga whale (Beaufort Sea)	X	X	X	X	X	X
Beluga whale (Eastern Chukchi Sea)	X	X	X	X	X	X
Beluga whale (Eastern Bering Sea)						
Beluga whale (Bristol Bay)	X	X	X	X	X	X
Beluga whale (Cook Inlet)	X	X	X	X	X	X
Narwhal (Unidentified)						
Killer whale (ENP Alaska Resident)						
Killer whale (ENP Northern Resident)						
Killer whale (ENP Gulf of Alaska, Aleutian Islands, and Bering Sea Transient)	X		X	X	X	X
Killer whale (AT1 Transient)	X	X		X	X	
Killer whale (West Coast Transient)	X	X	X	X	X	X
Pacific white-sided dolphin (North Pacific)						
Harbor porpoise (Southeast Alaska)	X	X		X		
Harbor porpoise (Gulf of Alaska)	X			X		
Harbor porpoise (Bering Sea)	X	X		X		X
Dall's porpoise (Alaska)						
Sperm whale (North Pacific)				X		X
Baird's beaked whale (Alaska)						
Cuvier's beaked whale (Alaska)						
Stejneger's beaked whale (Alaska)						
Humpback whale (Western North Pacific)				X	X	X
Humpback whale (Central North Pacific)	X			X	X	X
Fin whale (Northeast Pacific)				X		X
Minke whale (Alaska)						
North Pacific right whale (Eastern North Pacific)	X	X	X	X		
Bowhead whale (Western Arctic)	X		X	X	X	X

Appendix 2. Stock summary table (last revised 12/30/2020). N/A indicates data are unknown. UNDET (undetermined) PBR indicates data are available to calculate a PBR level but a determination has been made that calculating a PBR level using those data is inappropriate (see Stock Assessment Report (SAR) for details). N_{EST} is the AFSC Marine Mammal Laboratory's best estimate of the size of the population; Strategic status: S = Strategic, NS = Not Strategic. NOTE: This summary table has been reformatted/revised to be consistent with the summary tables in the U.S. Pacific and Atlantic SARs.

Species	Stock name	SAR updated	N_{EST}	CV N_{EST}	N_{MIN}	R_{MAX}	F_R	PBR	Total annual mortality/ serious injury	Annual U.S. commercial fishery mortality/ serious injury	Annual Native subsistence mortality	Strategic status	SAR last revised	Last survey year(s) for estimating abundance	Comments
Steller sea lion	Western U.S.	Y	52,932		52,932	0.12	0.1	318	254	37	209	S	2019	2018-2019	N_{EST} is best estimate of counts, which have not been corrected for animals at sea during abundance surveys.
Steller sea lion	Eastern U.S.	N	43,201		43,201	0.12	1.0	2,592	112	24	11	NS	2019	2017	N_{EST} is best estimate of counts, which have not been corrected for animals at sea during abundance surveys.
Northern fur seal	Eastern Pacific	Y	608,143	0.2	514,738	0.086	0.5	11,067	387	3.4	373	S	2019	2014-2018	Survey years = Sea Lion Rock - 2014; St. Paul and St. George Is. – 2014, 2016, 2018; Bogoslof Is. – 2015.
Harbor seal	Aleutian Islands	N	5,588		5,366	0.12	0.3	97	90	0.4	90	NS	2019	2018	
Harbor seal	Pribilof Islands	N	229		229	0.12	0.5	7	0	0	0	NS	2019	2018	N_{EST} is best estimate of counts, which have not been corrected for animals at sea during abundance surveys.

Species	Stock name	SAR updated	N_{EST}	CV N_{EST}	N_{MIN}	R_{MAX}	F_R	PBR	Total annual mortality/ serious injury	Annual U.S. commercial fishery mortality/ serious injury	Annual Native subsistence mortality	Strategic status	SAR last revised	Last survey year(s) for estimating abundance	Comments
Harbor seal	Bristol Bay	N	44,781		38,254	0.12	0.7	1,607	20	3.8	15	NS	2019	2017	
Harbor seal	North Kodiak	N	8,677		7,609	0.12	0.5	228	38	0.3	37	NS	2019	2017	
Harbor seal	South Kodiak	N	26,448		22,351	0.12	0.7	939	127	1.2	126	NS	2019	2017	
Harbor seal	Prince William Sound	N	44,756		41,776	0.12	0.5	1,253	413	24	387	NS	2019	2015	
Harbor seal	Cook Inlet/Shelikof Strait	N	28,411		26,907	0.12	0.5	807	107	2.5	104	NS	2019	2018	
Harbor seal	Glacier Bay/Icy Strait	N	7,455		6,680	0.12	0.3	120	104	0	104	NS	2019	2017	
Harbor seal	Lynn Canal/Stephens Passage	N	13,388		11,867	0.12	0.3	214	50	0	50	NS	2019	2016	
Harbor seal	Sitka/Chatham Strait	N	13,289		11,883	0.12	0.5	356	77	0	77	NS	2019	2015	
Harbor seal	Dixon/Cape Decision	N	23,478		21,453	0.12	0.5	644	69	0	69	NS	2019	2015	
Harbor seal	Clarence Strait	N	27,659		24,854	0.12	0.5	746	40	0	40	NS	2019	2015	
Spotted seal	Bering	Y	461,625		423,237	0.12	1.0	25,394	5,254	1	5,253	NS	2017	2012-2013	
Bearded seal	Beringia	Y				0.12	0.5		6,709	1.8	6,707	S	2019	2012-2013	N_{EST}, N_{MIN}, and PBR have been calculated, however, important caveats exist; see SAR text for details.
Ringed seal	Arctic	Y				0.12	0.5		6,459	5	6,454	S	2019	2012-2013	N_{EST}, N_{MIN}, and PBR have been calculated, however, important caveats exist; see SAR text for details.

Species	Stock name	SAR updated	N_{EST}	CV N_{EST}	N_{MIN}	R_{MAX}	F_R	PBR	Total annual mortality/ serious injury	Annual U.S. commercial fishery mortality/ serious injury	Annual Native subsistence mortality	Strategic status	SAR last revised	Last survey year(s) for estimating abundance	Comments
Ribbon seal		Y	184,697		163,086	0.12	1.0	9,785	163	0.9	162	NS	2018	2012-2013	
Beluga whale	Beaufort Sea	Y	39,258	0.229	N/A	0.04	1.0	UNDET	104	0	104	NS	2017	1992	
Beluga whale	Eastern Chukchi Sea	Y	13,305	0.51	8,875	0.04	1.0	178	56	0	56	NS	2017	2017	
Beluga whale	Eastern Bering Sea	N	6,994	0.37	N/A	0.04	1.0	UNDET	206	0.2	206	NS	2017	2000	
Beluga whale	Bristol Bay	Y	2,040	0.26	1,645	0.04	1.0	33	19		19	NS	2017	2016	
Beluga whale	Cook Inlet	Y	279	0.061	267	0.04	0.1		0	0	0	S	2019	2014-2018	Survey years = 2014, 2016, and 2018. PBR has been calculated, however, important caveats exist; see SAR text for details.
Narwhal	Unidentified	N	N/A		N/A	0.04	0.5	N/A	0	0	0	NS	2016		
Killer whale	Eastern North Pacific Alaska Resident	N	2,347	N/A	2,347	0.04	0.5	24	1	1	0	NS	2016	2012	N_{EST} is based on counts of individuals identified from photo-ID catalogues.
Killer whale	Eastern North Pacific Northern Resident (British Columbia)	N	302	N/A	302	0.029	0.5	2.2	0.2	0	0	NS	2019	2018	N_{EST} is based on counts of individuals identified from photo-ID catalogues.
Killer whale	Eastern North Pacific Gulf of Alaska, Aleutian Islands, and Bering Sea Transient	Y	587	N/A	587	0.04	0.5	5.9	0.8	0.8	0	NS	2016	2012	N_{EST} is based on counts of individuals identified from photo-ID catalogues.

Species	Stock name	SAR updated	N_{EST}	CV N_{EST}	N_{MIN}	R_{MAX}	F_R	PBR	Total annual mortality/ serious injury	Annual U.S. commercial fishery mortality/ serious injury	Annual Native subsistence mortality	Strategic status	SAR last revised	Last survey year(s) for estimating abundance	Comments
Killer whale	AT1 Transient	Y	7	N/A	7	0.04	0.1		0	0	0	S	2019	2019	N_{EST} is based on counts of individuals identified from photo-ID catalogues. PBR has been calculated, however, important caveats exist; see SAR text for details.
Killer whale	West Coast Transient	Y	349	N/A	349	0.04	0.5	3.5	0.4	0.2	0	NS	2013	2018	N_{EST} is based on counts of individuals identified from photo-ID catalogues in an analysis of a subset of data from 1958 to 2018.
Pacific white-sided dolphin	North Pacific	N	26,880	N/A	N/A	0.04	0.5	UNDET	0	0	0	NS	2018	1990	
Harbor porpoise	Southeast Alaska	Y				0.04	0.5		34	34	0	S	2019	2010-2012	N_{EST}, N_{MIN}, and PBR have been calculated, however, important caveats exist; see SAR text for details.
Harbor porpoise	Gulf of Alaska	Y	31,046	0.21	N/A	0.04	0.5	UNDET	72	72	0	S	2019	1998	
Harbor porpoise	Bering Sea	Y			N/A	0.04	0.5	UNDET	0.4	0	0	S	2019	2008	N_{EST} has been calculated, however, important caveats exist; see SAR text for details.
Dall's porpoise	Alaska	N	83,400	0.097	N/A	0.04	1.0	UNDET	38	38	0	NS	2018	1991	

Species	Stock name	SAR updated	N_{EST}	CV N_{EST}	N_{MIN}	R_{MAX}	F_R	PBR	Total annual mortality/ serious injury	Annual U.S. commercial fishery mortality/ serious injury	Annual Native subsistence mortality	Strategic status	SAR last revised	Last survey year(s) for estimating abundance	Comments
Sperm whale	North Pacific	Y				0.04	0.1		3.5	3.3	0	S	2019	2015	N_{EST}, N_{MIN}, and PBR have been calculated, however, important caveats exist; see SAR text for details.
Baird's beaked whale	Alaska	N	N/A		N/A	0.04	0.5	N/A	0	0	0	NS	2013		
Cuvier's beaked whale	Alaska	N	N/A		N/A	0.04	0.5	N/A	0	0	0	NS	2013		
Stejneger's beaked whale	Alaska	N	N/A		N/A	0.04	0.5	N/A	0	0	0	NS	2013		
Humpback whale	Western North Pacific	Y	1,107	0.300	865	0.07	0.1	3.0	2.8	0.9	0	S	2019	2004-2006	
Humpback whale	Central North Pacific - entire stock	Y	10,103	0.300	7,891	0.07	0.3	83	26	9.8	0	S	2019	2004-2006	
Fin whale	Northeast Pacific	Y				0.04	0.1		0.6	0	0	S	2019	2013	N_{EST}, N_{MIN}, and PBR have been calculated, however, important caveats exist; see SAR text for details.
Minke whale	Alaska	N	N/A		N/A	0.04	0.5	N/A	0	0	0	NS	2018		
North Pacific right whale	Eastern North Pacific	Y	31	0.226	26	0.04	0.1		0	0	0	S	2019	2008	PBR has been calculated, however, important caveats exist; see SAR text for details.
Bowhead whale	Western Arctic	Y	16,820	0.052	16,100	0.04	0.5	161	56	0.2	56	S	2019	2011	

Appendix 3. Percent observer coverage in Alaska commercial fisheries 1990-2018 (last revised 12/30/2020).

Fishery name[a]	Method for calculating observer coverage[b]	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018
Gulf of Alaska (GOA) groundfish trawl	% of observed biomass	55	38	41	37	33	44	37	33	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
GOA flatfish trawl	% of observed biomass	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	39.2	35.8	36.8	40.5	35.9	40.6	76.9	29.2	24.2	31	28	22	26	31	42	46	47	54	39	56	34
GOA Pacific cod trawl	% of observed biomass	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	20.6	16.4	13.5	20.3	23.2	27.0	82.5	21.4	22.8	25	24	38	31	41	25	10	12	13	13	11	25
GOA pollock trawl	% of observed biomass	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	37.5	31.7	27.5	17.6	26.0	31.4	96.1	24.2	26.5	27	34	43			27	15	14	23	27	19	21
GOA rockfish trawl	% of observed biomass	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	51.4	49.8	50.2	51.0	37.2	48.4	74.1	51.4	49.1	88	87	91			95	95	96	93	98	98	95
GOA longline	% of observed biomass	21	15	13	13	8	18	16	15	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
GOA Pacific cod longline	% of observed biomass	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	3.8	5.7	6.1	4.9	11.4	12.6	21.4	3.7	10.2	45	32	43	29	30	13	29	31	36	30	40	29
GOA halibut longline	% of observed biomass	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	51.3	47.1	51.1	43.0	41.4	9.6	36.4	6.5	2.8	N/A	N/A	N/A		2.3	0.6	4.2	11	9.4	9.5	4.6	6.4
GOA rockfish longline	% of observed biomass	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	1.0	1.4	0.2	1.3	4.9	2.5	0	0	3.1	N/A	N/A	83			0	0	3.2	10	6.7	12	0
GOA sablefish longline	% of observed biomass	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	16.9	14.0	15.2	12.4	13.7	9.4	37.7	10.4	11.2	37	35	38	15	14	14	14	19	20	14	12	9.8
GOA finfish pots	% of observed biomass	13	9	9	7	7	7	5	4	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
GOA Pacific cod pot	% of observed biomass	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	6.7	5.7	7.0	5.8	7.0	4.0	40.6	3.8	2.9	14	18	13			9.6	8.4	8.7	14	8.3	2.9	8.8
Bering Sea/Aleutian Islands (BSAI) finfish pots	% of observed biomass	43	36	34	41	27	20	17	18	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A

Fishery name[a]	Method for calculating observer coverage[b]	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018
BSAI Pacific cod pot	% of observed biomass	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	14.6	16.2	8.5	14.7	12.1	12.4	33.1	14.4	12.4	30	23	29	21	20	19	18	21	27	21	13	21
BS sablefish pot	% of observed biomass	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	42.1	44.1	62.6	38.7	40.6	21.4	72.5	44.3	35.3	N/A	N/A	N/A			39	13	11	9	23	19	33
AI sablefish pot	% of observed biomass	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	100	50.3	68.2	60.6	69.4	47.5	51.2	64.4	18.7	N/A	N/A	N/A			40	0	0	86	88	33	55
BSAI groundfish trawl	% of observed biomass	74	53	63	66	64	67	66	64	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
BSAI Atka mackerel trawl	% of observed biomass	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	65.0	77.2	86.3	82.4	98.3	95.4	96.6	97.8	96.7	94	100	99	100	99	100	99	100	100	98	100	100
BSAI flatfish trawl	% of observed biomass	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	59.4	66.3	64.5	57.6	58.4	63.9	68.2	68.3	67.8	72	100	100	99	99	100	100	100	100	99	100	100
BSAI Pacific cod trawl	% of observed biomass	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	55.3	50.6	51.7	57.8	47.4	49.9	75.1	52.8	46.8	52	56	64	66	60	68	80	80	72	68	68	73
BSAI pollock trawl	% of observed biomass	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	66.9	75.2	76.2	79.0	80.0	82.2	92.8	77.3	73.0	85	85	86	86	98	98	98	98	99	99	99	99
BSAI rockfish trawl	% of observed biomass	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	85.4	85.6	85.1	65.3	79.9	82.6	94.1	71.0	80.6	88	98	99	99	99	100	100	100	100	100	100	100
BSAI longline	% of observed biomass	80	54	35	30	27	28	29	33	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
BSAI Greenland turbot longline	% of observed biomass	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	31.6	30.8	52.8	33.5	37.3	40.9	39.3	33.7	36.2	64	74	74	59	59	57	52	56	52	60	56	62
BSAI Pacific cod longline	% of observed biomass	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	34.4	31.8	35.2	29.5	29.6	29.8	25.7	24.6	26.3	63	63	61	64	57	51	66	64	62	57	58	55
BSAI halibut longline	% of observed biomass	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	38.9	48.4	55.3	67.2	57.4	20.3	44.5	27.9	26.4	N/A	N/A	N/A		16	1.8	13	11	13	10	6.9	8.2
BSAI rockfish longline	% of observed biomass	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	41.5	21.4	53.0	26.9	36.0	74.9	37.9	36.3	46.8	88	N/A	100			34	49	100	70	53	0	83

Fishery name[a]	Method for calculating observer coverage[b]	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018
BSAI sablefish longline	% of observed biomass	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	19.5	28.4	24.4	18.9	30.3	10.4	50.9	19.3	11.2	48	49	56			27	42	35	34	22	6.9	7.7
Prince William Sound salmon drift gillnet	% of estimated sets observed	4	5	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.
Prince William Sound salmon set gillnet	% of estimated sets observed	3	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.
Alaska Peninsula/Aleutian Islands salmon drift gillnet (South Unimak area only)	% of estimated sets observed	4	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.
Cook Inlet salmon drift gillnet	% of fishing days observed	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	1.6	3.6	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.
Cook Inlet salmon set gillnet	% of fishing days observed	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	0.16-1.1	0.34-2.7	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.
Kodiak Island salmon set gillnet	% of fishing days observed	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	6.0	not obs.	not obs.	4.9	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.
Yakutat salmon set gillnet	% of fishing days observed	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	5.3	7.6	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.
Southeast Alaska salmon drift gillnet (Districts 6, 7, and 8)	% of fishing days observed	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	not obs.	6.4	6.6	not obs.	not obs.	not obs.	not obs.	not obs.

[a]From 1990 to 1997, most federally-regulated commercial fisheries in Alaska were named using gear type and fishing location. In 2003, the naming convention changed to define fisheries based on gear type, fishing location, and target fish species. Bycatch data collected from 1998 to present are analyzed using these fishery definitions. The use of "N/A" for either pooled or separated fisheries indicates that we do not have effort data for a particular fishery for that year.

[b]Observer coverage in the groundfish fisheries (trawl, longline, and pots) was determined by the percentage of tons caught which were observed. Observer coverage in the groundfish fisheries is assigned according to vessel length; where vessels greater than 125 feet have 100% coverage, vessels 60-125 feet have 30% coverage, and vessels less than 60 feet are not observed. Observer coverage in the groundfish fisheries varies by statistical area; the pooled percent coverage for all areas is provided here. Observer coverage in the drift gillnet fisheries was calculated as the percentage of the estimated sets that were observed. Observer coverage in the set gillnet fishery was calculated as the percentage of estimated setnet hours (determined by number of permit holders and the available fishing time) that were observed.

REFERENCES

Breiwick, J. M. 2013. North Pacific marine mammal bycatch estimation methodology and results, 2007-2011. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-260, 40 p.

Manly, B. F. J. 2006. Incidental catch and interactions of marine mammals and birds in the Cook Inlet salmon driftnet and setnet fisheries, 1999-2000. Final Report to NMFS Alaska Region. 98 p.

Manly, B. F. J. 2007. Incidental take and interactions of marine mammals and birds in the Kodiak Island salmon set gillnet fishery, 2002 and 2005. Final Report to NMFS Alaska Region. 221 p.

Manly, B. F. J. 2009. Incidental catch of marine mammals and birds in the Yakutat salmon set gillnet fishery, 2007 and 2008. Final Report to NMFS Alaska Region. 96 p.

Manly, B. F. J. 2015. Incidental takes and interactions of marine mammals and birds in districts 6, 7, and 8 of the Southeast Alaska salmon drift gillnet fishery, 2012 and 2013. Final Report to NMFS Alaska Region. 52 p.

Perez, M. A. 2006. Analysis of marine mammal bycatch data from the trawl, longline, and pot groundfish fisheries of Alaska, 1998-2004, defined by geographic area, gear type, and target groundfish catch species. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-167, 194 p.

Perez, M. A. Unpubl. ms. Bycatch of marine mammals by the groundfish fisheries in the U.S. EEZ of Alaska, 2005. 67 p. Available from Marine Mammal Laboratory, AFSC, 7600 Sand Point Way NE, Seattle, WA 98115.

Wynne, K. M., D. Hicks, and N. Munro. 1991. 1990 salmon gillnet fisheries observer programs in Prince William Sound and South Unimak Alaska. Annual Report NMFS/NOAA Contract 50ABNF000036. 65 p. Available from NMFS Alaska Region, Office of Marine Mammals, P.O. Box 21668, Juneau, AK 99802.

Wynne, K. M., D. Hicks, and N. Munro. 1992. 1991 marine mammal observer program for the salmon driftnet fishery of Prince William Sound Alaska. Annual Report NMFS/NOAA Contract 50ABNF000036. 53 p. Available from NMFS Alaska Region, Office of Marine Mammals, P.O. Box 21668, Juneau, AK 99802.

Appendix 4. Stock Assessment Reports published by the U.S. Fish and Wildlife Service.

POLAR BEAR (*Ursus maritimus*): Chukchi/Bering Seas Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Polar bears are circumpolar in their distribution in the northern hemisphere. They occur in several largely discrete stocks or populations (Harington 1968). Polar bear movements are extensive and individual activity areas are enormous (Garner *et al.* 1990, Amstrup *et al.* 2000). The parameters used by Dizon *et al.* (1992) to classify stocks based on the phylogeographic approach were considered in the determination of stock separation in Alaska. Several polar bear stocks are known to be shared between countries (Amstrup *et al.* 1986, Amstrup and DeMaster 1988). Lentfer hypothesized that in Alaska two stocks exist, the Southern Beaufort Sea (SBS) and the Chukchi/Bering seas (CBS), based upon: (a) variations in levels of heavy metal contaminants of organ tissues (Lentfer 1976, Lentfer and Galster 1987); (b) morphological characteristics (Manning 1971, Lentfer 1974, Wilson 1976); (c) physical oceanographic features which segregate the Chukchi Sea and Bering Sea stock from the Beaufort Sea stock (Lentfer 1974); and (d) movement information collected from mark and recapture studies of adult female bears (Lentfer 1974, 1983) (Figure 1). Information on contaminants (Woshner *et al.* 2001, Evans 2004a, Evans 2004b, Kannan *et al.* 2005, Smithwick *et al.* 2005, Verreault *et al.* 2005, Muir *et al.* 2006, Smithwick *et al.* 2006, Kannan *et al.* 2007, Rush *et al.* 2008) and movement data using satellite collars (Amstrup *et al.* 2004, Amstrup *et al.* 2005) continue to support the presence of these two stocks.

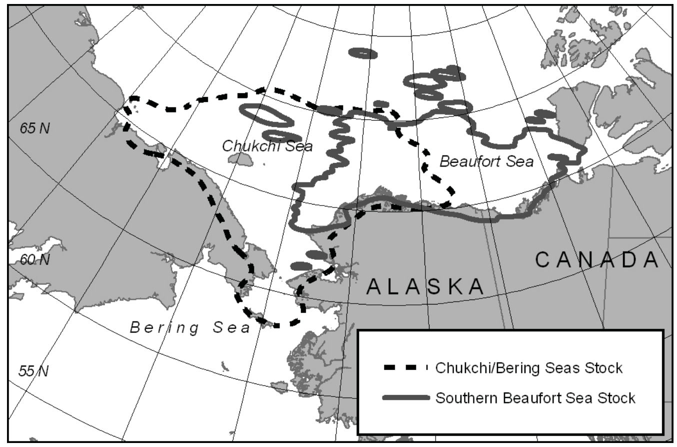

Figure 1. Map of the Southern Beaufort Sea and the Chukchi/Bering seas polar bear stocks.

The CBS population is widely distributed on the pack ice in the Chukchi Sea and northern Bering Sea and adjacent coastal areas in Alaska and Russia. The northeastern boundary of the Chukchi/Bering seas stock is near the Colville Delta in the central Beaufort Sea (Garner *et al.*1990, Amstrup 1995, Amstrup *et al.* 2005) and the western boundary is near Chauniskaya Bay in the Eastern Siberian Sea. The boundary between the Eastern Siberian Sea stock and the Chukchi Sea stock is designated based on movements of adult female polar bears captured in the Bering and Chukchi seas region. Female polar bears initially captured and radio collared on Wrangel Island exhibited no movement into the Eastern Siberian Sea, while female polar bears captured and radio collared in the Eastern Siberian Sea, exhibited only limited short term movement into the western Chukchi Sea (Garner *et al.* 1990). The Chukchi/Bering seas stock extends into the Bering Sea and its southern boundary is determined by the annual extent of pack ice (Garner *et al.* 1990). Adult female polar bears captured from the Southern Beaufort Sea stock may make seasonal movements into the Chukchi Sea in an area of overlap located between Point Hope and Colville Delta, centered near Point Lay (Garner *et al.* 1990, Garner *et al.* 1994, Amstrup 1995, Amstrup *et al.* 2002, Amstrup *et al.* 2005). Probabilistic distribution information for zones of overlap between the Chukchi/Bering seas and the Southern Beaufort Sea population exist (Amstrup *et al.* 2004, Amstrup *et al.* 2005). Telemetry data indicate that these bears, marked in the Beaufort Sea, spend about 25% of their time in the northeastern Chukchi Sea, whereas females captured in the Chukchi Sea spend only 6% of their time in the Beaufort Sea (Amstrup 1995). Average activity areas of females in the Chukchi/Bering seas from 1986–1988 (244,463 km^2, range 144,659–351,369 km^2) (Garner *et al.* 1990) were more extensive than the Beaufort Sea from 1983– 1985 (96,924 km^2, range 9,739–269,622 km^2) (Amstrup 1986) or from 1985–1995 (166,694 km^2, range 14,440–616,800 km^2) (Amstrup *et al.* 2000). Radio collared adult females spent a greater proportion of their time in the Russian region than in the American region (Garner *et al.* 1990). Historically polar bears ranged as far south as St. Matthew Island (Hanna 1920) and the Pribilof Islands (Ray 1971) in the Bering Sea.

Analysis of mitochondrial DNA indicates little differentiation of the Alaska polar bear stocks (Cronin *et al.* 1991, Scribner *et al.* 1997, Cronin *et al.* 2006). Using 16 highly variable micro-satellite loci, Paetkau *et al.* (1999) determined that polar bears throughout the arctic (19 populations) are genetically similar. Genetically, polar bears in the southern Beaufort Sea differed more from polar bears in the Chukchi/Bering seas than from polar bears in the northern Beaufort Sea (Paetkau *et al.* 1999).

While genetically similar, demographic and movement data of the CBS population, indicates a high degree of site fidelity, suggesting that the stocks should be managed separately (Amstrup 2000, Amstrup *et al.* 2000, Amstrup *et al.* 2001a, Amstrup *et al.* 2002, Amstrup *et al.* 2004, Amstrup *et al.* 2005).

Past management has consistently distinguished between the southern Beaufort Sea and the Chukchi/Bering seas stocks. The Inuvialuit of the Inuvialuit Game Council (IGC), Northwest Territories, and the Inupiat of the North Slope Borough (NSB), Alaska, polar bear management agreement for the Southern Beaufort Sea stock was based on stock boundaries described previously (Brower *et al.* 2002, Nageak *et al.* 1991, Treseder and Carpenter 1989) and reaffirmed by the information in this stock assessment report.

POPULATION SIZE

Polar bears typically occur at low densities throughout their circumpolar range (DeMaster and Stirling 1981). It has been difficult to obtain a reliable population estimate for this population due to the vast and inaccessible nature of the habitat, movement of bears across international boundaries, logistical constraints of conducting studies in Russian territory, and budget limitations (Amstrup and DeMaster 1988, Garner *et al.* 1992, Garner *et al.* 1998, Evans *et al.* 2003). The Chukchi Sea population is estimated to comprise 2,000 animals, based on extrapolation of aerial den surveys (Lunn *et al.* 2002). Estimates of the population have been derived from observations of dens and aerial surveys (Chelintsev 1977, Stishov 1991a, Stishov 1991b, Stishov *et al.* 1991); however, these estimates (see below) have wide confidence intervals and are considered to be of little value for management and cannot be used to evaluate status and trends for this population.

Minimum Population Estimate

A reliable population estimate for the Chukchi/Bering seas stock currently does not exist. Lentfer, in the Administrative Law Judge (ALJ) proceeding to waive the Marine Mammal Protection Act of 1972 (MMPA) moratorium on taking and return management to the State of Alaska (ALJ 1977), estimated the size of the Chukchi/Bering seas population stock (Wrangel Island to western Alaska) at 7,000, and Chapman estimated the Alaska population (both stocks) at 5,550 to 5,700 (ALJ 1977). Lentfer and Chapman's estimates (ALJ 1977), however, were not based on rigorous statistical analysis of population data and variance estimates could not be calculated. Amstrup *et al.* (1986) estimated densities (1976–129 km^2/bear, 1981–211 km^2/bear) based on mark and recapture of 266 polar bears near Cape Lisburne on the Chukchi Sea, but a population estimate for the Chukchi Sea was not developed at that time. An August 2000 aerial survey of polar bears in the Eastern Chukchi Sea resulted in density estimates of (0.00748 bear/km^2, or 147 km^2/bear, C.V. = 0.38) (Evans *et al.* 2003). A population estimate was not derived from this density since the study area included only a portion of the total area used by the population.

Amstrup and DeMaster (1988) estimated the Alaska population (both stocks) at 3,000 to 5,000 animals based on densities calculated previously by Amstrup *et al.* (1986). The area that the estimate applied to and the variance associated with the estimate were not provided for in the 1988 population estimate (Amstrup and DeMaster 1988). A crude population estimate for the Chukchi/Bering seas stock of 1,200 to 3,200 animals was derived by subtracting the Beaufort Sea population estimate of 1,800 animals (Amstrup 1995) from the total Alaska statewide estimate of 3,000 to 5,000 (Amstrup and DeMaster 1988). The IUCN Polar Bear Specialist Group (IUCN 2006) estimated this population to be approximately 2,000 animals based on extrapolation of multiple years of denning data for Wrangel Island, assuming that 10% of the population dens annually as adult females. However, confidence in this estimate is low due to the lack of current denning estimates and reliable data with measurable levels of precision (IUCN 2006). Nonetheless, an N_{MIN} of 2,000 is the best available information we have at this time.

Current Population Trend

Prior to the 20th century, when Alaska's polar bears were hunted primarily by Alaskan Natives, both stocks probably existed at near carrying capacity (K). The size of the Beaufort Sea stock declined substantially in the late 1960's and early 1970's (Amstrup *et al.* 1986) due to excessive sport harvest. Similar declines could have occurred in the Chukchi Sea, although there are no population data to support this assumption. Since passage of the MMPA, the southern Beaufort Sea population grew during the late 1970's and 1980's and then stabilized during the 1990's (Amstrup *et al.* 2001b). Based on demographic data 2001 to 2006, the overall population growth rate in the Southern Beaufort Sea population declined approximately 0.3% per year (Hunter *et al.* 2007). Until 1992 it is likely that the Chukchi/Bering seas stock mimicked the growth pattern and later stability of Southern Beaufort Sea stock, since both stocks experienced similar management and harvest histories. However, since 1992 the CBS population has faced different stressors than the SBS population. These include increased harvest in Russia (150 – 250 bears/yr) (Kochnev 2006, Ovsyanikov 2006, Eduard Zdor personal communication) and greater loss of summer sea ice habitat from global warming (Overland and Wang 2007), which suggest that using the growth rate for the Southern Beaufort Sea may not be applicable. The status of the Chukchi/Bering seas stock was listed as data deficient (Aars *et al.* 2006) due to the lack of abundance estimates with measurable levels of precision. The population is believed to be declining and the status relative to historical levels is believed to be reduced based on harvest levels that were demonstrated to be unsustainable in the past.

MAXIMUM NET PRODUCTIVITY RATES

Polar bears are long lived, mature at a relatively old age, have an extended breeding interval, and have small litters (Lentfer *et al.* 1980, DeMaster and Stirling 1981). Population/stock specific data to estimate R_{MAX} are not available for the Chukchi/Bering seas polar bear stock. The Southern Beaufort Sea is one of four polar bear populations with long-term data sets and as it overlaps with the Chukchi/Bering seas stock using the default value for R_{MAX} for the Southern Beaufort Sea seems reasonable as it is based on empirical data. Survival rates for the Southern Beaufort Sea stock (Regehr *et al.* 2006), which can be used in a Leslie matrix model, suggest that under optimal conditions and in the absence of human perturbations the population could increase at a rate of between 4 and 6%. Amstrup (1995) projected an annual intrinsic growth rate (including natural mortality but not human-caused mortality) of 6.03% for the Southern Beaufort Sea stock using a Leslie type matrix of recapture data. Since the Chukchi/Bering seas area is one of the most productive areas in the Arctic using the 6.03% for the Chukchi/Bering seas polar bear stock seems reasonable.

POTENTIAL BIOLOGICAL REMOVAL (PBR)

Under the 1994 reauthorized MMPA, the potential biological removal (PBR) level is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: PBR = $(N_{MIN})(½\ R_{MAX})(F_R)$. Wade and Angliss (1997) recommend a default recovery factor (F_R) of 0.5 for a threatened population or when the status of a population is unknown. We used 0.5 as the recovery factor since reliable population estimates to assess population trends are not available. In the following calculation: $(N_{MIN})(½\ R_{MAX})(F_r)$ = PBR (Wade and Angliss 1997) the minimum population estimate, N_{MIN} was 2,000; the maximum rate of increase R_{MAX} was 6.03%; and the recovery factor F_R was 0.50. Therefore, the PBR level for the Chukchi/Bering seas stock is 30 bears per year. However, confidence in these numbers is low due to dated and extrapolated population information and, therefore, the PBR value has little utility for management purposes.

ANNUAL HUMAN CAUSED MORTALITY AND SERIOUS INJURY

Fisheries Information

Polar bear stocks in Alaska have no direct interaction with commercial fisheries activities. Consequently, the total fishery mortality and serious injury rate for the Chukchi/Bering seas stock is zero.

Alaska Native Subsistence Harvest

Historically, polar bears have been killed for subsistence, handicrafts, and recreation. Based on records of skins shipped from Alaska for 1925–53, the estimated annual statewide harvest averaged 120 bears, taken primarily by Native hunters. Recreational hunting by non-native sports hunters using aircraft was common from 1951–72, increasing statewide annual harvest to 150 during 1951–60 and to 260 during 1960–72 (Amstrup *et al.* 1986, Schliebe *et al.* 1995). Hunting by non-Natives has been prohibited since 1973 when provisions of the MMPA went into effect. This reduced the mean annual statewide harvest for both populations to 98 during 1980–2007 (SD=40; range 48–242) (USFWS unpublished data). The annual harvest from the Chukchi/Bering seas stock was 92/year in the 1980s, 49/year in the 1990s, and 43/year in the 2000s. More recently, the 2003−2007 average Alaska harvest for the Chukchi/Bering seas stock in Alaska was 37 and the sex ratio was 66M:34F.

Under the MMPA, an exemption was made for Alaska Natives living in coastal communities to allow them to hunt polar bears for subsistence and making of handicrafts provided that the hunt was not done in a wasteful manner. Recently, harvest levels by Alaska Natives from the Chukchi/Bering seas stock have been declining (Figure 2). The sex ratio of known-sex bears harvested since 1980 has remained relatively consistent at 66% males and 34% females (Schliebe *et al.* 2006).

The number of unreported kills in Alaska since 1980 to the present time is approximately 7% based on: (a) tagging information; (b) interviews with local hunters; and (c) law enforcement investigations. No user agreement, similar to that between the Inuvialuit and Inupiat for the Beaufort Sea stock, exists for the Bering/Chukchi stock. Harvest levels are not limited at this time.

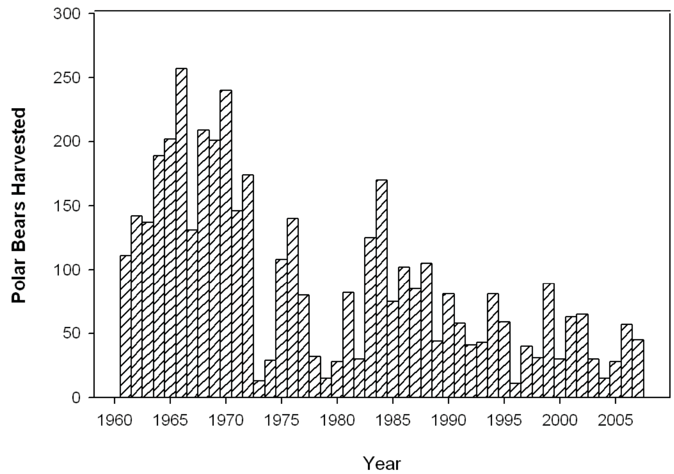

Figure 2. Annual Alaska polar bear harvest from the Chukchi/Bering Seas stock, 1961-2007.

Other Removals

Russia prohibited all hunting of polar bears in 1956 in response to perceived population declines caused by over-harvest. In Russia, only a small number of animals, less than 3–5 per year, were removed for placement in zoos prior to 1986 (Uspenski 1986) and a few were killed in defense of life. No bears were taken for zoos or circuses from 1993 to 1995 (Belikov 1998). The occurrence of increased takes of problem bears in Chukotka was acknowledged in 1992, and Belikov (1993) estimated that up to 10 problem bears were killed annually in all of the Russian Arctic. Increased illegal hunting of polar bears in the Russian Arctic was also recognized to have begun in 1992. While the magnitude of the illegal harvest in Russia from the Chukchi/Bering seas stock is unquantified, reports indicate that a substantial number of bears, 150–250/yr (Kochnev 2006), or alternatively 120–150/yr (Eduard Zdor pers. comm.), are being harvested. Combining the reported Chukotka harvest with the documented Alaska harvest indicates that up to 200 bears may have been harvested from this population in many years. Harvest levels similar to these are believed to have caused population depletion by the early 1970s. Belikov *et al.* (2006) indicated that the current level of poaching in Russia poses a serious threat to the population. No serious injuries, other than the mortalities discussed here, have been reported for the Chukchi/Bering seas stock.

No orphaned cubs from the Alaskan Chukchi/Bering seas stock were placed in zoos since 2002. Illegal harvest has not been detected in Alaska. Oil and gas exploration in the Bering/Chukchi region of Alaska, began again in 2006, primarily during the open water season has resulted in minimal interaction with polar bears; there was no evidence of mortality or serious injury.

STATUS OF STOCK

Polar bears in the Chukchi/Bering seas stock are currently classified as depleted under the MMPA and listed as threatened under the U.S. Endangered Species Act of 1973 (ESA) as amended. Reliable estimates of the minimum population, PBR level, and human-caused mortality or serious injury in Chukotka are currently not available

The ongoing level of the subsistence hunting in western Alaska and Chukotka is a concern. There is no incidental mortality or serious injury of polar bear in any U.S. commercial fishery. The primary concerns for this population are habitat loss resulting from climate change, potential over-harvest, human activities including industrial activities occurring within the near-shore environment, and potential effects of contaminants on nutritionally stressed populations. The Chukchi/Bering seas polar bear stock is designated as a strategic stock because the population is listed as threatened under the ESA.

Conservation Issues and Habitat Concerns

Oil and Gas Exploration

In 2008, the Minerals Management Service held an oil and gas lease sale for offshore blocks in the eastern Chukchi Sea. Polar bears from Chukchi/Bering seas stock seasonally use the shallow, productive, ice-covered waters of the eastern Chukchi Sea for feeding, breeding, and movements. The Fish and Wildlife Service (USFWS) works to monitor and mitigate potential impacts of oil and gas activities on polar bears through incidental take regulations (ITR) as authorized under the Marine Mammal Protection Act. Activities operating under these regulations must adopt measures to: ensure that the total of such incidental taking of polar bears remains negligible; minimize impacts to their habitat; and ensure no unmitigable adverse impact on their availability for Alaska Native subsistence use. ITR also specify monitoring requirements that provide a basis for evaluating potential impacts of current and future activities on marine mammals.

Climate Change

Polar bears evolved over thousands of years to life in a sea ice environment. They depend on the sea ice-dominated ecosystem to support essential life functions. Sea ice provides a platform for hunting and feeding, for seeking mates and breeding, for movement to terrestrial maternity denning areas and occasionally for maternity denning, for resting, and for long-distance movements. The sea ice ecosystem supports ringed seals, the primary prey for polar bears, and other marine mammals that are also part of their prey base.

Sea ice is rapidly diminishing throughout the Arctic and large declines in optimal polar bear habitat have occurred in the Southern Beaufort and Chukchi Seas between the two time periods, 1985–1995 and 1996–2006 (Durner et al 2009). In addition, it is predicted that the greatest declines in 21st century optimal polar bear habitat will occur in Chukchi and Southern Beaufort Seas (Durner *et al.* 2009a). Patterns of increased temperatures, earlier onset of and longer melting periods, later onset of freeze-up, increased rain-on-snow events, and potential reductions in snowfall are occurring. In addition, positive feedback systems (i.e., the sea-ice albedo feedback mechanism) and naturally occurring events, such as warm water intrusion into the Arctic and changing atmospheric wind patterns, can operate to amplify the effects of these phenomena. As a result, there is fragmentation of sea ice, a dramatic increase in the extent of open water areas seasonally, reduction in the extent and area of sea ice in all seasons, retraction of sea ice away from productive continental shelf areas throughout the polar basin, reduction of the amount of heavier and more stable multi-year ice, and declining thickness and quality of shore-fast ice (Parkinson et al 1999, Rothrock *et al.* 1999, Comiso 2003, Fowler *et al.* 2004, Lindsay and Zhang 2005, Holland *et al.* 2006, Comiso 2006, Serreze *et al.* 2007, Stroeve *et al.* 2008).

The Chukchi/Bering seas and the Southern Beaufort Sea population stocks are currently experiencing the initial effects of changes in sea ice conditions (Rode *et al.* 2007, Regehr *et al.* 2007, Hunter *et al.* 2007). These populations are vulnerable to large-scale dramatic seasonal fluctuations in ice movements, decreased abundance and access to prey, and increased energetic costs of hunting. The USFWS is working on measures to protect polar bears and their habitat.

Subsistence Harvest

Past differences in management regimes between the United States and Russia have made coordination of studies on the shared Alaska-Chukotka polar bear population difficult. In the former Soviet Union, hunting of polar bears was banned nationwide in 1956. Recently, Russia's ability to enforce that ban has been difficult due to logistical and financial constraints. In Alaska, subsistence hunting of polar bears by Alaska Natives is currently unrestricted under section 101(b) of the MMPA provided that the take is for subsistence purposes or creating authentic articles of Alaska Native handicrafts and conducted in a non-wasteful manner. While several joint research and management projects have been successfully undertaken in the past between the United States and Russia, today comparable efforts are either no longer occurring or are unilateral in scope.

The bilateral "Agreement between the United States and the Russian Federation on the Conservation and Management of the Alaska-Chukotka Polar Bear Population (Agreement)" was signed by the governments of the United States and the Russian Federation on October 16, 2000, with subsequent advice and consent provided by the U.S. Senate. Among other provisions the Agreement recognizes the needs of Native people to harvest polar bears for subsistence purposes and includes provisions for developing sustainable harvest limits, allocation of the harvest between jurisdictions, and compliance and enforcement. Each jurisdiction is entitled to up to one-half of a harvest limit to be determined by a future the joint Commission. The Agreement reiterates requirements of the 1973 multi-lateral agreement and includes restrictions on harvesting denning bears, females with cubs, or cubs less than one year old, and prohibitions on the use of aircraft, large motorized vessels, and snares or poison for hunting polar bears.

On January 12, 2007, President Bush signed into law H.R. 5946, the "Magnuson-Stevens Fishery Conservation and Management Reauthorization Act of 2006." This Act includes Title X implementing the Agreement. This action allows for the establishment of the commission and development of enforceable harvest management agreements. The Russian Federation and the United States have completed documents necessary to implement the Agreement within Russia and the United States. The USFWS is currently developing recommendations for the Bilateral Commission that will direct research and establish sustainable and enforceable harvest limits needed to address current potential population declines due to over-harvest of the population.

CITATIONS

Aars, J., N.J. Lunn, and A.E. Derocher. eds. 2006. Polar bears: proceedings of the 14th working meeting of the IUCN/SSC Polar Bear Specialist Group, 20-24 June, Seattle, Washington, USA. IUCN, Gland, Switzerland. 189 pp.

Administrative Law Judge. 1977. Environmental Impact Statement: Consideration of a waiver of the moratorium and return of management of certain marine mammals to the State of Alaska. 2 Volumes.

Amstrup, S.C. 1986. Research on polar bears in Alaska, 1983-1985. pp. 85-115. In Proceedings of the Ninth Working Meeting of the IUCN/SSC Polar Bear Specialist Group, Edmonton, Canada. IUCN, Gland, Switzerland, and Cambridge, U.K. 152 pp.

Amstrup, S.C. 1995. Movements, distribution, and population dynamics of polar bears in the Beaufort Sea. Ph.D. Dissertation. University of Alaska Fairbanks. Fairbanks, Alaska, 299 pp.

Amstrup, S.C. 2000. Polar Bear. In The Natural History of an Oil Field: Development and Biota. Edited by J.C. Truett and S.R. Johnson. Academic Press, Inc., New York, New York, USA pp. 133-157.

Amstrup, S.C., and D.P. DeMaster. 1988. Polar bear, *Ursus maritimus*. Pages 39-45 in J.W. Lentfer, ed. Selected Marine Mammals of Alaska: Species Accounts with Research and Management Recommendations. Marine Mammal Commission, Washington, D.C.

Amstrup, S.C., I. Stirling, and J.W. Lentfer. 1986. Past and present status of polar bears in Alaska. Wildlife Society Bulletin. 14:241-254.

Amstrup, S.C., G. Durner, I. Stirling, N.J. Lunn, and F. Messier. 2000. Movements and distribution of polar bears in the Beaufort Sea. Canadian Journal of Zoology. 78:948-966.

Amstrup, S.C., G.M. Durner, T.L. McDonald, D.M. Mulcahy, and G.W. Garner. 2001a. Comparing movement patterns of satellite-tagged male and female polar bears. Canadian Journal of Zoology. 79:2147-2158.

Amstrup, S.C., T.L. McDonald, and I. Stirling. 2001b. Polar bears in the Beaufort Sea: A 30-year mark-recapture case history. Journal of Agricultural, Biological, and Environmental Statistics Vol.(2): 221-234.

Amstrup, S.C., G. M. Durner, A. S. Fischbach, K. Simac, and G. Weston-York. 2002. Polar Bear Research in the Beaufort Sea. pp. 109-125. In N. Lunn, E. W. Born, and S. Schliebe (eds). Proceedings of the Thirteenth Working Meeting of the IUCN/SSC Polar Bear Specialist Group, Nuuk, Greenland. IUCN, Gland, Switzerland, and Cambridge, U.K. vii + 153 pp.

Amstrup, S.C., T.L. McDonald, and G.M. Durner. 2004. Using satellite radiotelemetry data to delineate and manage wildlife populations. Wildlife Society Bulletin. 32:661-679.

Amstrup, S.C., G.M. Durner, I. Stirling, and T.L. McDonald. 2005. Allocating harvests among polar bear stocks in the Beaufort Sea. Arctic. 58:247-259.

Belikov, S.E. 1993. Status of polar bear populations in the Russian Arctic 1993. pp. 115-121. In Ø. Wiig, G.W. Garner (eds.), Proceedings of the Eleventh Working Meeting of the IUCN/SSC Polar Bear Specialist Group. IUCN, Gland, Switzerland, and Cambridge, U.K. v + 192 pp.

Belikov, S.E. 1998. Research and management of polar bear populations in the Russian Arctic. pp. 113-114. In A.E. Derocher, G. Garner, N. Lunn, and Ø. Wiig (eds). Proceedings of the Twelfth Working Meeting of the IUCN/SSC Polar Bear Specialist Group. 3-7 February, 1997. Oslo, Norway. IUCN, Gland, Switzerland, and Cambridge, U.K. v + 159 pp.

Belikov, S.E., A.N. Boltunov, N.G. Ovsianikov, G.I. Belchanskiy, and A.A. Kochnev. 2006. Polar bear management and research in Russia 2000-2004. pp. 153-155. In J. Aars, N.J. Lunn, and A.E. Derocher (eds.), Proceeding of the 14th Working Meeting of the IUCN/SSC Polar Bear Specialist Group, 20-24 June 2005, Seattle, Washington, U.S.A. IUCN, Gland, Switzerland, and Cambridge, U.K.

Brower, C.D., A. Carpenter, M. Branigan, W. Calvert, T. Evans, A.S. Fischbach, J. Nagy, S. Schliebe, and I. Stirling. 2002. The polar bear management agreement for the Southern Beaufort Sea: An evaluation of the first ten years of a unique conservation agreement. Arctic 56:362-372.

Chelintsev, N.G. 1977. Determination of the absolute number of dens based on the selective counts. Pp. 66-85, In Uspenski, S.M. (ed.). The polar bear and its conservation in the Soviet Arctic. Moscow, Central Laboratory on Nature Conservation. (in Russian with English summary).

Comiso, J.C. 2003. Warming trends in the Arctic from clear sky satellite observations. Journal of Climate 16:3498-3510.

Comiso, J.C. 2006. Arctic warming signals from satellite observations, Weather 61(3):70-76.

Cronin, M.A., S.C. Amstrup, G.W. Garner, and E.R. Vyse. 1991. Interspecific and intraspecific mitochondrial DNA variation in North American bears (Ursus). Canadian Journal of Zoology. 69:12:2985-2992.

Cronin, M.A., S.C. Amstrup, K.T. Scribner. 2006. Microsatellite DNA and mitochondrial DNA variation in polar bears (*Ursus maritimus*) in the Beaufort and Chukchi seas, Alaska. Canadian Journal of Zoology 84:655-660.

DeMaster, D.P., and I. Stirling. 1981. *Ursus maritimus*. Mammalian Species: 1-7.

Dizon, A.E., C. Lockyer, W.F. Perrin, D.P. DeMaster, and J. Sisson. 1992. Rethinking the stock concept: a phylogeographic approach. Conservation Biology 6:24-36.

Durner, G.M., D.C. Douglas, R.M. Nielson, S.C. Amstrup, T.L. McDonald, I. Stirling, M. Mauritzen, E.W. Born, Ø. Wiig, E. DeWeaver. M.C. Serreze, S.E. Belikov, M.M. Holland, J. Maslanik, J. Aars, D.C. Bailey, and A.E. Derocher. 2009. Predicting 21st century polar bear habitat distribution from global climate models. Ecological Monographs 79(1): 25-58.

Evans, T.J. 2004a. Concentrations of selected essential and non-essential elements in adult male polar bears (*Ursus maritimus*) from Alaska. U.S. Fish and Wildlife Service Technical Report. MMM 04-02. 68pp.

Evans, T. J. 2004b. PCBs and chlorinated pesticides in adult male polar bears (*Ursus maritimus*) from Alaska. U.S. Fish and Wildlife Service Technical Report. MMM 04-01. 61pp

Evans, T.F., A.S. Fischbach, S. Schliebe, B. Manley, S. Kalxdorff, and G. York. 2003. Polar bear aerial survey in the Eastern Chukchi Sea: A Pilot Study. Arctic 56(4):359-366.

Fowler, C., W.J. Emery and J. Maslanik. 2004. Satellite-derived evolution of Arctic sea ice age: October 1978 to March 2003. Geoscience and Remote Sensing Letters, IEEE, Volume 1, Issue 2, April 2004. pp.71–74.

Garner, G.W., S.T. Knick, and D.C. Douglas. 1990. Seasonal movements of adult female polar bears in the Bering and Chukchi seas. International Conference on Bear Research and Management 8:219-226.

Garner, G.W., L.L. McDonald, D.S. Robson, D.P. Young Jr., and S.M. Arthur. 1992. Literature review: population estimation methodologies applicable to the estimation of abundance of polar bears. Internal Report, U.S.F.W.S. 102 pp.

Garner, G.W., S.E. Belikov, M.S. Stishov, V.G. Barnes, and S.A. Arthur. 1994. Dispersal patterns of maternal polar bears from the denning concentration on Wrangel Island. International Conference on Bear Research and Management 9(1):401-410.

Garner, G.W., M.S. Stishov, Ø. Wiig, A. Boltunov, G.I. Belchansky, D.C. Douglas, L.L. McDonald, D.M. Mulcahy, and S. Schliebe. 1998. Polar bear research in western Alaska, eastern and western Russia 1993-1996. pp. 125-129. In A.E. Derocher, G. Garner, N. Lunn, and Ø. Wiig (eds.), Proceedings of the Twelfth Working Meeting of the IUCN/SSC Polar Bear Specialist Group. 3-7 February, 1997. Oslo, Norway. IUCN, Gland, Switzerland, and Cambridge, U.K. v + 159 pp.

Hanna, G.D. 1920. Mammals of the St. Matthew Islands, Bering sea. Journal of Mammalogy 1:118-122.

Harington, C.R. 1968. Denning habits of the polar bear (*Ursus maritimus* Phipps). Canadian Wildlife Service Report, Series 5. 33 pp.

Holland, M., C.M. Bitz, and B. Tremblay. 2006. Future abrupt reductions in summer Arctic sea ice. Geophysical Research Letters 33 L25503: doi 10.1029/200661028024: 1-5.

Hunter, C.M., H. Caswell, M.C. Runge, E.V. Regehr, S.A. Amstrup, and I. Stirling. 2007. Polar bears in the Southern Beaufort Sea II: Demography and population growth in relation to sea ice conditions. U.S. Geological Survey, Alaska Science Center, Administrative Report. 46 pp.

IUCN. 2006. Status of the polar bear. pp. 33-56 in J. Aars J., N.J. Lunn, and A.E. Derocher (eds.), Proceedings of the 14th Working Meeting of the IUCN/SSC Polar Bear Specialist Group, Seattle, United States. IUCN, Gland, Switzerland, and Cambridge, U.K.

Kannan, K., S.H. Yun, and T.J. Evans. 2005. Chlorinated, brominated, and perfluorinated contaminants in livers of polar bears from Alaska. Environmental Science and Technology. 39:9057-9063.

Kannan, K., T. Agusa, T.J. Evans, and S. Tanabe. 2007. Trace element concentrations in livers of polar bears from tow population in northern and western Alaska. Archives of Environmental Contaminants and Toxicology 53:473-482.

Kochnev, A.A. 2006. Research on polar bear autumn aggregations on Chukotka, 1989-2004. pp. 157-166. In J. Aars, N.J. Lunn, and A.E. Derocher (eds.) Proceeding of the 14th Working Meeting of the IUCN/SSC Polar Bear Specialist Group, 20-24 June 2005, Seattle, Washington, U.S.A. IUCN, Gland, Switzerland, and Cambridge, U.K. Lentfer, J.W. 1974. Discreteness of Alaskan polar bear populations. Proceedings of the International Congress of Game Biologists 11:323-329.

Lentfer, J.W. 1976. Environmental contaminants and parasites in polar bears. Alaska Department of Fish and Game, Pittman-Robertson Project Report. W-17-4 and W-17-5. 22 pp.

Lentfer, J.W. 1983. Alaskan polar bear movements from mark and recovery. Arctic 36:282-288.

Lentfer, J.W., and W.A. Galster. 1987. Mercury in polar bears from Alaska. Journal of Wildlife Diseases 23:338-341.

Lentfer, J.W., R.J. Hensel, J.R. Gilbert, and F.E. Sorensen. 1980. Population characteristics of Alaskan polar bears. International Conference on Bear Research and Management 3: 109-115.

Lindsay, R.W., and J. Zhang. 2005. The thinning of the Arctic sea ice, 1988-2003: have we passed a tipping point? J. Climate 18:4879-4894.

Lunn, N.J., S. Schliebe, and E.W. Born, eds. 2002. Polar bears: Proceedings of the 13th working meeting of the IUCN/SSC Polar Bear Specialist Group. IUCN, Gland, Switzerland, and Cambridge, U.K. vii +153pp.

Manning, T.H. 1971. Geographical variation in the polar bear *Ursus maritimus* Phipps. Canadian Wildlife Service Report Series No. 13. 27 pp.

Muir, D.C.G., S. Backus, A.E. Derocher, R. Dietz, T.J. Evans, G.W. Gabrielsen, J. Nagy, R.J. Norstrom, C. Sonne, I. Stirling, M.K. Taylor, and R.J. Letcher. 2006. Brominated flame retardants in polar bears (*Ursus maritimus*) from Alaska, the Canadian Arctic, East Greenland and Svalbard. Environmental Science and Technology 40(2):449-455.

Nageak, B.P., C.D.N. Brower, and S.L. Schliebe. 1991. Polar bear management in the southern Beaufort Sea: An Agreement between the Inuvialuit Game Council and the North Slope Borough Fish and Game Committee. In Transactions of North American Wildlife and Natural Resources Conference 56:337-343.

Ovsyanikov, N.G. 2006. Research and conservation of polar bears on Wrangel Island. Pp. 167-171, In J. Aars, N.J. Lunn, and A.E. Derocher (eds.), Proceeding of the 14th Working Meeting of the IUCN/SSC Polar Bear Specialist Group, 20-24 June 2005, Seattle, Washington, U.S.A. IUCN, Gland, Switzerland, and Cambridge, U.K.

Overland, J.E., and M. Wang. 2007. Future regional Arctic sea ice declines. Geophysical Research Letters 34: L17705.

Paetkau, D., S.C. Amstrup, E.W. Born, W. Calvert, A.E. Derocher, G.W. Garner, F. Messier, I. Stirling, M.K. Taylor, Ø. Wiig, and C. Strobeck. 1999. Genetic Structure of the world's polar bear populations. Molecular Ecology 8:1571-1584.

Parkinson, C.L., D.J. Cavalieri, P. Gloersen, H.J. Zwally, and J.C. Comiso. 1999. Arctic sea ice extents, areas, and trends, 1978-1996. Journal of Geophysical Research 104(C9):20837-20856.

Ray, C.E. 1971. Polar bear and mammoth on the Pribilof Islands. Arctic 24:9-19.

Regehr, E.V., S.C. Amstrup, and I. Stirling. 2006. Polar bear population status in the Southern Beaufort Sea. Report Series 2006-1337, U.S. Department of the Interior, U.S. Geological Survey, Anchorage, Alaska. 55 pp.

Regehr, E.V., C.M. Hunter, H. Caswell, S.C. Amstrup, and I. Stirling. 2007. Polar bears in the Southern Beaufort Sea I: Survival and breeding in relation to sea ice conditions, 2001-2006. U.S. Geological Survey, Alaska Science Center, Administrative Report. 45pp.

Rode, K.D., S.C. Amstrup, and E.V. Regehr. 2007. Polar Bears in the southern Beaufort Sea III: stature, mass, and cub recruitment in relationship to time and sea ice extent between 1982 and 2006. U.S. Dept. of the Interior, U.S. Geological Survey Administrative Report, Reston, Virginia. 28 pp.

Rothrock, D.A., Y. Yu, and G.A. Maykut. 1999. Thinning of the Arctic sea-ice cover, Geophysical Research Letters 26:3469-3472.

Rush, S.A., K. Borga, R. Dietz, E.W. Born, C. Sonne, T. Evans, D.C.G. Muir, R.L. Letcher, R.J. Norstrom, and A.T. Fisk. 2008. Geographic Distribution of selected elements in the livers of polar bears from Greenland, Canada, and the United States. Environmental Pollution 153 (3): 618-626.

Schliebe, S.L., S.C. Amstrup, and G.W. Garner. 1995. The status of polar bear in Alaska, 1993. Pp. 125-139, In Ø. Wiig, G.W. Garner (eds.), Proceedings of the Eleventh Working Meeting of the IUCN/SSC Polar Bear Specialist Group, Copenhagen, Denmark. IUCN, Gland, Switzerland, and Cambridge, UK. v + 192 pp.

Schliebe, S., T.J. Evans, S. Miller, C. Perham, J. Wilder, and L.J. Lierheimer. 2006. Polar bear management in Alaska 2000-2004. Pp. 63-76, In J. Aars, N.J. Lunn, and A.E. Derocher (eds.), Proceeding of the 14th Working Meeting of the IUCN/SSC Polar Bear Specialist Group, 20-24 June 2005, Seattle, Washington, U.S.A. IUCN, Gland, Switzerland, and Cambridge, U.K.

Scribner, K.T., G.W. Garner, S.C. Amstrup, and M.A. Cronin. 1997. Population genetic studies of the polar bear (*Ursus maritimus*): a summary of available data and interpretation of results. Pp. 185-196, In Dizon, S., J. Chivers, and W. Perrin (eds.), Molecular genetics of marine mammals, incorporating the proceedings of a workshop on the analysis of genetic data to address problems of stock identity as related to management of marine mammals. Spec. Pub. #3 of the Society of Marine Mammalogy.

Serreze, M.C., M.M. Holland, and J. Stroeve. 2007. Perspectives on the Arctic's shrinking sea-ice cover. Science 315: 1533-1536.

Smithwick, M., J.W. Martin, S.A. Mabury, K. Solomon, C. Sonne, E.W. Born, R. Dietz, A.E. Derocher, R.L. Letcher, T.J. Evans, G.W. Gabrielsonm, J. Nagy, I. Stirling, and D.C.G. Muir. 2005. A circumpolar study of perfluoroalkyl contaminants in polar bears (*Ursus maritimus*). Environmental Science and Technology 39(15):5517-5523.

Smithwick, M.J., R.J. Norstrom, S.A. Maybury, K. Solomon, T.J. Evans, I. Stirling, M.K. Taylor, and D.C.G. Muir. 2006. Temporal trends of perfluoroalkyl contaminants in polar bears (*Ursus maritimus*) from two locations in the North American Arctic, 1972 -2002. Environmental Science and Technology. 40(4):1139-1143.

Stishov, M.S. 1991a. Results of aerial counts of the polar bear dens on the arctic coast of the extreme Northern Asia. Pp. 90-92, In Amstrup, S.C., and Wiig, Ø. (eds.), Polar Bears: Proceedings of the Tenth Working Meeting of the IUCN/SSC Polar Bear Specialist Group. IUCN, Gland, Switzerland, and Cambridge, U.K.

Stishov, M.S. 1991b. Distribution and number of polar bear maternity dens on Wrangel and Herald islands, in 1985-1989, pp. 91-115 in Amerirkhavov, A.M. (ed.), Population and Communities of Mammals on Wrangel Island. Moscow, CNIL Glavokhhoty RSFSR. (in Russian).

Stishov, M.S., G.W. Garner, S.M. Arthur, and V.G.B. Barnes Jr. 1991. Distribution and relative abundance of maternal polar bear dens in the Chukotka Peninsula region, U.S.S.R. p. 67 in Abstracts, Ninth Biennial Conference on the Biology of the Marine Mammals, 5-9 December 1991, Chicago, Illinois, U.S.A.

Stroeve, J., M. Serreze, S. Drobot, S. Gearheard, M. Holland, J. Maslanik, W. Meier, and T. Scambos. 2008. Arctic Sea Ice Extent Plummets in 2007. EOS, Transactions, American Geophysical Union 89(2):13-14.

Treseder, L., and A. Carpenter. 1989. Polar bear management in the Southern Beaufort Sea. Info. N. 15(4):2-4.

USFWS. Unpublished data (polar bear harvest information). Available from: USFWS Marine Mammals Office, 1011 East Tudor Road, Anchorage, AK 99503.

Uspenski, S.M. 1986. Research and management of polar bear populations in the USSR, 1981-85. Pages 133-136 in Proceedings of the Ninth Working Meeting of the IUCN/SSC Polar Bear Specialist Group, IUCN, Gland, Switzerland, and Cambridge, U.K.

Verrault, J., D.C.G. Muir, R.J. Norstrom, I. Stirling, A.T. Fisk, G.W. Gabrielsen, A.E. Derocher, T.J. Evans, R. Dietz, C. Sonne, G.M. Sandala, W. Gebbink, F.F. Riget, E.W. Born, M.K. Taylor, J. Nagy, and R.J. Letcher. 2005. Chlorinated hydrocarbon contaminants and metabolites in polar bears (*Ursus maritimus*) from Alaska, Canada, East Greenland, and Svalbard: 1996-2002. The Science of the Total Environment 351-352:369-390.

Wade, P.R., and R. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report in the GAMMS Workshop, April 3-5, 1996, Seattle, WA. 93 pp.

Wilson, D.E. 1976. Cranial variation in polar bears. International Conference Bear Research and Management 3:447-453.

Woshner, V.M., T.M. O'Hara, G.R. Bratton, and V.R. Beasley. 2001. Concentrations and interactions of selected essential and non-essential elements in ringed seals and polar bears of Arctic Alaska. Journal of Wildlife Diseases. (37):711-721.

Zdor, Eduard. Personal Communication. Executive Director, Association of Traditional Marine Mammal Hunters of Chukotka.

POLAR BEAR (*Ursus maritimus*): Southern Beaufort Sea Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Polar bears are circumpolar in their distribution in the northern hemisphere. They occur in several largely discrete stocks or populations (Harington 1968). Polar bear movements are extensive and individual activity areas are enormous (Garner et al. 1990, Amstrup et al. 2000). The parameters used by Dizon et al. (1992) to classify stocks based on the phylogeographic approach were considered in the determination of stock separation in Alaska. Several polar bear stocks are known to be shared between countries (Amstrup et al. 1986, Amstrup and Demaster 1988). Lentfer hypothesized that two Alaska stocks exist, the Southern Beaufort Sea, and the Chukchi/Bering Seas, based upon: (a) variations in levels of heavy metal contaminants of organ tissues (Lentfer 1976, Lentfer and Galster 1987); (b) morphological characteristics (Manning 1971; Lentfer 1974; Wilson 1976); (c) physical oceanographic features which segregate stocks (Lentfer 1974) and; (d) movement information collected from mark and recapture studies of adult female bears (Lentfer 1983) (Figure 1). Information on contaminants (Woshner et al. 2001, Evans 2004a, Evans 2004b, Kannan et al. 2005, Smithwick et al. 2005, Verreault et al. 2005, Muir et al. 2006, Smithwick et al. 2006, Kannan et al. 2007, Rush et al. 2008) and movement data using satellite collars (Amstrup et al. 2004, Amstrup et al. 2005) continue to support the existence of these two stocks.

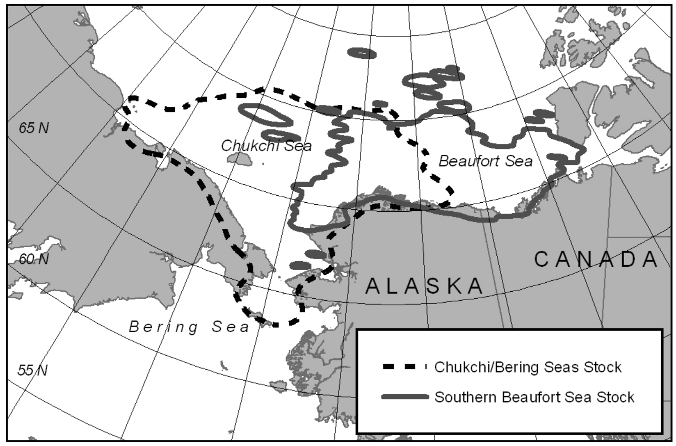

Figure 1. Map of the Southern Beaufort Sea and the Chukchi/ Bering seas polar bear stocks.

Amstrup et al. (2000) demonstrated that the eastern boundary of the Southern Beaufort Sea stock occurs south of Banks Island and east of the Baillie Islands, Canada. The bears in the Northern Beaufort Sea and Southern Beaufort Sea populations spend the summer on pack ice and move toward the coast during fall, winter, and spring (Durner et al. 2004). The range of the two populations previously overlapped extensively in the vicinity of the Baille Islands, Canada (Amstrup 2000) but recent data no longer support this degree of overlap (Amstrup et al. 2005). Recent analysis of polar bear movements using satellite telemetry from 2000 to 2006 (Amstrup et al. 2004, Amstrup et al. 2005), capture and recapture data (Regehr et al. 2006, Stirling et al. 2007), and harvest information suggest that the eastern population boundary has shifted westward to near the village of Tuktoyaktuk, Canada. The assignment of this new boundary could be adjusted somewhat based on local management considerations; however, it will probably necessitate a downward readjustment of the population size of the Southern Beaufort Sea stock to correspond with the smaller geographic area. The proposed boundary change is under consideration and has not been accepted by the parties to the Polar Bear Management Agreement for the Southern Beaufort Sea between the Inuvialuit Game Council of Canada and the North Slope Borough of Alaska. For the purposes of this report, we continue to use the previously published boundaries for the Southern Beaufort Sea population delineated by Amstrup et al. (2000). The western boundary is near Point Hope. An extensive area of overlap between the Southern Beaufort Sea stock and the Chukchi/Bering seas stock occurs between Point Barrow and Point Hope, centered near Point Lay (Garner et al. 1990, Garner et al. 1994, Amstrup et al. 2000). The southern boundary of the Northern Beaufort Sea stock in the Canadian Arctic was delineated by Bethke et al. (1996). Telemetry data indicates that adult female polar bears marked in the Southern Beaufort Sea spend about 25% of their time in the northeastern Chukchi Sea, whereas females captured in the Chukchi Sea spend only 6% of their time in the Southern Beaufort Sea (Amstrup 1995). However, polar bears are not dispersed evenly throughout their range. To access ringed and bearded seals, polar bears in the Southern Beaufort Sea concentrate in shallow waters less that 300 m deep over the continental shelf and in areas with >50% ice cover (Stirling et al. 1999, Durner et al. 2004, Durner et al. 2006a, Durner et al. 2009). Polar bears from this population have historically denned on both the sea ice and land. Thinning of the sea ice in recent years has caused a decline in the number of polar bears denning on the sea ice. Fischbach et al. (2007) found that the proportion of dens on the pack ice declined from 62% from 1985—1994 to 37% in 1998-2004. The main terrestrial denning areas for the Southern Beaufort Sea population in Alaska occur on the barrier islands from Barrow to Kaktovik and along coastal areas up to 25 miles inland including the Arctic National Wildlife Refuge to Peard Bay, west of Barrow (Amstrup and Gardner 1994, Amstrup 2000, Durner et al. 2001, Durner et al. 2006b).

In response to changes in the sea ice characteristics and declines in sea ice habitat over the continental shelf during the summer and late fall, some polar bears have changed distribution to search for seals and to access the remains of subsistence harvested bowhead whales (Schliebe et al. 2008). It is expected that changes in the distribution and movements may occur with increasing frequency in the future (Durner et al. 2009, Schliebe et al. 2008) Polar bears may also become more nutritionally stressed due to global climate changes in the Arctic (Stirling and Parkinson 2006) and, thus, continued monitoring is required to document these changes.

Analysis of mitochondrial DNA and microsatellite DNA loci indicates little differentiation of the Alaska polar bear stocks (Cronin et al. 1991, Scribner et al. 1997, Cronin et al. 2006). Using 16 highly variable micro satellite loci, Paetkau et al. (1999) determined that polar bears throughout the arctic (19 populations) were genetically very similar. Genetically, polar bears in the Southern Beaufort Sea differed more from polar bears in the Chukchi/Bering Seas than from polar bears in the Northern Beaufort Sea (Paetkau et al. 1999, Thiemann et al. 2008). While genetically similar, demographic and movement data indicates a high degree of site fidelity, suggesting that the stocks should be managed separately (Amstrup 2000, Amstrup et al. 2000, Amstrup et al. 2001a, Amstrup et al. 2002, Amstrup et al. 2004, Amstrup et al. 2005).

POPULATION SIZE

Polar bears occur at low densities throughout their circumpolar range (DeMaster and Stirling 1981). They are long lived, mature late, have an extended breeding interval, and have small litters (Lentfer et al. 1980, DeMaster and Stirling 1981, Amstrup 2003). Accurate population estimates for the Alaskan populations have been difficult to obtain because of low population densities, inaccessibility of the habitat, movement of bears across international boundaries, and budget limitations (Amstrup and DeMaster 1988, Garner et al. 1992). Research on the Southern Beaufort Sea population began in 1967 and is one of only four polar bear populations with long term (>20 yrs) data.

Amstrup et al. (1986) estimated the Southern Beaufort Sea stock at 1,778 (S.D. $\pm$ 803; C.V. = 0.45) during the 1972-83 period. Amstrup (1995) estimated the Southern Beaufort Sea stock near 1,480 animals in 1992. Amstrup (USGS unpublished data) using data for the 1986-98 period (excluding 4 unsampled years), estimated the population at 2,272 in 2001. This total population estimate was based on as estimate of 1,250 females (C.V. = 0.17) and a sex ratio of 55% females (Amstrup et al. 2001b). The population estimate of 1,526 (95% CI =1211−1841; C.V. = 0.106) (Regehr et al. 2006), which is based on open population capture-recapture data collected from 2001 to 2006, is considered the most current and valid population estimate.

Minimum Population Estimate

N_{MINn} is calculated as follows $N/\exp(0.842 * (\ln(1+CV(N)^2))^{½})$ and is 1,397 bears for population size of 1,526 and C.V. of 0.106. This population estimate applies to an area that extends from Pt. Barrow in the west, east to the Baillie Islands in Canada.

Current Population Trend

Prior to the 20th century, when Alaska's polar bears were hunted primarily by Natives, both the Chukchi/Bering seas and Southern Beaufort Sea stocks probably existed near carrying capacity (K). Once harvest by non-Natives became common in the Southern Beaufort Sea in the early 1960s, the size of these stocks declined substantially (Amstrup et al. 1986, Amstrup 1995). Since passage of the Marine Mammal Protection Act (MMPA) in 1972, both Alaska polar bear stocks seem to have increased; this is based on: (a) mark and recapture data; (b) observations by Natives and residents of coastal Alaska and Russia; (c) catch per unit effort indices (USGS unpublished data); (d) reports from Russian scientists (Uspenski and Belikov 1991); and (e) harvest statistics on the age structure of the population. Recapture data from the stock indicated a population growth rate of 2.4% from 1981 to 1992 (Amstrup 1995).

The Southern Beaufort Sea stock experienced little or no growth during the 1990's (Amstrup et al. 2001b). Declining survival, recruitment, and body size (Regehr et al. 2006, Regehr et al. 2007), and low growth rates (λ) during years of reduced sea ice during the summer and fall (2004 and 2005), and an overall declining growth rate of 3% per year from 2001-2005 (Hunter et al. 2007) indicates that the Southern Beaufort Sea population is now declining.

MAXIMUM NET PRODUCTIVITY RATES

Population/stock specific data to estimate R_{MAX} are not available for the stock. Taylor et al. (1987) estimated the sustainable yield of the female component of the population at < 1.6% per annum. The following information is used to understand the R_{MAX} determination. From 1981-92, when the population was increasing, vital rates of polar bears in the Southern Beaufort Sea were as follows: average age of sexual maturity (females) was 6 years; average COY litter size was 1.67; average reproductive interval was 3.68 years; and average annual natural mortality (nM), which varies by age class, ranged from 1-3% for adults (Amstrup 1995).

Amstrup (1995) projected an annual intrinsic growth rate (including natural mortality but not human-caused mortality) of 6.03% for the Southern Beaufort Sea stock using a Leslie type matrix of recapture data. This analysis mimics a life history scenario where environmental resistance is low and survival high.

POTENTIAL BIOLOGICAL REMOVAL (PBR)

Under the 1994 reauthorized MMPA, the potential biological removal (PBR) level is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: PBR = $(N_{MIN})(\frac{1}{2} R_{MAX})(F_R)$. Wade and Angliss (1997) recommend a default recovery factor (F_R) of 0.5 for a threatened population or when the status of a population is unknown. In the following calculation: $(N_{MIN})(\frac{1}{2} R_{MAX})(F_r)$ = PBR (Wade and Angliss 1997) the minimum population estimate, N_{MIN} was 1,397; the maximum rate of increase R_{MAX} was 6.03%; and the recovery factor F_R was 0.5. Therefore, the PBR level for the Southern Beaufort Sea stock is 22 bears per year.

ANNUAL HUMAN CAUSED MORTALITY AND SERIOUS INJURY

Fisheries Information

Polar bear stocks in Alaska have no direct interaction with commercial fisheries activities. Consequently, the total fishery mortality and serious injury rate for the Southern Beaufort Sea stock is zero.

Alaska Native Subsistence Harvest

Historically, polar bears have been killed for subsistence, handicrafts, and recreation (sport hunting). Based upon records of skins shipped from Alaska, the estimated annual statewide harvest (both stocks) for 1925−53 averaged 120 bears taken primarily by Native hunters. Sport hunting using aircraft was common from 1951−72, increasing annual harvest in Alaska to 150 during 1951-60 and to 260 during 1960−72 (Amstrup et al. 1986; Schliebe et al.1995). The annual harvest for the Southern Beaufort Sea stock was 81/year from 1960−1972. Although polar bear hunting was prohibited by the MMPA, an exemption was made for Alaska Natives living in coastal communities to allow them to hunt polar bears for subsistence and making of handicrafts provided that the hunt was not done in a wasteful manner. The cessation of sport hunting in 1972 reduced the mean annual combined harvest for both Alaskan stocks to 98 during 1980−2007 (SD=40; range 48−242) (USFWS unpublished data). The annual harvest from the Southern Beaufort Sea was 39/year in the 1980s, 33/year in the 1990s, and 32/year in the 2000s. More recently, the 2003−2007 average Alaska harvest for the Southern Beaufort Sea in Alaska was 33 and the sex ratio was 67M:33F. During the same time period the average Canadian harvest for the Southern Beaufort Sea was 21.0 and the sex ratio was 45M:55F. The combined average annual Alaska and Canada harvest during the past five years was 53.6. Figure 2 illustrates the annual Alaska polar bear harvest and trend for the Southern Beaufort Sea stock from 1961−2007. No serious injuries, other than the mortalities discussed here, have been reported for the Southern Beaufort Sea stock.

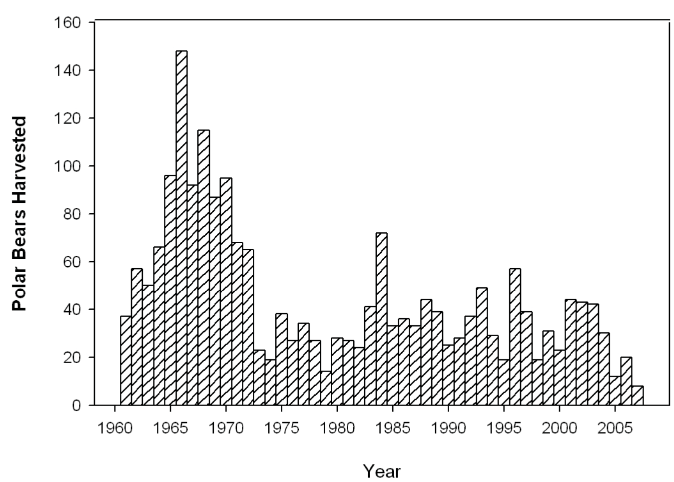

Figure 2. Annual Alaska polar bear harvest from the Southern Beaufort Sea stock, 1961-2007.

During the 1980−2007 period the Alaska harvest from the Southern Beaufort Sea accounted for 34% of the total Alaska kill (annual mean=33 bears) with the remaining 66% occurring in the Chukchi Sea. The sex ratio of the harvest from 1980−2007 in the Southern Beaufort Sea was 69M:31F.

Other Removals

Orphaned cubs are occasionally removed from the wild and placed in zoos; no cubs were placed into public display facilities during the past five years. One bear died as a result of research mortality and two bears were euthanized during the last five years. Activities operating under "incidental take" regulations, associated with the oil and gas industry, have the potential to impact polar bears and their habitat. During the past five years no lethal takes related to industrial activities of polar bears have occurred. Three lethal takes related to oil and gas activities have been documented in the Southern Beaufort Sea: one at an offshore drilling site in the Canadian Beaufort Sea (1968); one bear at the Stinson site in the Alaska Beaufort Sea (1990); and one bear that ingested ethylene glycol stored at an offshore island in the Alaska Beaufort Sea (1988). In 1993, a polar bear was killed at the Oliktok remote radar defense site when it broke into a residence and severely mauled a worker.

STATUS OF STOCK

The Southern Beaufort Sea Stock is currently classified as depleted under the MMPA and listed as threatened under the U.S. Endangered Species Act of 1973 (ESA), as amended. The primary concerns for this population are loss of the sea ice habitat due in part to climate changes in the Arctic, potential overharvest, and current and proposed human activities including industrial activities occurring in the nearshore and offshore environment. Recent data on the vital rates, population estimate, and growth rates for the Southern Beaufort Sea suggests that this population stock is declining. Because of its status as a threatened species under the ESA, the Southern Beaufort Sea population is designated as a strategic stock.

Conservation Issues and Habitat Concerns

Oil and Gas Exploration

The Minerals Management Service (MMS) (2004) estimated an 11 percent chance of a marine spill greater than 1,000 barrels in the Beaufort Sea from the Beaufort Sea Multiple Lease Sale in Alaska. Amstrup et al. (2006) evaluated the potential effects of a hypothetical 5,912-barrel oil spill (the largest spill thought possible from a pipeline spill) on polar bears from the Northstar offshore oil production facility in the southern Beaufort Sea, and found that there is a low probability that a large number of bears (i.e., 25–60) might be affected by such a spill. For the purposes of this scenario, it was assumed that a polar bear would die if it came in contact with the oil. Amstrup et al. (2006) found that 0–27 bears could potentially be oiled during the open water conditions in September; and from 0–74 bears in mixed ice conditions during October. If such a spill occurred, particularly during the broken ice period, the impact of the spill could be significant to the Southern Beaufort Sea polar bear population (Amstrup et al. 2006, 65 FR 16828; March 30, 2000). At the time that Amstrup did this analysis, the sustainable harvest yield per year for the Southern Beaufort Sea polar bear population, based on a stable population size of 1,800 bears, was estimated to be 81.1 bears (1999–2000 to 2003–2004) (Lunn et al. 2005). For the same time period, the average harvest was 58.2 bears, leaving an additional buffer of 23 bears that could have been removed from the population. Therefore, an oil spill that resulted in the death of greater than 23 bears, which was possible based on the range of oil spill-related mortalities from the previous analysis, could have had population level effects for polar bears in the southern Beaufort Sea. However, the harvest figure of 81 bears may no longer be sustainable for the Southern Beaufort Sea population so, given the average harvest rate cited above, fewer than 23 oil spill-related mortalities could result in a population decline or increase the time required for recovery.

The Fish and Wildlife Service (USFWS) works to monitor and mitigate potential impacts of oil and gas activities on polar bears through incidental take regulations (ITR) as authorized under the Marine Mammal Protection Act. Activities operating under these regulations must adopt measures to: ensure that the total taking of polar bears remains negligible; minimize impacts to their habitat; and ensure no unmitigable adverse impact on their availability for Alaska Native subsistence use. ITR also specify monitoring requirements that provide a basis for evaluating potential impacts of current and future activities on marine mammals.

Climate Change

Polar bears evolved over thousands of years to life in a sea ice environment. They depend on the sea ice-dominated ecosystem to support essential life functions. Sea ice provides a platform for hunting and feeding, for seeking mates and breeding, for movement to terrestrial maternity denning areas and occasionally for maternity denning, for resting, and for long-distance movements. The sea ice ecosystem supports ringed seals, the primary prey for polar bears, and other marine mammals that are also part of their prey base.

Sea ice is rapidly diminishing throughout the Arctic and large declines in optimal polar bear habitat have occurred in the Southern Beaufort and Chukchi Seas between the two time periods, 1985–1995 and 1996–2006 (Durner et al. 2009). In addition, it is predicted that the greatest declines in 21st century optimal polar bear habitat will occur in Chukchi and Southern Beaufort Seas (Durner et al. 2009). Patterns of increased temperatures, earlier onset of and longer melting periods, later onset of freeze-up, increased rain-on-snow events, and potential reductions in snowfall are occurring. In addition, positive feedback systems (i.e., the sea-ice albedo feedback mechanism) and naturally occurring events, such as warm water intrusion into the Arctic and changing atmospheric wind patterns, can operate to amplify the effects of these phenomena. As a result, there is fragmentation of sea ice, a dramatic increase in the extent of open water areas seasonally, reduction in the extent and area of sea ice in all seasons, retraction of sea ice away from productive continental shelf areas throughout the polar basin, reduction of the amount of heavier and more stable multi-year ice, and declining thickness and quality of shore-fast ice (Parkinson et al 1999, Rothrock et al. 1999, Comiso 2003, Fowler et al. 2004, Lindsay and Zhang 2005, Holland et al. 2006, Comiso 2006, Serreze et al. 2007, Stroeve et al. 2008).

The Chukchi/Bering Seas and the Southern Beaufort Sea population stocks are currently experiencing the initial effects of changes in sea ice conditions (Rode et al. 2007, Regehr et al. 2007, Hunter et al. 2007). These populations are vulnerable to large-scale dramatic seasonal fluctuations in ice movements, decreased abundance and access to prey, and increased energetic costs of hunting. The USFWS is working on measures to protect polar bears and their habitat.

Subsistence Harvest

Recognition that the polar bears in the southern Beaufort Sea were shared between Canada and the Alaska led to the development of the Polar Bear Management Agreement for the Southern Beaufort Sea between the Inuvialuit of the Inuvialuit Game Council (IGC), Canada and the Inupiat of the North Slope Borough (NSB) Alaska in 1988 (Nageak et al. 1991, Treseder and Carpenter 1989, Prestrud and Stirling 1994, Brower et al. 2002). Since initiation of this local user agreement in 1988, the combined Alaska/Canada mean harvest from this stock has been 56.9 bears per year (1988-2007). The harvest in Canada is limited primarily to Native hunters and is regulated by a quota system (Prestrud and Stirling 1994, Brower et al. 2002). Canada has a well regulated and controlled harvest, which has resulted in accurate harvest reporting, strict controls on the harvest, and efficient monitoring and enforcement. The harvest management system in Alaska is voluntary and is less efficient overall than the Canadian system (Brower et al 2002).

The calculation of a PBR level for the Southern Beaufort Sea stock is required by the MMPA even though the subsistence harvest quota is managed under the authority of the Polar Bear Agreement between the NSB and the IGC. Accordingly, the quota from the Board of Commissioners for the Polar Bear Agreement takes precedence over the PBR estimate for the purposes of managing the Alaska Native subsistence harvest from this stock. The Southern Beaufort Sea population is currently thought to be declining; therefore, overharvest could hasten the decline or prevent and/or slow the recovery. Analysis is currently underway to evaluate the effects of different harvest levels on the population dynamics of the Southern Beaufort Sea population.

CITATIONS

Amstrup, S.C. 1995. Movements, distribution, and population dynamics of polar bears in the Beaufort Sea. Ph.D. Dissertation. University of Alaska-Fairbanks, Fairbanks, Alaska, 299 pp.

Amstrup, S.C., 2000. Polar Bear. In The Natural History of an Oil Field: Development and Biota. Edited by J.C. Truett and S.R. Johnson. Academic Press, Inc., New York, New York, USA pp. 133-157.

Amstrup, S.C. 2003. Polar Bear (Ursus maritimus). Pages 587-610 in Feldhammer, G.A., B.C. Thompson, and J.A. Chapman, eds., Wild Mammals of North America — Biology, Management, and Conservation. John Hopkins University Press. Baltimore, Maryland. pp. 587-610.

Amstrup, S.C., and D.P. DeMaster. 1988. Polar bear, Ursus maritimus. Pages 39-45 in J.W. Lentfer, ed., Selected Marine Mammals of Alaska: Species Accounts with Research and Management Recommendations. Marine Mammal Commission, Washington, D.C.

Amstrup, S., and C. Gardner. 1994. Polar bear maternity denning in the Beaufort Sea. Journal of Wildlife Management 58(1):1-10.

Amstrup, S.C., I. Stirling, and J.W. Lentfer. 1986. Past and present status of polar bears in Alaska. Wildlife Society Bulletin. 14:241-254.

Amstrup, S.C., G. Durner, I. Stirling, N.J. Lunn, and F. Messier. 2000. Movements and distribution of polar bears in the Beaufort Sea. Canadian Journal of Zoology. 78:948-966.

Amstrup, S.C., G.M. Durner, T.L. McDonald, D.M. Mulcahy, and G.W. Garner. 2001a. Comparing movement patterns of satellite-tagged male and female polar bears. Canadian Journal of Zoology. 79:2147-2158.

Amstrup, S.C., T.L. McDonald, and I. Stirling. 2001b. Polar bears in the Beaufort Sea: A 30-year mark-recapture case history. Journal of Agricultural, Biological, and Environmental Statistics, Vol. 6(2): 221-234.

Amstrup, S.C., G. M. Durner, A. S. Fischbach, K. Simac, and G. Weston-York. 2002. Polar Bear Research in the Beaufort Sea. in N. Lunn, E. W. Born, and S. Schliebe (eds.), Proceedings of the Thirteenth Working Meeting of the IUCN/SSC Polar Bear Specialist Group. 23-28 June, 2001, Nuuk, Greenland. IUCN, Gland, Switzerland, and Cambridge, UK. vii + 153 pp.

Amstrup, S.C., T.L. McDonald, and G. Durner. 2004. Using satellite radiotelemetry data to delineate and manage wildlife populations. Wildlife Society Bulletin. 32(3):661-679.

Amstrup, S.C., G.M. Durner, I. Stirling, and T.L. McDonald. 2005. Allocating harvests among polar bear stocks in the Beaufort Sea. Arctic. 58:247-259.

Amstrup, S.C., G.M. Durner, T.L. McDonald, and W.R. Johnson. 2006. Estimating potential effects of hypothetical oil spills on polar bears. Unpublished Report. U.S. Geological Survey, Alaska Science Center, Anchorage, Alaska. 56 pp.

Bethke, R., M. Taylor, S. Amstrup, and F. Messier. 1996. Population delineation of polar bears using satellite collar data. Ecological Applications 6(1):311-317.

Brower, C. D., A. Carpenter, M. Branigan, W. Calvert, T. Evans, A. Fischbach, J. Nagy, S. Schliebe, and I. Stirling. 2002. The polar bear management agreement for the southern Beaufort Sea: An evaluation of the first ten years of a unique conservation agreement. Submitted to: Arctic 56:362-372.

Comiso, J.C. 2003. Warming trends in the Arctic from clear sky satellite observations. Journal of Climate 16:3498-3510.

Comiso, J.C. 2006. Arctic warming signals from satellite observations, Weather 61(3):70-76.

Cronin, M.A., S.C. Amstrup, G.W. Garner, and E.R. Vyse. 1991. Interspecific and intraspecific mitochondrial DNA variation in North American bears (Ursus). Canadian Journal of Zoology. 69:12:2985-2992.

Cronin, M.A., S.C. Amstrup, and K.T. Scribner. 2006. Microsatellite DNA and mitochondrial DNA variation in polar bears (Ursus maritimus) in the Beaufort and Chukchi seas, Alaska. Canadian Journal of Zoology 84:655-660.

DeMaster, D.P., and I. Stirling. 1981. Ursus maritimus. Mammalian Species:1-7.

Dizon, A.E., C. Lockyer, W.F. Perrin, D.P. DeMaster, and J. Sisson. 1992. Rethinking the stock concept: a phylogeographic approach. Conservation Biology. 6:24-36.

Durner, G.M., S.C. Amstrup, and K.J. Ambrosius. 2001. Remote identification of polar bear maternal den habitat in Northern Alaska. Arctic 54(2):115-121.

Durner, G.M., S.C. Amstrup, R. Nielson, and T. McDonald. 2004. The use of sea ice habitat by female polar bears in the Beaufort Sea. U.S. Geological Survey, Alaska Science Center, Anchorage, Alaska. OCS study, MMS 2004-014. 41 pp.

Durner, G.M., D.C. Douglas, R.M. Nielson, and S.C. Amstrup. 2006a. A model for autumn pelagic distribution of adult female polar bears in the Chukchi Sea, 1987-1994. USGS Science Center, Anchorage, Alaska. Contract Completion Report 70181-5-N240. 67 pp.

Durner, G.M., S.C. Amstrup, and K.J. Ambrosius. 2006b. Polar bear maternal den habitat in the Arctic national Wildlife refuge, Alaska. Arctic 59(1):31-36.

Durner, G.M., D.C. Douglas, R.M. Nielson, S.C. Amstrup, T.L. McDonald, I Stirling, M. Mauritzen, E.W. Born, Ø. Wiig, E. DeWeaver, M.C. Serreze, S.E. Belikov, M.M. Holland, J. Maslanik, J. Aars, D.C. Bailey, and A.E. Derocher. 2009. Predicting 21st century polar bear habitat distribution from global climate models. Ecological Monographs 79(1): 25-58.

Evans, T. J. 2004a. Concentrations of selected essential and non-essential elements in adult male polar bears (Ursus maritimus) from Alaska. U.S. Fish and Wildlife Service Technical Report. MMM 04-02. 68pp.

Evans, T. J. 2004b. PCBs and chlorinated pesticides in adult male polar bears (Ursus maritimus) from Alaska. U.S. Fish and Wildlife Service Technical Report. MMM 04-01. 61pp.

Fischbach, A.S., S.C. Amstrup, and D.C. Douglas. 2007. Landward and eastward shift of Alaskan polar bear denning associated with recent sea ice changes. Polar Biology 30:1395-1405: doi.1007/s00300-007-0300-4.

Fowler, C., W.J. Emery, and J. Maslanik. 2004. Satellite-derived evolution of Arctic sea ice age: October 1978 to March 2003. Geoscience and Remote Sensing Letters, IEEE, Volume 1, Issue 2, April 2004 pp.71–74.

Garner, G.W., S.T. Knick, and D.C. Douglas. 1990. Seasonal movements of adult female polar bears in the Bering and Chukchi seas. International Conference on Bear Research and Management 8:219-226.

Garner, G.W., L.L. McDonald, D.S. Robson, D.P. Young Jr., and S.M. Arthur. 1992. Literature review: population estimation methodologies applicable to the estimation of abundance of polar bears. Internal Report, U.S.FWS. 102pp.

Garner, G.W., L.L. McDonald, S.M. Arthur, and T.L. Olson. 1994. Operating procedures: Pilot polar bear survey Beaufort Sea: 1994. Internal Report, U.S.FWS, 39 pp.

Harington, C.R. 1968. Denning habits of the polar bear (Ursus maritimus Phipps). Canadian Wildlife Service Report, Series 5. 33 pp.

Holland, M., C.M. Bitz, and B. Tremblay. 2006. Future abrupt reductions in summer Arctic sea ice. Geophysical Research Letters 33 L25503: doi 10.1029/200661028024: 1-5.

Hunter, C.M., H. Caswell, M.C. Runge, E.V. Regehr, S.C. Amstrup, and I. Stirling. 2007. Polar bears in the Southern Beaufort Sea II: Demography and population growth in relation to sea ice conditions. U.S. Geological Survey, Alaska Science Center, Administrative Report. 46 pp.

Kannan, K., S.H. Yun, and T.J. Evans. 2005. Chlorinated, brominated, and perfluorinated contaminants in livers of polar bears from Alaska. Environmental Science and Technology. 39:9057-9063.

Kannan, K., T. Agusa, T.J. Evans, and S. Tanabe. 2007. Trace element concentrations in livers of polar bears from tow population in northern and western Alaska. Archives of Environmental Contaminants and Toxicology 53:473-482.

Lentfer, J.W. 1974. Discreteness of Alaskan polar bear populations. Proceedings of the International Congress of Game Biologists 11:323-329.

Lentfer, J.W. 1976. Environmental contaminants and parasites in polar bears. Alaska Department of Fish and Game, Pittman-Robertson Project Report. W-17-4 and W-17-5. 22 pp.

Lentfer, J.W. 1983. Alaskan polar bear movements from mark and recovery. Arctic 36:282-288.

Lentfer, J.W., R.J. Hensel, J.R. Gilbert, and F.E. Sorensen. 1980. Population characteristics of Alaskan polar bears. International Conference on Bear Research and Management 3: 109-115.

Lentfer, J.W., and W.A. Galster. 1987. Mercury in polar bears from Alaska. J. Wildlife Diseases 23:338-341.

Lindsay, R.W., and J. Zhang. 2005. The thinning of the Arctic sea ice, 1988-2003: have we passed a tipping point? J. Climate 18: 4879-4894.

Lunn, N.J., M. Branigan, L. Carpenter, K. Chaulk, B. Doidge, J. Galipeau, D. Hedman, M. Huot, R. Maraj, M. Obbard, R. Otto, I. Stirling, M. Taylor, and S. Woodley. 2005. Polar bear management in Canada 2001-2004. Pages 101-116 in Jon Aars, N.J. Lunn, and A.E. Derocher (eds). Proceedings of the 14th working meeting of the IUCN/SSC Polar Bear Specialist Group, June 2005, Seattle, Washington, U.S.A. 22pp.

Manning, T.H. 1971. Geographical variation in the polar bear Ursus maritimus Phipps. Canadian Wildlife Service Report Series No. 13. 27 pp.

Minerals Management Service (MMS). 2004. Proposed oil and gas lease sale 195, Beaufort Sea Planning Area Environmental Assessment. U.S. Department of the Interior, MMS, Anchorage, Alaska, USA. 101 pp. + Appendices.

Muir, D.C.G., S. Backus, A.E. Derocher, R. Dietz, T.J. Evans, G.W. Gabrielsen, J. Nagy, R.J. Norstrom, C. Sonne, I. Stirling, M.K. Taylor, and R.J. Letcher. 2006. Brominated flame retardants in polar bears (Ursus maritimus) from Alaska, the Canadian Arctic, East Greenland and Svalbard. Environmental Science and Technology 40(2):449-455.

Nageak, B.P., C.D.N. Brower, and S.L. Schliebe. 1991. Polar bear management in the southern Beaufort Sea: An Agreement between the Inuvialuit Game Council and the North Slope Borough Fish and Game Committee. In Transactions of North American Wildlife and Natural Resources Conference. 56:337-343.

Paetkau, D., S.C. Amstrup, E.W. Born, W. Calvert, A.E. Derocher, G.W. Garner, F. Messier, I. Stirling, M.K. Taylor, Ø. Wiig, and C. Strobeck. 1999. Genetic Structure of the world's polar bear populations. Molecular Ecology. 8:1571-1584.

Parkinson, C.L., D.J. Cavalieri, P. Gloersen, H.J. Zwally, and J.C. Comiso. 1999. Arctic sea ice extents, areas, and trends, 1978-1996. Journal of Geophysical Research 104(C9):20837-20856.

Prestrud, P., and I. Stirling. 1994. The international polar bear agreement and the current status of polar bear conservation. Aquatic Mammals 20:113-124.

Regehr, E.V., S.C. Amstrup, and I. Stirling. 2006. Polar bear population status in the southern Beaufort Sea. U.S. Geological Survey Open File Report 2006-1337. 20 pp.
Regehr, E.V., C.M. Hunter, H. Caswell, S.C. Amstrup, and I. Stirling. 2007. Polar bears in the Southern Beaufort Sea I: Survival and breeding in relation to sea ice conditions, 2001-2006. U.S. Geological Survey, Alaska Science Center, Administrative Report. 45pp.
Rode, K.D., S.C. Amstrup, and E.V. Regehr. 2007. Polar Bears in the southern Beaufort Sea III: stature, mass, and cub recruitment in relationship to time and sea ice extent between 1982 and 2006. U.S. Dept. of the Interior, U.S. Geological Survey Administrative Report, Reston, Virginia. 28 pp.
Rothrock, D.A., Y. Yu, and G.A. Maykut. 1999. Thinning of the Arctic sea-ice cover, Geophysical Research Letters 26: 3469-3472.
Rush, S.A., K. Borga, R. Dietz, E.W. Born, C. Sonne, T. Evans, D.C.G. Muir, R.L. Letcher, R.J. Norstrom, and A.T. Fisk. 2008. Geographic Distribution of selected elements in the livers of polar bears from Greenland, Canada, and the United States. Environmental Pollution 153 (3): 618-626.
Schliebe, S., K.D. Rode, J.S. Gleason, J. Wilder, K. Proffitt, T.J. Evans, and S. Miller. 2008. Effects of sea ice extent and food availability on spatial and temporal distribution of polar bears during the fall open-water period in the Southern Beaufort Sea. Polar Biology. 12 pp.
Schliebe, S.L., S.C. Amstrup, and G.W. Garner. 1995. The status of polar bear in Alaska, 1993. In Ø. Wiig, G.W. Garner (eds.), Proceedings of the Eleventh Working Meeting of the IUCN/SSC Polar Bear Specialist Group. IUCN, Gland, Switzerland, and Cambridge, UK. v + 192 pp.
Scribner, K.T., G.W. Garner, S.C. Amstrup, and M.A. Cronin. 1997. Population genetic studies of the Polar Bear (Ursus maritimus): a summary of available data and interpretation of results. Pp. 185-196, In Dizon, S., J. Chivers, and W. Perrin (eds.), Molecular genetics of marine mammals, incorporating the proceedings of a workshop on the analysis of genetic data to address problems of stock identity as related to management of marine mammals. Spec. Pub. #3 of the Society of Marine Mammalogy.
Serreze, M.C., M.M. Holland, and J. Stroeve. 2007. Perspectives on the Arctic's shrinking sea-ice cover. Science 315: 1533-1536.
Smithwick, M., J.W. Martin, S.A. Mabury, K. Solomon, C. Sonne, E.W. Born, R. Dietz, A.E. Derocher, R.L. Letcher, T.J. Evans, G.W. Gabrielsonm, J. Nagy, I. Stirling, and D.C.G. Muir. 2005. A circumpolar study of perfluoroalkyl contaminants in polar bears (Ursus maritimus). Environmental Science and Technology 39 (15): 5517-5523.
Smithwick, M. J., R. J. Norstrom, S.A. Maybury, K. Solomon, T.J. Evans, I. Stirling, M.K. Taylor, and D.C.G. Muir. 2006. Temporal trends of perfluoroalkyl contaminants in polar bears (Ursus maritimus) from two locations in the North American Arctic, 1972 -2002. Environmental Science and Technology. 40 (4): 1139-1143.
Stirling, I., and C.L. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (Ursus maritimus) in the Canadian Arctic. Arctic 99(3): 261-275.
Stirling, I., N.J. Lunn, and J. Iacozza. 1999. long-term trends in the population ecology of polar bears in western Hudson Bay in relation to climate change. Arctic 52:294306.
Stirling, I, T.L. McDonald, E.S. Richardson, and E.V. Regehr. 2007. Polar bear population status in the Northern Beaufort Sea. U.S. Geological Survey, Alaska Science Center, Administrative Report. 31 pp.
Stroeve, J., M. Serreze, S. Drobot, S. Gearheard, M. Holland, J. Maslanik, W. Meier, and T. Scambos. 2008. Arctic Sea Ice Extent Plummets in 2007. EOS, Transactions, American Geophysical Union 89(2):13-14.
Taylor, M.K., D.P. DeMaster, F.L. Bunnell, and R.E. Schweinsburg. 1987. Modeling the sustainable harvest of female polar bears. J. of Wildlife Management. 51:811-820.
Thiemann, G.W., A.E. Derocher, and I. Stirling. 2008. Polar bear Ursus maritimus conservation in Canada: an ecological basis for identifying designatable units. Oryx. 42(4): 504-515
Treseder, L., and A. Carpenter. 1989. Polar bear management in the Southern Beaufort Sea. Info. N. 15(4):2-4.
USFWS. Unpublished data (polar bear harvest information). Available from: USFWS Marine Mammals Office, 1011 East Tudor Road, Anchorage, AK 99503.
USGS (U.S. Geological Service) Unpublished data (research data). Available from: USGS/Alaska Science Center, 4210 University Drive, Anchorage, AK 99508.
Uspenski, S.M., and S.E. Belikov. 1991. Polar bear populations in the Arctic: Current state, studies, and management (1985-87). In S.C. Amstrup and Ø. Wiig (eds.), Proceedings of the Tenth Working Meeting of the IUCN/SSC Polar Bear Specialist Group, IUCN, Gland, Switzerland, and Cambridge, U.K.

Verrault, J., D.C.G. Muir, R.J. Norstrom, I. Stirling, A.T. Fisk, G.W. Gabrielsen, A.E. Derocher, T.J. Evans, R. Dietz, C. Sonne, G.M. Sandala, W. Gebbink, F.F. Riget, E.W. Born, M.K. Taylor, J. Nagy, and R. J. Letcher. 2005. Chlorinated hydrocarbon contaminants and metabolites in polar bears (Ursus maritimus) from Alaska, Canada, East Greenland, and Svalbard: 1996-2002. The Science of the Total Environment 351-352:369-390.

Wade, P.R., and R. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report in the GAMMS Workshop, April 3-5, 1996, Seattle, WA. 93pp.

Wilson, D.E. 1976. Cranial variation in polar bears. International Conference Bear Research and Management 3:447-453.

Woshner, V.M., T.M. O'Hara, G.R. Bratton, and V.R. Beasley. 2001. Concentrations and interactions of selected essential and non-essential elements in ringed seals and polar bears of Arctic Alaska. Journal of Wildlife Diseases. (37):711-721.

Revised: April 2014

PACIFIC WALRUS (*Odobenus rosmarus divergens*): Alaska Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

The family Odobenidae is represented by a single modern species, *Odobenus rosmarus,* of which two subspecies are generally recognized: the Atlantic walrus (*O. r. rosmarus*) and the Pacific walrus (*O. r. divergens*). The two subspecies occur in geographically isolated populations. The Pacific walrus is the only stock occurring in United States waters and considered in this account.

Pacific walruses range throughout the continental shelf waters of the Bering and Chukchi Seas, occasionally moving into the East Siberian Sea and the Beaufort Sea (Figure 1). During the summer months most of the population migrates into the Chukchi Sea; however, several thousand animals, primarily adult males, aggregate near coastal haulouts in the Gulf of Anadyr, Russia; Bering Strait, and Bristol Bay, Alaska. During the winter breeding season walruses are found in three concentration areas of the Bering Sea where open leads, polynyas, or thin ice occur (Fay *et al.* 1984, Garlich-Miller *et al.* 2011a). While the specific location of these groups varies annually and seasonally depending upon the extent of the sea ice, generally one group occurs near the Gulf of Anadyr, another south of St. Lawrence Island, and a third in the southeastern Bering Sea south of Nunivak Island into northwestern Bristol Bay. However, Pacific walruses are currently managed as a single panmictic population. Scribner *et al.* (1997) found no difference in mitochondrial and nuclear DNA among walruses sampled shortly after the breeding season from four areas of the Bering Sea (Gulf of Anadyr, Koryak Coast, Southeast Bering Sea, and St. Lawrence Island).

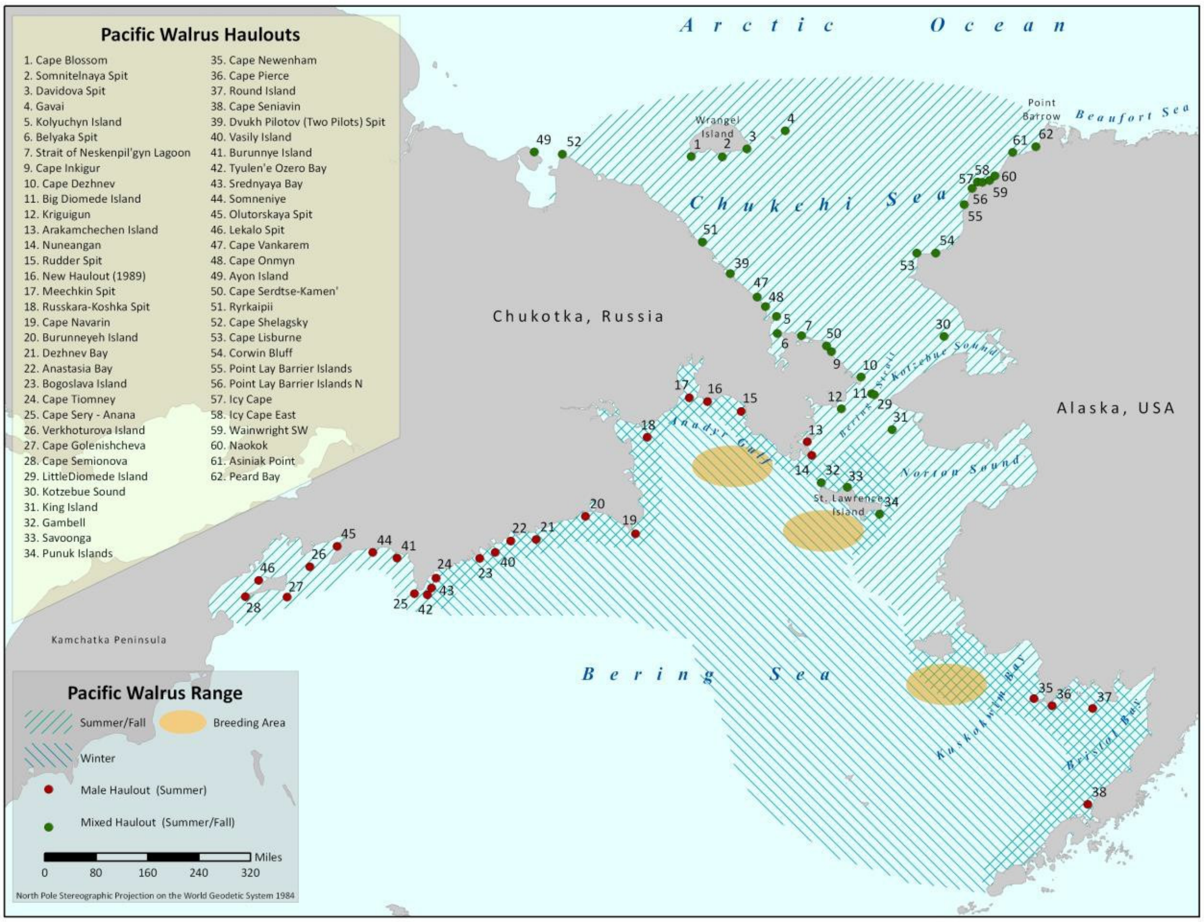

Figure 1. Seasonal distribution, breeding areas, and coastal haulouts of Pacific walruses in the Bering and Chukchi Seas. Modified from Smith 2010.

Pacific walruses typically use sea-ice as a resting platform between feeding dives, as a birthing substrate, for shelter from storms, isolation from predators, and passive transportation (Fay 1982). Historically, the summer distribution of walruses in the Chukchi Sea occurred primarily on sea ice over the continental shelf from the Alaska to Chukotka coasts with large numbers of animals near Hanna Shoal in the United States and Wrangel Island in the Russian Federation. A few animals would be observed utilizing haulouts along both the Alaska and Chukotka coasts, particularly in the fall. While the overall geographic range of Pacific walruses has not changed, over the past decade the number of walruses coming to shore along the coastline of the Chukchi Sea in both Alaska and Chukotka has increased from the hundreds to thousands to greater than 100,000 (Kavry *et al.* 2008, Garlich-Miller *et al.* 2011*a*, Jay *et al.* 2011). Additionally, adult female and young walruses are arriving at these coastal haulouts as much as a month earlier and staying at the coastal haulouts a week or two longer. In fall 2007, 2009, 2010, and 2011 large walrus aggregations (3,000 to 20,000) were observed along the Alaska coast (Garlich-Miller *et al*. 2011*a*). This increased use of coastal haulouts is a function of the loss of summer sea ice over the continental shelf (Garlich-Miller *et al.* 2011*a*). Summer sea-ice extent in the Chukchi Sea has decreased by about 12% per decade (NSIDC 2012); retreating off the shallow continental shelf and remaining only over deep Arctic Ocean waters where walruses cannot reach the benthos to feed. Declines in Chukchi Sea ice extent, duration, and thickness are projected to continue in a linear fashion into the foreseeable future (Douglas 2010).

POPULATION SIZE

The size of the Pacific walrus population has never been known with certainty. Based on large sustained harvests in the 18th and 19th centuries, Fay (1982) speculated that the pre-exploitation population was represented by a minimum of 200,000 animals. Since that time, population size has fluctuated markedly in response to varying levels of human exploitation (Fay *et al*. 1989). Large-scale commercial harvests reduced the population to 50,000 to 100,000 animals in the mid-1950s (Fay *et al.* 1997). The population is believed to have increased rapidly in size during the 1960s and 1970s in response to reductions in hunting pressure (Fay *et al.* 1989).

Between 1975 and 1990, aerial surveys were carried out by the United States and Russia at five-year intervals, producing mean population estimates ranging from 201,039 to 234,020 animals with 95% confidence intervals that include zero (Table 1). The estimates generated from these surveys are considered minimum values and because of the large associated variances they are not suitable for detecting population trends (Hills and Gilbert 1994, Gilbert *et al.* 1992). Further, these earlier figures largely underestimate the population because they were not adjusted for walruses in the water, a proportion of the population that may be as high as 65 to 87 percent (Born and Knutsen 1997, Gjertz *et al*. 2001, Jay *et al*. 2001, Born *et al*. 2005, Acquarone *et al*. 2006, Lydersen *et al*. 2008) and, because walrus tend to aggregate in large closely packed groups when hauled out on ice or land, it was difficult to obtain accurate counts of animals observed. Efforts to survey the Pacific walrus population were suspended at that time due to unresolved problems with survey methods, which produced population estimates with unknown bias and unknown or large variances that severely limited their utility (Gilbert *et al.* 1992, Gilbert 1999).

An international workshop on walrus survey methods in 2000 concluded that it would not be possible to obtain a population estimate with adequate precision for tracking trends using the existing aerial survey methods and any feasible amount of survey effort (Garlich-Miller and Jay 2000). Two major problems were identified: (1) accurately counting walruses in large groups, and (2) accounting for walruses in the water that were not available to be counted. Remote sensing systems were viewed as having great potential to address the first problem (Udevitz *et al.* 2001) as well as being able to sample larger areas per unit of time (Burn *et al*. 2006). To address the second problem U.S. Geological Survey (USGS) scientists developed satellite transmitters that recorded the haul-out status (in water or out) of individual walruses, which was used to estimate the proportion of animals in the water and correct walrus counts (Udevitz *et al.* 2009). These technological advances led to a joint United States-Russian Federation survey in March and April of 2006. This survey effort was timed to occur when the majority of Pacific walrus were hauled out on sea ice habitats across the continental shelf of the Bering Sea in order to capture as much of the population as possible.

The goal of the 2006 survey was to estimate the size of the Pacific walrus population (Speckman *et al.* 2011). However, some areas known to be important to walruses were not surveyed in 2006 because of poor weather and therefore the 2006 estimate is also considered to be an underestimate. The number of Pacific walruses within the area surveyed in 2006 was estimated at 129,000 with a 95% confidence interval of 55,000 to 507,000 (Speckman *et al.* 2011).

Table 1. Point estimates (95% confidence interval) of Pacific walrus population size, 1975-2006 from cooperative United States – Russian aerial surveys and original references.

Year	Population Estimate	References
1975	221,350 (–20,000-480,000)[a]	Gol'tsev 1976, Estes and Gilbert 1978, Estes and Gol'tsev 1984
1980	246,360 (–20,000-540,000)[a]	Johnson *et al.* 1982, Fedoseev 1984
1985	234,020 (–20,000-510,000)[a]	Gilbert 1986, 1989a, 1989b; Fedoseev and Razlivalov 1986
1990	201,039 (–19,000-460,000)[a]	Gilbert *et al.* 1992
2006	129,000 (55,000-507,000)	Speckman *et al*. 2011

[a]95% confidence intervals are from Figure 1 in Hills and Gilbert (1994).

Minimum Population Estimate

Under section 3(27) of the Marine Mammal Protection Act (MMPA), a "minimum population estimate" is defined as "an estimate of the number of animals in a stock that (A) is based on the best available scientific information on abundance, incorporating the precision and variability associated with such information and (B) provides reasonable assurance that the stock size is equal to or greater than the estimate." The estimate derived from the joint United States-Russian Federation survey conducted in March and April 2006 (Speckman *et al*. 2011) represents the minimum population estimate for the Pacific walrus. Because the 2006 survey used the most advanced technologies developed to address the problems identified in earlier aerial survey methods and was timed to capture as much of the population as possible (see above discussion under **POPULATION SIZE**), the survey's estimate of 129,000 individuals, with a 95%

confidence interval of 55,000 to 507,000 (Speckman *et al*. 2011), constitutes the best available scientific information on the size of the Pacific walrus population, taking into account the precision and variability associated with such estimates on abundance. The estimate from the 2006 survey is also negatively biased (Speckman *et al*. 2011), which provides reasonable assurance that the walrus population size is greater.

Current Population Trend

The 2006 estimate is lower than previous estimates of Pacific walrus population size (Table 1) and is known to be biased low to an unknown degree (Garlich-Miller *et al.* 2011*a*). However, estimates of population size from 1975 to the present (Table 1) are not directly comparable (Fay *et al.* 1997, Gilbert 1999) because of differences in survey methods, timing of surveys, and segments of the population surveyed. Therefore, while these estimates do not provide a good basis for inference with respect to population trends, there is other evidence supporting the hypothesis that the Pacific walrus population has declined from a peak in the late 1970s and 1980s.

Walrus researchers in the 1970s and 1980s were concerned that the population had reached or exceeded carrying capacity, and predicted that density-dependent mechanisms would begin to cause a decrease in population size (Fay and Stoker 1982b, Fay *et al*. 1986, Sease 1986, Fay *et al*. 1989). Estimates of demographic parameters from the late 1970s and 1980s support the idea that population growth was slowing (Fay and Stoker 1982a, Fay *et al*. 1986, Fay *et al*. 1989). Garlich-Miller *et al*. (2006) found that the median age of first reproduction for female walruses decreased in the 1990s, which is consistent with a reduction in density-dependent pressures. In addition, data on calf/cow ratios collected from harvested animals is consistent with a population peak in the late 1970s (i.e., low estimates in the late 1970s and 1980s) and subsequent population decline, and indicates that the population is currently below carrying capacity (MacCracken 2012).

The current working hypothesis, based on the available data, is that commercial and subsistence harvests prior to the 1960s limited the population; adoption of harvest quotas in the 1960s resulted in a population increase until the carrying capacity (about 300,000; according to Fay *et al.* (1997)) was reached in the 1970 to 1980s and productivity began to decline. The subsequent lack of harvest quotas in the United States beginning in 1979 and the reduced productivity levels resulted in another population decline, and the population is once again likely limited primarily by subsistence harvest, although other factors such as haulout mortalities may also be important (Udevitz et al. 2013). Garlich-Miller *et al*. (2011*a*) predicted that changing sea ice dynamics will result in further population declines in the future, but could not specify the magnitude or rate of decline. Given the suite of challenges associated with walrus aerial surveys, many of which cannot be overcome (e.g., poor weather, extensive area, estimate imprecision), it is clear that new approaches to evaluate population status and trend need to be explored. The U.S. Fish and Wildlife Service (Service) is developing a project to test the feasibility of genetic mark-recapture methods to estimate population size and trend. The successful development of a repeatable, unbiased, and precise estimate of population size will greatly facilitate our walrus conservation efforts including those directed at harvest management (USFWS 1994).

MAXIMUM NET PRODUCTIVITY RATES

Estimates of net productivity rates for walrus populations have ranged from 3 to 13% per year with most estimates between 5-10% (Chapskii 1936; Mansfield 1959; Krylov 1965, 1968; Fedoseev and Gol'tsev 1969; Sease 1986; DeMaster 1984; Sease and Chapman 1988; Fay *et al.* 1997). Chivers (1999) developed an individual age-based model of the Pacific walrus population using published estimates of survival and reproduction. The model yielded a maximum population growth rate (R_{MAX}) of 8%, which we use as the maximum net productivity rate in this assessment. Empirical estimates of age-specific survival rates for free ranging walruses are not available.

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) of a marine mammal stock is defined in the MMPA as the maximum number of animals, not including natural mortalities that may be removed from a marine mammal stock while allowing that stock to reach or maintain its optimal sustainable population. The PBR is the product of the following factors: (A) the minimum population estimate of the stock, (B) one-half the maximum theoretical or estimated net productivity rate of the stock at a small population size, and (C) a recovery factor between 0.1-1.0 (MMPA §3(20)). Mathematically, PBR = N_{MIN} x 0.5 R_{MAX} x F_R; where N_{MIN} is minimum populations size, R_{MAX} the net productivity rate, and F_R a recovery factor. The F_R for the Pacific walrus is 0.5 (NMFS 2005) because the population is a candidate for listing under the U.S. Endangered Species Act of 1973, as amended (ESA) (USFWS 2011). The net productivity rate is estimated as 0.08 (Chivers 1999). Therefore, for the Pacific walrus population:

$$N_{MIN} = 129{,}000$$

$$R_{MAX} = 0.08$$

$$F_R = 0.5$$

$$PBR = (129{,}000 \times [0.5*0.08]*0.5) = 2{,}580$$

HUMAN CAUSED MORTALITY AND SERIOUS INJURY

Human Caused Mortalities

Subsistence Harvest

Over the past 60 years the Pacific walrus population has sustained estimated annual harvest removals ranging from 3,184 to 16,127 animals (mean = 6,440; Figure 2). Harvest levels since 2006 are 5 to 68% lower than this long-term average. It is not known whether recent reductions in harvest levels reflect changes in walrus abundance or hunting opportunities, but hunters consistently state that more frequent and severe storms are affecting hunting effort (EWC 2003, Oozeva *et al.* 2004). Other factors affecting harvest levels included: 1) the cessation of Russian commercial walrus harvests after 1990; and 2) changes in political, economic, and social conditions of subsistence hunters in Alaska and Chukotka.

The Service uses the average annual harvest over the past five years as an estimate of current harvest levels in the United States and Russia. Total U.S. annual harvest is estimated using data collected by direct observation in selected communities and through the statewide regulatory Marking, Tagging, and Reporting Program. The two sources of data are combined to calculate annual reporting compliance and to correct for any unreported harvest. Total U.S. subsistence harvest is estimated as the sum of reported and estimated unreported harvests. Harvest estimates in Russia were collected through both an observer program and a reporting program instituted by the Russian Federation.

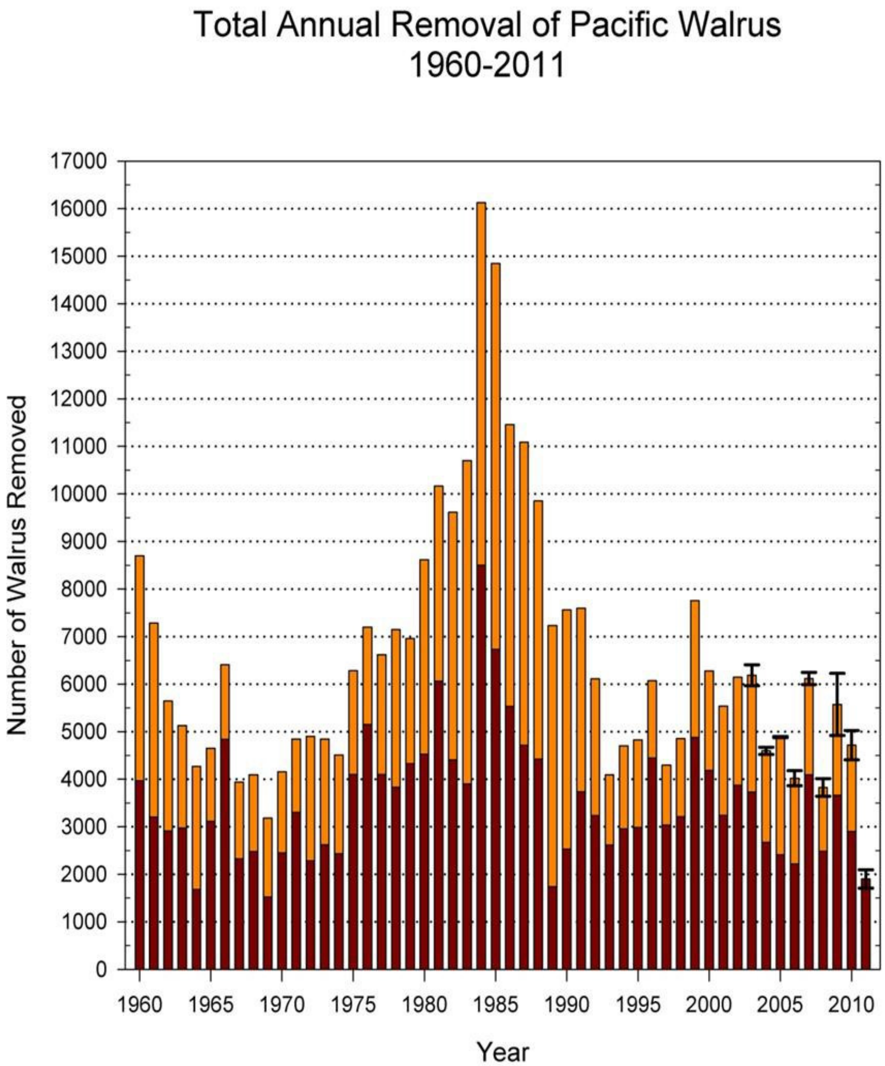

Figure 2. Total annual harvest removals for the Pacific walrus population from 1960 to 2011. Error bars for 2003-2011 denote the standard error around the estimate. Russian data for 2011 not included.

Using data collected between 1952 and 1972, Fay *et al*. (1994) estimated that 42% of walruses that were shot at were lost after being hit. All walruses that have been shot with a firearm are either killed immediately or assumed to be mortally wounded; however, they are often not retrievable if they die in the water and sink or if they are wounded and escape (Fay *et al.* 1994). We recognize that hunting equipment and techniques have improved since Fay *et al.* (1994) published their estimate; however, that estimate is still the best available. We therefore multiply the estimated harvest by 1.42 to adjust for walruses shot but not retrieved (i.e., struck and lost), resulting in a more accurate estimate of total number of walruses harvested.

Harvest mortality levels from 2006 to 2010 are estimated at 3,828 to 6,119 walruses per year (Table 2). The sex-ratio of the reported U.S. walrus harvest over this time period was 1.3:1 males to females. The sex-ratio of the reported Russian walrus harvest was 3.1:1 males to females based on harvest information collected by ChukotTINRO from 1999 to 2009 (Kochnev 2010).

Impacts of climate change on future subsistence harvests of walruses are difficult to predict (Holversrud 2008). Changes in walrus distribution, abundance, and health; sea ice characteristics and distribution; length and timing of the hunting season; and weather and sea state during the hunting season can all influence hunting success. Recent harvests are lower than historic levels and more frequent storms during the traditional hunting season, which limit hunting opportunities, appear to be a contributing factor. Holversrud (2008) predicted that climate change would result in a decline in the subsistence harvest of marine mammals. Garlich-Miller *et al*. (2011a) predicted that walrus harvest levels would remain relatively stable. Since 2006, the estimated total removal of walruses has fluctuated from year to year by an average of 3%, but is highly variable (e.g., 2006 to 2007, a 52% increase; and 2007 to 2008, a 60% decline).

Although fewer walruses are currently being harvested overall, of those animals harvested more are being harvested earlier in the spring and earlier in the winter than during the previous 20 years demonstrating that hunters will likely adapt to changing hunting and sea ice conditions. Harvest levels must be assessed within the context of the best available information on walrus population size, weather and climate, and political, economic, and social conditions of subsistence hunters in Alaska and Chukotka. Garlich-Miller et al. (2011a) assumed that summer sea ice loss would result in a reduced walrus population over time and that subsistence harvests could become unsustainable if not reduced in concert with any decline in the population. The recent adoption of trip limit ordinances by the Native Villages of Gambell and Savoonga and the acquisition of a Tribal Wildlife Grant to ensure administration of those ordinances is a positive development in this arena.

Table 2. Mean (standard error) harvest of Pacific walruses, 2006-2010. Russian harvest information was provided by ChukotTINRO and the Russian Agricultural Department. United States harvest information was collected by the U.S. Fish and Wildlife Service, and adjusted for unreported walruses using a mark-recapture method. Total harvest includes a struck and lost factor of 42% (Fay *et al*. 1994).

Year	Total harvest	United States harvest	Russian harvest
2006	4,022(157)	1,286(91)	1,047
2007	6,119(127)	2,376(74)	1,173
2008	3,828(185)	1,442(107)	778
2009	5,547(654)	2,123(379)	1,110
2010	4,716(308)	1,682(178)	1,053
Five year mean	4,852(346)	1,782(200)	1,032(67)

Cooperative Agreements have been developed annually between the Service and the Eskimo Walrus Commission since 1997 to facilitate the participation of subsistence hunters in activities related to the conservation and management of the walrus in Alaska. This co-management process is on-going. Ensuring that harvest levels remain sustainable is a goal shared by subsistence hunters and resource managers in the United States and Russian Federation. Achieving this management goal will require continued investments in co-management relationships, harvest monitoring programs, international coordination, and research.

Fisheries Related Mortalities and Injuries

A complete list of fisheries and marine mammal interactions is published annually by the National Oceanic and Atmospheric Administration (NOAA)-Fisheries, the most recent of which was published on August 29, 2013 (NOAA 2013). Pacific walruses occasionally interact with trawl and longline gear of groundfish fisheries. No data are available on incidental catch of walruses in fisheries operating in Russian waters, although trawl and longline fisheries are known to operate there. In Alaska each year, fishery observers monitor a percentage of commercial fisheries and report injury and mortality of marine mammals incidental to these operations. Overall, 13 fisheries, with observers, operate in Alaska within the range of the Pacific walrus in the Bering Sea, and could potentially interact with them.

Mortalities

Incidental mortality during 2006-2010 was observed in only one fishery, the Bering Sea/Aleutian Island flatfish non-pelagic trawl (Table 3); which, according to NOAA-Fisheries is a Category II Commercial Fishery with an estimated 34 vessels and/or persons participating. Observer coverage for this fishery averaged 88% during 2006-2010. The mean number of observed mortalities was one walrus per year, with a range of zero to three (Table 3). The total estimated annual fishery-related incidental mortality in Alaska was two walruses per year. We consider fishery related mortality to be insignificant.

Table 3. Summary of incidental mortality of Pacific walruses in the Bering Sea/Aleutian Islands flatfish trawl fishery from 2006-2010 and estimated mean annual mortality. Data provided by the National Marine Fisheries Service.

Year	Observer coverage (%)	Observed mortality	Estimated mortality	95% CI
2006	68	2	3	1 – 6
2007	72	1	3	1 – 5
2008	100	1	1	0.6 – 1.4
2009	100	0	0	
2010	100	2	2	1 – 3
Five year mean(SE[a])	88(7)	1(0.4)	2(0.6)	

[a]standard error.

Injuries

No incidental injury was observed during this time period; therefore, annual serious injury is estimated to be zero.

Other Removals

Between 2006-2011, satellite transmitters were affixed to 348 walruses, and collections of skin and blubber samples with biopsy darts were attempted from 183 walruses. No mortalities or serious injuries were directly associated with those research activities. However, in 2011, walruses at the Point Lay, Alaska haulout cleared the beach as USGS researchers, ferried by local guides, boated past resulting in the death of one calf (Jay 2012).

Up to 52 orphaned walrus calves were captured in Russia and placed on public display between 2006-2010. In addition, 3 calves were found on the beach near Barrow, Alaska in 2012 and taken into captivity. Based on this information, about 19 (standard error = 17) walruses per year were removed from the wild due to other human activities.

Total Estimated Human-Caused Mortality and Serious Injury

The average (standard error) total annual human-caused mortality or removal is 4,873 (346) walruses (2 due to fisheries interactions, 4,852 due to harvest, and 19 due to other human activities). There is no evidence that levels of human-caused serious injury are significant at this point.

Mortalities at coastal haulouts are due to several natural sources (poor condition, old age, injuries, predation, etc.) and occur at all haulouts at an unknown background level. Mortalities due to human caused stampedes also occur but are hard to quantify – most events are observed after the fact (Fay and Kelley 1980, Fischbach *et al.* 2009), some may go undetected, and carcasses can be redistributed during storms and consumed by predators. In 2007, more than 3,200 haulout mortalities were attributed to disturbance events along the Russian coast, but none were noted in Alaska. In 2008, few haulout mortalities were observed (0 in the United States, 165 in Russia) as remnant ice in the Chukchi Sea allowed walruses to stay offshore. In 2009, 131 calves were apparently trampled in a disturbance event at Icy Cape, Alaska (Fischbach *et al.* 2009) and another 53 were reported from other locations in Alaska with 453 counted in Russia. In 2010, 680 carcasses were counted at four haulouts in Russia (A. Kochnev, pers. comm.) and less than 200 were observed at Point Lay, Alaska (USFWS, unpubl. data). In 2011, 376 carcasses were counted in Russia (A. Kochnev, pers. comm.) and about 100 carcasses were found at the Point Lay haulout (USFWS, unpubl. data). Haulout management programs in Russia and the United States may be a successful management tool in reducing disturbance related mortalities compared to the extreme event in 2007.

STATUS OF STOCK

Pacific walrus are not designated as depleted under the MMPA; however, we have determined that listing the Pacific walrus as endangered or threatened under the ESA is warranted, but precluded by higher priority listing actions (USFWS 2011). Based on the best available information, the estimated incidental mortality and serious injury related to commercial fisheries (two walruses per year) is less than one percent of PBR and therefore can be considered insignificant and approaching a zero mortality and serious injury rate. However, the total human-caused removals exceed the PBR of 2,580. Therefore, the Pacific walrus is classified as a strategic stock.

EMERGING CONSERVATION ISSUES

A status review for the Pacific walrus was completed in 2011 in response to the ESA listing petition (Garlich-Miller *et al.* 2011*a*, and is available at: *http://alaska.fws.gov/fisheries/mmm/walrus/pdf/review_2011.pdf*. That review provides a comprehensive analysis of the stressors currently affecting the Pacific walrus population. The major findings of that analysis have been incorporated into this document in the appropriate sections. Readers should refer to Garlich-Miller *et al.* (2011*a*) for additional information on topics not covered by this stock assessment report.

Chukchi Coast Haulout Use

Over the past decade, the number of walruses coming to shore in summer and fall along the coastline of the Chukchi Sea in both Alaska and Russia has increased (Kavry *et al.* 2008, Garlich-Miller *et al.* 2011*a*) coincident with the earlier and more extensive melting of sea ice. In fall 2007, 2009, 2010, and 2011, large aggregations of females and young (about 3,000 to 30,000) were observed along the Alaska coast. An area of concern is the amount of walrus prey within the foraging range of coastal haulouts (Garlich-Miller *et al*. 2011*a*). As more walruses use coastal haulouts more frequently and for longer periods each year, prey populations could be depleted. Malnourished walruses have been reported from Chukotka (Ovsyanikov *et al*. 2008, A.A. Kochnev personal communication) and they are also regularly observed in Alaska (Garlich-Miller *et al.* 2011a); however, the majority of walruses observed at fall haulouts in Alaska in 2010 and 2011 were in good physical condition.

Ocean Acidification

The effect of ocean acidification (OA) on walrus prey is another issue of concern because lower pH levels can interfere with invertebrate shell formation and erode existing shells. No information is available about potential impacts on specific walrus prey species. Uncertainty regarding the general effects of ocean acidification has been summarized by the National Research Council (2010:1): “The major changes in ocean chemistry caused by increasing atmospheric CO_2 are well understood and can be precisely calculated, despite some uncertainty resulting from biological feedback processes. However, the direct biological effects of ocean acidification are less certain and will vary among organisms, with some coping well and others not at all." Consequently, although we recognize that effects to calcifying organisms that are important prey items for Pacific walruses may occur in the foreseeable future from ocean acidification, we do not know which species may be able to adapt and thrive, which may decline, or the ability of the walrus to depend on alternative prey items. The prey base of walrus includes over 100 taxa of benthic invertebrates from all major phyla (Sheffield and Grebmeier 2009). Although walruses are highly adapted for obtaining bivalves, they also have the potential to switch to other prey items if bivalves and other calcifying invertebrate populations decline. Whether other prey items would fulfill walrus nutritional needs over their life span is unknown (Sheffield and Grebmeier 2009), and there also is uncertainty about the extent to which other suitable non-bivalve prey might be available, due to uncertainty about the effects of ocean acidification and the effects of ocean warming.

Subsistence Harvest

Recent subsistence harvests are lower than historic levels due to a faster spring migration and more frequent severe storms that have limited hunting opportunities during the spring migration (Kapsch *et al.* 2010). Garlich-Miller *et al.* (2011*a*) predicted that walrus harvest levels would remain relatively stable as hunters adapt to changing hunting conditions, but that summer sea ice loss will result in a reduced walrus population over time, and therefore subsistence harvests could become unsustainable if not reduced similarly. The Service, in cooperation with the Russian Federation, has a comprehensive harvest monitoring program in place that provides detailed information on harvest trends and characteristics. We will continue to cooperatively monitor harvest levels into the future, a key component to maintaining a sustainable harvest.

Oil and Gas Exploration

In 2008, the Minerals Management Service (now the Bureau of Ocean Energy Management) held an oil and gas lease sale for offshore blocks in the eastern Chukchi Sea. In 2009, 2010, and 2011 a number of seismic surveys were conducted in the lease sale area. A significant portion of the Pacific walrus population migrates into the Chukchi Sea region each summer, and the shallow, productive, ice covered waters of the eastern Chukchi Sea are considered particularly important habitat for female walruses and their dependent young. The Hanna Shoal area seems to be particularly attractive to walruses summering in the Chukchi Sea likely due to both high prey abundance and shallow waters. The Service works to monitor and mitigate potential impacts of oil and gas activities on walruses through Incidental Take Regulations (ITR) as authorized under the MMPA. Entities operating under these regulations must adopt measures to ensure that impacts to walruses remain negligible, minimize impacts to their habitat, and ensure no unmitigable adverse impact on their availability for Alaska Native subsistence use. These regulations also specify monitoring requirements that provide a basis for evaluating potential impacts of current and future activities on marine mammals. The current ITRs were renewed in 2013 for another five years. The Service included a thorough analysis of the monitoring data collected in association with previous ITRs when it issued the current ITRs.

The Service (2011) concluded that at current levels, oil and gas exploration posed a relatively minor threat to the Pacific walrus population. However, we noted that a large oil spill could significantly impact the population depending on timing, location, amount and type of oil, efficacy of response efforts, etc.; the current ITRs also provided special considerations to limit potential impacts to walrus utilizing the Hanna Shoal area.

International Commercial Shipping

As summer sea ice melts earlier in the year and the open water extends further north, opportunities for commercial shipping through the arctic increase (Garlich-Miller *et al.* 2011*a*). Transits through the Bering Strait increased significantly between 2009 and 2010 (M. Williams, pers. comm.) and are currently outpacing regulatory efforts to define shipping channels, seasons of use, and mitigation measures to reduce ship strikes, etc. Commercial shipping is expected to increase in the future, but several scenarios are possible depending on economics and international regulatory efforts. Shipping is not currently impacting the Pacific walrus population and not expected to be a major source of mortality in the future.

Disease

During summer and fall 2011, about 130 ringed seals (*Pusa hispida*) were found on the beaches on northwest Alaska with skin lesions and hair loss suggestive of a viral infection. About 48% of those seals were found dead and the others were lethargic. During September 2011, 6% of the walruses at the Point Lay haulout had similar skin lesions, but were otherwise in good physical condition. The majority of affected walruses were subadults and some of those had healed lesions, indicating that the disorder is not necessarily fatal. However, a number of dead calves at the haulout had both skin lesions and signs of trampling trauma (Garlich-Miller *et al.* 2011*b*) and the ultimate cause of death is not known at this time.

In December 2011, the National Marine Science Fisheries (NMFS) declared the seal mortalities an unusual mortality event (UME) and, with the Service concurrence, included walrus in the UME, due to the similarities of the lesions. No causative agent has been identified and it is not known if the same agent is infecting both species. The symptoms appear to be less severe in walruses than in ringed seals in terms of prevalence and mortalities. Sampling of Pacific walrus' tissues and comprehensive laboratory analyses is continuing as part of the UME investigation.

CITATIONS

Acquarone, M., E.W. Born, and J.R. Speakman. 2006. Field metabolic rates of walrus (*Odobenus rosmarus*) measured by doubly labeled water method. Aquatic Mammals 32: 363-369.

Born, E. W., M. Acquarone, L.Ø. Knutsen, and L. Toudal. 2005. Homing behaviour in an Atlantic walrus (*Odobenus rosmarus rosmarus*). Aquatic Mammals 31:23-33.

Born, E.W. and L.Ø. Knutsen. 1997. Haul-out and diving activity of male Atlantic walruses (*Odobenus rosmarus rosmarus*) in NE Greenland. Journal of Zoology 243:381-396.

Burn, D.M., M.A. Webber, and M.S. Udevitz. 2006. Application of airborne thermal imagery to surveys of Pacific walrus. Wildlife Society Bulletin 34:51-58.

Chapskii, K.K. 1936. The walrus of the Kara Sea. Results of an investigation of the life history, geographical distribution, and stock of walruses in the Kara Sea. Transactions of the Arctic Institute 67:1-124.

Chivers, S.J. 1999. Biological indices for monitoring population status of walruses evaluated with an individual-based model. Pages 239-247, *In* Garner, G.W., S.C. Amstrup, J.L. Laake, B.F.J. Manly, L.L. McDonald, and D.G. Robertson (eds.), Marine Mammal Survey and Assessment Methods. A. A. Balkema, Rotterdam.

DeMaster, D.P. 1984. An analysis of a hypothetical population of walruses. Pages 77-80, *In* F.H. Fay and G.A. Fedoseev (eds.), Soviet American Cooperative Research on Marine Mammals, vol. 1, Pinnipeds. NOAA, Technical Report, NMFS 12.

Douglas, D.C. 2010. Arctic sea ice decline: projected changes in timing and extent of sea ice in the Bering and Chukchi Seas. U.S. Geological Survey, Open-file Report 2010-1176, Reston, VA.

Estes, J.A., and J.R. Gilbert. 1978. Evaluation of an aerial survey of Pacific walruses (*Odobenus rosmarus divergens*). Journal of the Fisheries Research Board of Canada 35:1130-1140.

Estes, J.A., and V.N. Gol'tsev. 1984. Abundance and distribution of the Pacific walrus (*Odobenus rosmarus divergens*): results of the first Soviet American joint aerial survey, autumn 1975. Pages 67-76, *In* F.H. Fay and G.A. Fedoseev (eds.), Soviet American Cooperative Research on Marine Mammals, vol. 1, Pinnipeds. NOAA, Technical Report, NMFS 12.

EWC (Eskimo Walrus Commission). 2003. Conserving our culture through traditional management. Department of Natural Resources, Kawerak Inc., Nome, AK.

Fay, F.H. 1982. Ecology and biology of the Pacific walrus (*Odobenus rosmarus divergens*). North American Fauna 74. U.S. Fish and Wildlife Service, Washington, DC.

Fay, F.H. and B.P. Kelly. 1980. Mass natural mortality of walruses (*Odobenus rosmarus*) at St. Lawrence Island, Bering Sea, autumn 1978. Arctic 33:226-245.

Fay, F.H., B.P Kelly, P.H. Gehnrich, J.L. Sease, and A.A. Hoover. 1986. Modern populations, migrations, demography, trophics, and historical status of the Pacific walrus. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Outer

Continental Shelf Environmental Impact Assessment Program, Final Reports of Principal Investigators 37:231-376. National Oceanic and Atmospheric Administration, National Ocean Service, Anchorage, AK.

Fay, F.H., B.P. Kelly, and J.L. Sease. 1989. Managing the exploitation of Pacific walruses: a tragedy of delayed response and poor communication. Marine Mammal Science 5:1-16.

Fay, F.H., B.P. Kelly, P.H. Gehnrich, J.L. Sease, and A.A. Hoover. 1984. Modern populations, migrations, demography, trophics, and historical status of the Pacific Walrus, final report. Outer Continental Shelf Environmental Assessment Program, Institute of Marine Science, University of Alaska-Fairbanks, Fairbanks, AK.

Fay, F.H., J.J. Burns, S.W. Stoker, and J.S. Grundy. 1994. The struck-and-lost factor in Alaskan walrus harvests. Arctic 47:368-373.

Fay, F.H., L.L. Eberhardt, B.P. Kelly, J.J. Burns, and L.T. Quakenbush. 1997. Status of the Pacific walrus Population, 1950-1989. Marine Mammal Science 13:537-565.

Fay, F.H. and S.W. Stoker. 1982a. Analysis of reproductive organs and stomach contents from walruses taken in the Alaskan Native harvest, spring 1980. Final Report. U.S. Fish and Wildlife Service, Anchorage, AK.

Fay, F.H. and S.W. Stoker. 1982b. Reproductive success and feeding habits of walruses taken in the 1982 spring harvest, with comparisons from previous years. Eskimo Walrus Commission, Department of Natural Resources, Kawerak Inc., Nome, AK.

Fedoseev, G.A. 1984. Present status of the population of walruses (*Odobenus rosmarus*) in the eastern Arctic and Bering Sea. Pages 73-85, *In* V.E. Rodin, A.S. Perlov, A.A. Berzin, G.M. Gavrilov, A.I. Shevchenko, N.S. Fadeev, and E.B. Kucheriavenko (eds.), Marine Mammals of the Far East. TINRO, Vladivostok, Russian Federation.

Fedoseev, G.A., and V.N. Gol'tsev. 1969. Age-sex structure and reproductive capacity of the Pacific walrus population. Zoological Journal 48:407-413.

Fedoseev, G.A., and E.V. Razlivalov. 1986. The distribution and abundance of walruses in the eastern Arctic and Bering Sea in autumn 1985. VNIRO, Magadan Branch, Russian Federation.

Fischbach, T.A., D.H. Monson, and C.V. Jay. 2009. Enumeration of Pacific walrus carcasses on beaches of the Chukchi Sea in Alaska following a mortality event, September 2009. U.S. Geological Survey, Open-file Report 2009-1291, Reston, VA.

Garlich-Miller, J., and C.V. Jay. 2000. Proceedings of a workshop concerning walrus survey methods. U.S. Fish and Wildlife Service, Marine Mammals Management, Technical Report 00-2, Anchorage, AK.

Garlich-Miller, J.L., L.T. Quakenbush, and J.F. Bromaghin. 2006. Trends in age structure and productivity of Pacific walruses harvested in the Bering Strait region of Alaska, 1952-2002. Marine Mammal Science 22:880-896.

Garlich-Miller, J., J.G. MacCracken, J. Snyder, M.M. Myers, E. Lance, A. Matz, and J.W. Wilder. 2011*a*. Status of the Pacific walrus (*Odobenus rosmarus divergens*). U.S. Fish and Wildlife Service, Marine Mammals Management, Anchorage, AK.

Garlich-Miller, J., W. Neakok, and R. Stimmelmayr. 2011*b*. Field report: walrus carcass survey, Point Lay, Alaska September 11-15, 2011. U.S. Fish and Wildlife Service, Marine Mammals Management, Anchorage, AK.

Gilbert, J.R. 1986. Aerial survey of Pacific walruses in the Chukchi Sea, 1985. Mimeo Report.

Gilbert, J.R. 1989a. Aerial census of Pacific walruses in the Chukchi Sea, 1985. Marine Mammal Science 5:17-28.

Gilbert, J.R. 1989b. Errata: Correction to the variance of products, estimates of Pacific walrus populations. Marine Mammal Science 5:411-412.

Gilbert, J.R. 1999. Review of previous Pacific walrus surveys to develop improved survey designs. Pages 75-84, *In* Garner, G.W., S.C. Amstrup, J.L. Laake, B.F.J. Manly, L.L. McDonald, and D.G. Robertson (eds.), Marine Mammal Survey and Assessment Methods. A. A. Balkema, Rotterdam.

Gilbert, J.R., G.A. Fedoseev, D. Seagars, E. Razlivalov, and A. LaChugin. 1992. Aerial census of Pacific walrus, 1990. U.S. Fish and Wildlife Service, Marine Mammals Management Technical Report 92-1, Anchorage, AK.

Gol'tsev, V.N. 1976. Aerial surveys of Pacific walruses in the Soviet sector during fall 1975. Procedural Report TINRO, Magadan, USSR (Russian Federation).

Gjertz, I., D. Griffiths, B.A. Krafft, C. Lydersen, and Ø Wiig. 2001. Diving and haul-out patterns of walruses *Odobenus rosmarus* on Svalbard. Polar Biology 24:314-319.

Hills, S. and J.R. Gilbert. 1994. Detecting Pacific walrus population trends with aerial survey — a review. Transactions North American Wildlife and Natural Resource Conference 59:201-210.

Hovelsrud, G.K., M. McKenna, and H.P. Huntington. 2008. Marine mammal harvests and other interactions with humans. Ecological Applications 18:S135-S147.

Jay, C.V. 2012. Annual report of 2011 activities under USFWS permit MA801652-6: Pacific walrus. U.S. Geological Survey, Alaska Science Center, Anchorage, AK.

Jay, C.V., B.G. Marcot, and D.C. Douglas. 2011. Projected status of the Pacific walrus (*Odobenus rosmarus divergens*) in the twenty-first century. Polar Biology 34:1065-1084.

Jay, C.V., S.D. Farley, and G.W. Garner. 2001. Summer diving behavior of male walruses in Bristol Bay, Alaska. Marine Mammal Science 17:617-631.

Johnson, A., J. Burns, W. Dusenberry, and R. Jones. 1982. Aerial survey of Pacific walruses, 1980. U.S. Fish and Wildlife Service, Marine Mammals Management, Anchorage, AK.

Kapsch, M.L., H. Eicken, and M. Robards. 2010. Sea ice distribution and ice use by indigenous walrus hunters on St. Lawrence Island. Pages 115-144, *In* I. Krupnik, C. Aporta, S. Gearheard, G.J. Laidler, and L.K. Holm, (eds.), SIKU: knowing our ice. Documenting Inuit sea-ice knowledge and use. Springer, Dordrecht.

Kavry, V.I., A.N. Boltunov, and V.V. Nikiforov. 2008. New coastal haulouts of walruses (*Odobenus rosmarus*) – response to the climate changes. Pages 248-251, *In* Collection of Scientific Papers from the Marine Mammals of the Holarctic V conference, Odessa, Ukraine, 14-18 October 2008.

Kochnev, A. 2010. Walrus harvest monitoring in Chukotka in 2009. Technical Report. Eskimo Walrus Commission, Department of Natural Resources, Kawerak Inc., Nome, AK.

Kochnev, A.A. Personal Communication. Head of Laboratory of Marine Mammals Study, ChukotTINRO, Pacific Research Institute of Fisheries and Oceanography, P.O. Box 29, Anadyr, Chukotka, 689000, Russian Federation.

Krylov, V.I. 1965. Determination of age, rate of growth, and analysis of the age structure of the catch of the Pacific walrus. Pages 210-211, *In* E.N. Pavlovskii, B.A. Zenkovich, S.E. Kleinenber, and K.K. Chapskii (eds.), Morskie Mlekopitaiushchie. Nauka, Moscow.

Krylov, V.I. 1968. On the present status of stocks of the Pacific walrus and prospects of their rational exploitation. Pages 189-204, *In* V.A. Arsen'ev and K.I. Panin (eds.), Lastonogie Severnoi Chasti Tikhogo Okeana. Pischevaya Promyshlennost', Moscow.

Lydersen, C., J. Aars, and K.M. Kovacs. 2008. Estimating the number of walruses in Svalbard from aerial surveys and behavioral data from satellite telemetry. Arctic 61:119-128.

MacCracken, J.G. 2012. Pacific walrus and climate change: observations and predictions. Ecology and Evolution, in press.

Mansfield, A.W. 1959. The walrus in the Canadian Arctic. Fisheries Research Board of Canada, Circular 2.

National Research Council. 2010. Ocean acidification: a national strategy to meet the challenges of a changing ocean. National Academies Press, Washington, DC.

NMFS (National Marine Fisheries Service). 2005. Revisions to Guidelines for Assessing Marine Mammals Stocks. Available at: *http://www.nmfs.noaa.gov/pr/pdfs/sars/gamms2005.pdf*.

NOAA (National Oceanic and Atmospheric Administration). 2013. List of fisheries 2013. *Federal Register* (78 FR 53336).

NSIDC (National Snow and Ice Data Center). 2012. Arctic sea ice new and analysis. University of Colorado, Boulder, CO. Available at: *http://nsidc.org/arcticseaicenews/.*

Oozeva, C., C. Noongwook, G. Noongwook, C. Alowa, and I. Krupnik. 2004. Watching ice and weather our way. Smithsonian Institution Press, Washington.

Ovsyanikov, N.G., I.E. Menyushina, and A.V. Bezrukov. 2008. Unusual Pacific walrus mortality at Wrangel Island in 2007. Pages 413-416, *In* Collection of Scientific Papers from the Marine Mammals of the Holarctic V conference, Odessa, Ukraine, 14-18 October 2008.

Scribner, K.T., S. Hills, S.R. Fain, and M.A. Cronin. 1997. Population genetics studies of the walrus (*Odobenus rosmarus*): a summary and interpretation of results and research needs. *In* A.E. Dizon, S.J. Chivers, and W.F. Perrin (eds.), Molecular Genetics of Marine Mammals. Marine Mammal Science, Special publication 3:173-184.

Sease, J.L. 1986. Historical status and population dynamics of the Pacific walrus. Thesis, University of Alaska-Fairbanks, Fairbanks, AK.

Sease, J.L., and D.G. Chapman. 1988. Pacific walrus (*Odobenus rosmarus divergens*). Pages 17-38, *In* J.W. Lentfer (eds.), Selected Marine Mammals of Alaska: species accounts with research and management recommendations. Marine Mammal Commission, NTIS PB88-178462, Washington, D.C.

Sheffield, G., and J.M. Grebmeier. 2009. Pacific walrus (*Odobenus rosmarus divergens*): differential prey digestion and diet. Marine Mammal Science 25:761-777.

Smith, M. A. 2010. Arctic Marine Synthesis: Atlas of the Chukchi and Beaufort Seas, Audubon Alaska and Oceana, Anchorage, AK.

Speckman, S.G., V.I. Chernook, D.M. Burn, M.S. Udevitz, A.A. Kochnev, C.V. Jay, A. Lisovsky, A.S. Fischbach, and R.B. Benter. 2011. Results and evaluation of a survey to estimated Pacific walrus population size, 2006. Marine Mammal Science 27:514-553.

Udevitz, M.S., J.R. Gilbert, and G.A. Fedoseev. 2001. Comparison of method used to estimate numbers of walruses on sea ice. Marine Mammal Science 17:601-616.

Udevitz, M.S., C.V. Jay, A.S. Fischbach, and J.L. Garlich-Miller. 2009. Modeling haul-out behavior of walruses in Bering Sea ice. Canadian Journal of Zoology 87:1111-1128.

Udevitz, M.S., R.L. Taylor, J.L. Garlich-Miller, L.T. Quakenbush, and J.A. Snyder. 2013. Potential population-level effects of increased haulout-related mortality of Pacific walrus calves. Polar Biology 36:291-298.

USFWS (U.S. Fish and Wildlife Service). 1994. Conservation plan for the Pacific walrus in Alaska. U.S. Fish and Wildlife Service, Marine Mammals Management, Anchorage, AK.

USFWS (U.S. Fish and Wildlife Service). 2011. Threatened and endangered plants and animals. 12-month finding for listing the Pacific Walrus (*Odobenus rosmarus divergens*). *Federal Register* 76:7634-7679.

USFWS unpublished data. Available from USFWS, Marine Mammals Management, Anchorage Regional Office, 1011 E Tudor Road, Anchorage, AK 99503.

Williams, M. 2010. Personal Communication. World Wildlife Federation.

NORTHERN SEA OTTER (*Enhydra lutris kenyoni*):

Southeast Alaska Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Sea otters occur in nearshore coastal waters of the U.S. along the North Pacific Rim from the Aleutian Islands to California. The species is most commonly observed within the 40-meter (m) (approximately 12.2 feet) depth contour because the animals require frequent access to benthic foraging habitat in subtidal and intertidal zones (Reidman and Estes 1990). Sea otters are not migratory and generally do not disperse over long distances, although movements of tens of kilometers (km) (tens of miles [mi]) are common (Garshelis and Garshelis 1984). Annual home range sizes of adult sea otters are relatively small, with male territories ranging from 4 to 11 square kilometers (km^2) (approximately 10.5 to 28.5 square miles [mi^2]) and adult female home ranges from a few to 24 km^2 (approximately 62 mi^2) (Garshelis and Garshelis 1984; Ralls *et al*. 1988; Jameson 1989). Due to their benthic foraging, sea otter distribution is largely limited by their ability to dive to the sea floor (Bodkin *et al*. 2004).

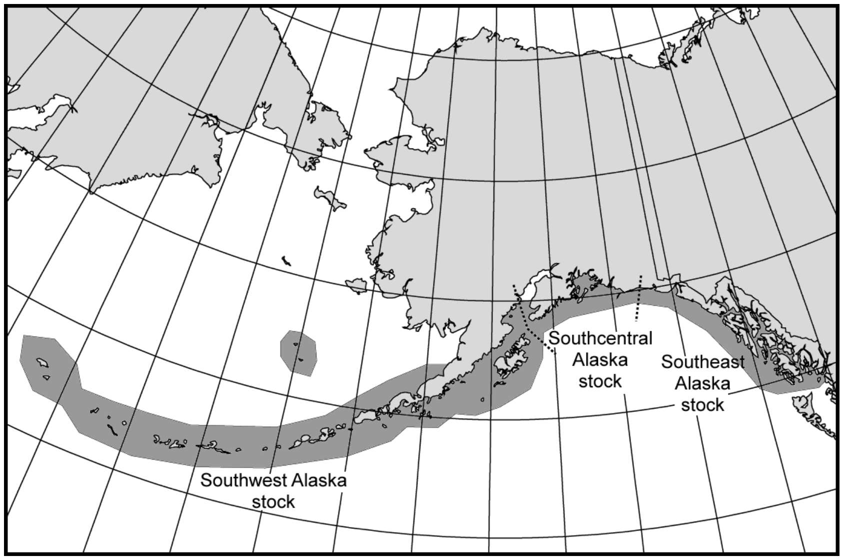

Figure 1. Approximate distribution and stock boundaries of northern sea otters in Alaska waters (shaded area).

The spatial scale at which sea otter populations are managed remains an important, although largely unexplored issue (Bodkin and Ballachey 2010) deserving further study. Bodkin and Ballachey (2010) used models of sea otter mortality to show that range-wide reductions and extirpations during the commercial fur trade of the 18th and 19th centuries occurred not simply because of excessive harvest, but because the harvest was not allocated proportional to the abundance and distribution of sea otters. This process of serial depletion was facilitated by the relatively sedentary nature of sea otters. To reduce the risk of overexploitation, sea otters must be managed on a spatial scale compatible with their well-known behavioral and reproductive biology (Bodkin and Monson 2002), incorporating traits such as home range and movements. These proposed scales for management are much smaller than the currently recognized stocks.

Gorbics and Bodkin (2001) applied the phylogeographic approach of Dizon *et al.* (1992) and used the best available data at the time to identify three sea otter stocks in Alaska: Southeast, Southcentral, and Southwest. The ranges of these stocks are defined as follows: (1) Southeast

Alaska stock extends from Dixon Entrance to Cape Yakataga; (2) Southcentral Alaska stock extends from Cape Yakataga to Cook Inlet including Prince William Sound, the Kenai Peninsula coast, and Kachemak Bay; and (3) Southwest Alaska stock includes the Alaska Peninsula and Bristol Bay coasts, and the Aleutian, Barren, Kodiak, and Pribilof Islands (Figure 1). This stock assessment report is focused on the Southeast stock of sea otters in Alaska.

POPULATION SIZE

Historically, sea otters occurred across the North Pacific Rim, ranging from Hokkaido, Japan, through the Kuril Islands, the Kamchatka Peninsula, the Commander Islands, the Aleutian Islands, peninsular and south coastal Alaska, and south to Baja California, Mexico (Kenyon 1969). In the early 1700s, the worldwide population was estimated to be between 150,000 (Kenyon 1969) and 300,000 individuals (Johnson 1982). Prior to large-scale commercial exploitation, indigenous peoples of the North Pacific hunted sea otters. Although it appears that harvests may have periodically led to local reductions of sea otters (Simenstad *et al.* 1978), the species remained abundant throughout its range until the mid-1700s. Following the arrival in Alaska of Russian explorers in 1741, extensive commercial harvest of sea otters over the next 150 years resulted in the near extirpation of the species. When sea otters were afforded protection by the International Fur Seal Treaty in 1911, probably fewer than 2,000 animals remained in thirteen remnant colonies (Kenyon 1969).

Although population recovery began following legal protection, no remnant colonies of sea otters existed in Southeast Alaska. As part of efforts to re-establish sea otters in portions of their historical range, otters from Amchitka Island and Prince William Sound were translocated to other areas (Jameson *et al.* 1982). These translocation efforts met with varying degrees of success. From 1965 to 1969, 412 otters (89% from Amchitka Island in southwest Alaska, and 11% from Prince William Sound in southcentral Alaska) were translocated to six sites in southeast Alaska (Jameson *et al.* 1982). In the first 20 years following translocation, these populations increased in numbers and expanded their range (Pitcher 1989).

Nearly all of the current population estimates for the Southeast Alaska stock were developed using the aerial survey methods of Bodkin and Udevitz (1999). The lone exception was a survey of the outer coastline from the western boundary of the stock at Cape Yakataga to Cape Spencer conducted by U.S. Geological Survey (USGS) in 2000. Thirty-two otters were estimated to be in that area (coefficient of variation [CV]=0.378). In 2005, the U.S. Fish and Wildlife Service (Service) surveyed Yakutat Bay (estimate number of otters [N]=1,582; CV=0.33; Gill and Burn 2007). In 2010, the Service surveyed the southern half (Kuiu and Kupreanof Islands south to the Canadian border) of Southeast Alaska (SSE) (N=12,873; CV=0.18; Gill and Burn unpublished data). The northern half (Admiralty and Baranof Islands north to Glacier Bay) of Southeast Alaska (NSE) was surveyed by the Service in 2011 (N=2,717; CV=0.22; Gill and Burn unpublished data). Glacier Bay (GB) National Park (NSE) was not included in the 2011 survey as USGS had separate plans to conduct replicate surveys in the Bay in 2012 to add to a long-term data set for the National Park (NP). The estimate from that 2012 survey is N=8,508; CV=0.20 (Esslinger *et al.* 2013). The most recent population estimates for the Southeast Alaska stock are presented in Table 1, which shows a total estimate of 25,712 sea otters for the stock.

Table 1. Abundance estimates for the Southeast Alaska stock of northern sea otters.

Survey Area	**Year**	**Unadjusted count**	**Adjusted Estimate**	**CV**	**N_{MIN}**	**Reference**
North Gulf of Alaska	2000	15	32	0.38	24	USGS unpublished data
Glacier Bay (NP)	2012		8,508	0.20	7,201	Esslinger, Bodkin, & Weitzman (2013)
Northern Southeast Alaska (NSE)	2011		2,717	0.22	2,270	Gill and Burn unpublished data
Southern Southeast Alaska (SSE)	2010		12,873	0.18	11,099	Gill and Burn unpublished data
Yakutat Bay	2005		1,582	0.33	1,203	Gill and Burn (2007)
Current Total			**25,712**		**21,798**	
2008 SAR Total			10,563		9,136	

Minimum Population Estimate

The minimum population estimate (N_{MIN}) for this stock is calculated using Equation 1 from the Potential Biological Removal Guidelines (Wade and Angliss 1997): $N_{MIN} = N/\exp(0.842 \times [\ln(1+[CV(N)]^2)]^{1/2})$. The N_{MIN} for each survey area is presented in Table 1. The estimated N_{MIN} for the entire Southeast Alaska stock is 21,798 sea otters.

Current Population Trend

The trend for this stock of sea otters has generally been one of growth (Pitcher 1989, Agler *et al.* 1995, Esslinger and Bodkin 2009). Comparing the current population estimate with that of the previous stock assessment reports suggests that this growth trend is continuing. The estimated population size (25,712) of this stock currently is more than double what was estimated in the previous (2008) stock assessment report (10,563). However, it is important to note that the population estimate published in the 2008 stock assessment report was based on survey data from 2002 and 2003. Therefore, we can only conclude that the Southeast population stock has doubled since 2003.

The 2010-2011 survey followed the same Bodkin and Udevitz (1999) methods as the 2002-2003 survey effort (Esslinger and Bodkin 2009) so results of those two surveys can be directly compared. In addition, all surveys in the GBNP time series followed the Bodkin and Udevitz (1999) method. The Service's 2010 survey of SSE showed an average annual increase of 12% per year over the last seven years and the Service's 2011 survey of NSE Alaska (minus GBNP) showed an average annual increase of 4% per year over the last nine years. The USGSs' survey of GBNP showed an average annual increase of 20% per year over the last six years. If we include the 2012 GBNP estimate with the estimate for the 2011 NSE Alaska the growth rate is about 14% per year in NSE Alaska which is in line with the growth rate for SSE Alaska. Hence, the northern and southern portions of Southeast Alaska appear to be growing at the same average annual rate; between 12-14% per year.

When compared to SSE, the sea otter population has also not appreciably expanded its range in NSE outside of GBNP since 2002 (Esslinger and Bodkin 2009, Gill and Burn unpublished data). However, otters have occupied appreciable new habitat in SSE since 2003 (Esslinger and Bodkin 2009, Gill and Burn unpublished data). There appear to be two major areas of expansion in SSE; otters have moved in large numbers along the northwest coast of Kuiu Island up into Keku Strait and then animals from this area have crossed Frederick Sound to the southern tip of Admiralty Island, and finally otters have expanded northward from the Barrier Islands through Tlevak Strait.

Sea otter abundance in Yakutat Bay has also increased, by an estimated 14.6% per year, over the last decade, likely through reproduction, although some amount of immigration cannot be ruled out (Gill and Burn 2007). During this process, otters appear to have expanded their range to include the western shores of Yakutat Bay.

Based on this information the current population trend for the Southeast Alaska stock is increasing.

MAXIMUM NET PRODUCTIVITY RATE

Estes (1990) estimated a population growth rate of 17 to 20% per year for northern sea otter populations expanding into unoccupied habitat in the Aleutian Islands, southeast Alaska, British Columbia, Washington State, and central California. Although maximum productivity rates (R_{MAX}) have not been measured through much of the sea otter's range in Alaska, in the absence of more detailed information, the rate of 20% calculated by Estes (1990) is considered the best available estimate of R_{MAX}. The Service's 2010 survey of SSE and 2011 survey of NSE shows a current growth rate of 12% and 4% respectively per year (minus GBNP). The USGS' 2012 survey of GBNP shows a current growth rate of 20% per year. Combining the data from NSE AK indicates that area is growing at a rate of 14% per year which compares to the rate of 12% per year in SSE AK. Consequently, we estimate the current net productivity rate for the entire Southeast Alaska population stock to be between 12-14% per year.

POTENTIAL BIOLOGICAL REMOVAL

Under the Marine Mammal Protection Act (MMPA), the potential biological removal (PBR) is defined as *the maximum number of animals, not including natural mortalities, that may be removed from a marine mammal stock while allowing that stock to reach or maintain its optimal sustainable population.* Potential biological removal is the product of the minimum population estimate (N_{MIN}), one-half the maximum theoretical net productivity rate, and a recovery factor (F_R): PBR = N_{MIN} x 0.5 R_{MAX} x F_R. The recovery factor for this stock is 1.0 (Wade and Angliss 1997) as population levels have been stable or increasing with a known human take. Thus, for the Southeast stock of sea otters, PBR = 2,179 animals (21,798 x 0.5(0.2) x 1.0).

ANNUAL HUMAN CAUSED MORTALITY

Fisheries Information

A complete list of fisheries and marine mammal interactions is published annually by the National Oceanic and Atmospheric Administration (NOAA) Fisheries, the most recent of which was published on August 29, 2013 (78 FR 53336). Fisheries that have been known to interact with sea otters in the Southwest and Southcentral Alaska stocks do occur in Southeast Alaska, specifically the Southeast Alaska salmon drift gillnet (474 vessels) and the Yakutat salmon set gillnet (167 participants) fisheries. Sea otters are also known to interact with pot fisheries in California (Hatfield *et al*. 2011); in Southeast Alaska, there are 415 crab pot fishery participants and 274 shrimp pot participants. There are also 243 miscellaneous finfish pot fishery participants across the entire state (numbers are not available for specific areas). Available information suggests that fisheries using other types of gear, such as trawl, longline, and purse seine, are less likely to have interactions with sea otters across their entire range in Alaska due to either the areas where such fisheries operate (i.e., outside of sea otter habitat), the specific gear used (i.e., otters are not going to tangle or get trapped in a longline), or both.

Although commercial fisheries in Alaska have observer programs that monitor and report injury and mortality of marine mammals incidental to their operations, a reliable estimate of the levels of commercial fisheries incidental mortality and serious injury relative to the southeast sea otter stock cannot be made because observer coverage is not sufficient and data are not collected consistently over time. Of the observer programs in operation within the stock, no incidents of sea otter incidental take were observed in trawl, longline, or pot groundfish fisheries in Southeast Alaska from 1989 to 2010 (Perez 2003, Perez 2006, Perez 2007, Manly 2009, Bridget Mansfield 2011 personal communication). However, there has been no observer effort to document by-catch in the salmon set or drift gillnet fisheries or in the crab or shrimp pot fisheries in Southeast Alaska. Hatfield *et al*. (2011) contend that significant sea otter mortality from pot fishery by-catch might easily go undetected, even when seemingly high levels of observer effort exist.

An additional source of information on the number of sea otters killed or injured incidental to commercial fishery operations in Alaska is found in fisher self-reports required of vessel owners by NOAA Fisheries. From 1990 to 1993, self-reported fisheries data showed no sea otter kills or injuries in Southeast Alaska. Self-reports were incomplete for 1994 and not available for 1995 or 1996. Between 1997 and 2010, there were no records of incidental take of sea otters by commercial fisheries in this region. Credle *et al.* (1994) considered fisher self-reports to be a minimum estimate of incidental take as these data are most likely negatively biased. Indeed, anecdotal observations have been reported to the Service within the last five years suggesting that sea otters do interact with crab pots in Southeast Alaska. As sea otters reoccupy portions of their former habitat in Southeast Alaska, co-occurrence with pot fisheries will increase and so will the likelihood of mortalities or serious injury.

Information is insufficient to determine whether or not the total fishery mortality and serious injury for the Southeast Alaska stock of the northern sea otter is insignificant and is approaching a zero mortality and serious injury rate.

Oil Spills

Activities associated with exploration, development, and transport of oil and gas resources can adversely impact sea otters and nearshore coastal ecosystems in Alaska. Sea otters rely on air trapped in their fur for conserving body heat and buoyancy. Contamination with oil drastically reduces the insulative value of the pelage, and consequently, sea otters are among the marine mammals most likely to be detrimentally affected by contact with oil. It is believed that sea otters can survive low levels of oil contamination (< 10% of body surface), but that greater levels (>25%) will lead to death (Costa and Kooyman 1981, Siniff *et al.* 1982). Vulnerability of sea otters to oiling was demonstrated by the 1989 *Exxon Valdez* oil spill in Prince William Sound. Total estimates of mortality caused by the spill for the Prince William Sound area vary from 750 (range 600-1,000) (Garshelis 1997) to 2,650 (range 500-5,000) (Garrot *et al.* 1993) otters. Statewide, it is estimated that 3,905 sea otters (range 1,904-11,257) died in Alaska as a result of the spill (DeGange *et al.* 1994), but none of these were from the Southeast Alaska stock.

There is currently no oil and gas development in Southeast Alaska. Tankers carrying oil south from the Trans-Alaska Pipeline typically travel offshore of Southeast Alaska. Information on oil spills compiled by the Alaska Department of Environmental Conservation from 2006 to 2010 indicates that there were no reported spills of crude oil in Southeast Alaska. In addition to spills that may occur in association with the development, production, and transport of crude oil, each year numerous spills of non-crude oil products in the marine environment occur from ships and shore facilities throughout Southeast Alaska. During that same time period, there was an average of 133 spills each year, ranging in size from less than 1 and up to 17,800 gallons (approximately 4 to 64,600 liters). The vast majority of these spills were small, with a mean size of 46 gallons (1,748 liters), and there is no indication that these small-scale spills have had an impact on the Southeast Alaska stock of northern sea otters at the population level.

Subsistence/Native Harvest Information

The MMPA exempts Alaska Natives from the prohibition on take of marine mammals, provided such taking is not wasteful and is done for subsistence use or for creating and selling authentic handicrafts or clothing. According to the Service's Law Enforcement records from 2006 to 2010, individuals were prosecuted for unlawful possession, transport, or sale of 208 sea otter hides or skulls taken within the range of the Southeast Alaska stock. During the same time period, there was one prosecution for unlawful take of a single sea otter hide. Data for subsistence harvest of sea otters in Southeast Alaska are collected by a mandatory Marking, Tagging and Reporting Program administered by the Service since 1988. Figure 2 provides a summary of subsistence harvest information for the Southeast stock from 1989 to 2010. The mean reported annual subsistence take during the past five complete calendar years (2006-2010) was 447 animals. This is an increase from the annual average of 322 sea otters hunted during the previous five-year period. Reported age composition from 2006 to 2010 was the same as the previous five years; 83% adults, 14% subadults, and 3% pups. Reported sex composition from 2006 to 2010 was also the same as the previous five years; 72% males, 27% females, and 1% of unknown sex.

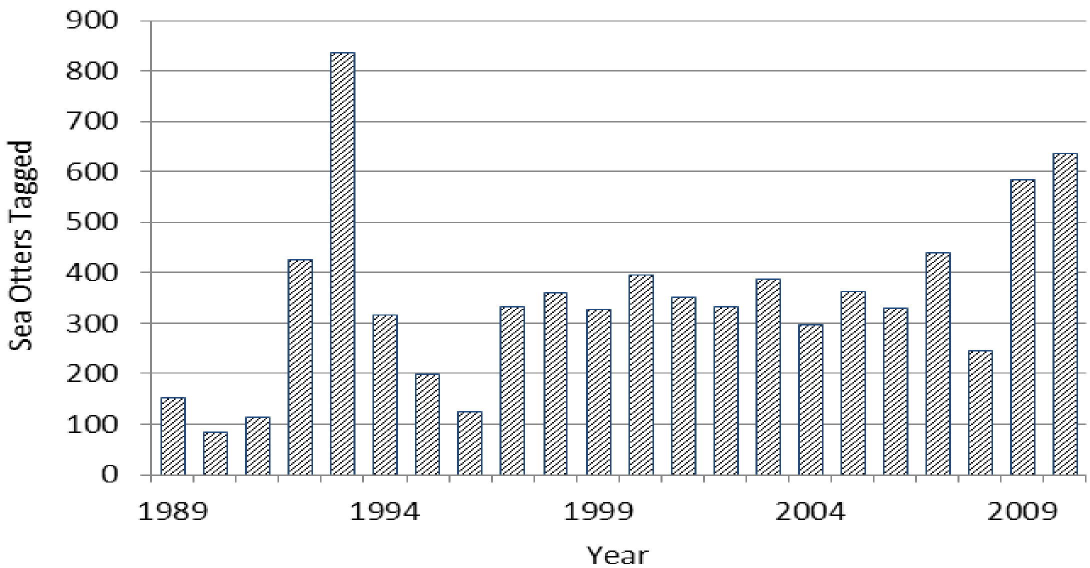

Figure 2. Reported subsistence harvest of northern sea otters from the Southeast Alaska stock, 1989 to 2010.

Research and Public Display

In the past five years, no sea otters were removed from the Southeast Alaska stock for public display. In 2011, 93 sea otters were captured and released for scientific research in the Southeast Alaska stock; the Service captured and released 31 sea otters in the Keku Strait region and the USGS captured and released 62 sea otters in Cross Sound and off of southern Baranof Island. There were no mortalities and serious injuries reported from either of these research efforts.

Other Factors

Since 2002 the Service has undertaken a health and disease study of northern sea otters from all three Alaskan stocks. On average, the Service conducts about 100 necropsies a year on sea otter carcasses to determine cause of death, disease incidence and status of general health parameters. Boat strike is a recurring cause of death across all three stocks. However, it has been determined in most of these cases that although trauma was the ultimate cause of death, there was a contributing factor, such as disease or biotoxin exposure, which incapacitated the animal and made it more vulnerable to boat strike.

In August 2006, the Working Group on Marine Mammal Unusual Mortality Events reviewed information provided by the Service, and declared that a dramatic increase in sea otter strandings in Kachemak Bay, in the Southcentral Alaska stock, since 2002 constituted an Unusual Mortality Event (UME) in accordance with section 404 of the MMPA. The disease that typifies this UME is caused by a *Streptococcus infantarius* infection and has been observed over a broad geographic range in Alaska, including a few cases from Southeast Alaska; however, the majority of cases have come from Kachemak Bay in the Southcentral Alaska stock. It is not clear if the observed stranding pattern is representative of overall sea otter mortality, or an artifact of having a well-developed stranding network in the Kachemak Bay area. The Service will continue to work with NOAA Fisheries and the Alaska SeaLife Center to develop the infrastructure for a statewide marine mammal stranding network in Alaska.

STATUS OF STOCK

The known level of direct human-caused mortality within the Southeast Alaska stock does not exceed the PBR level, and the Southeast Alaska stock is neither listed as "depleted" under the MMPA or listed as "threatened" or "endangered" under the Endangered Species Act of 1973, as amended, nor is it likely to be listed as such in the foreseeable future. The known level of direct human-caused mortality is 447 otters per year. It would require an annual rate of human-caused mortality from additional hunting or fisheries interactions of 1,733 more otters per year for the total amount of direct human-caused mortality to exceed PBR for this stock. Despite uncertainties regarding fishery mortality, we believe that it is unlikely this level is occurring at present. Therefore, the Southeast Alaska stock of the northern sea otter is classified as non-strategic. In addition, although the Service does not currently know the OSP for this stock, based on the known population level and our estimate of growth and considering the known level of human-caused mortality, we have determined that this stock is increasing and that human-caused mortality and serious injury is not likely to cause the stock to be reduced or to decrease its growth rate. Therefore, we would not expect the current level of human-caused mortality and serious injury to cause this stock to be reduced below its plausible OSP.

CITATIONS

Agler, B.A., S.J. Kendall, P.E. Seiser, and J.R. Lindell. 1995. Estimates of marine bird and sea otter abundance in Southeast Alaska during summer 1994. Migratory Bird Management, U.S. Fish and Wildlife Service, Anchorage, Alaska. 90pp.

Bodkin, J.L., and M.S. Udevitz. 1999. An aerial survey method to estimate sea otter abundance. Pages 13-26 *In:* G.W. Garner *et al.*, editors. Marine Mammal Survey and Assessment Methods. Balkema, Rotterdam, Netherlands.

Bodkin, J. L. and B. E. Ballachey. 2010. Modeling the effects of mortality on sea otter populations. USGS Scientific Investigation Report 2010-5096. 12pp.

Bodkin, J. L. and D. H. Monson. 2002. Sea otter population structure and ecology on Alaska. Arctic Research of the United States 16:31-36.

Bodkin, J. L., G. G. Esslinger, and D. H. Monson. 2004. Foraging depths of sea otters and implications to coastal marine communities. Marine Mammal Science 20(2):305-321. DOI: 10.1111/j.1748-7692.2004.tb01159.xCosta, D.P., and G.L. Kooyman. 1981. Effects of oil contamination in the sea otter *Enhydra lutris.* Outer Continental Shelf Environmental Assessment Program. NOAA Final Report. La Jolla, California.

Credle, V.A., D.P. DeMaster, M.M. Merlein, M.B. Hanson, W.A. Karp, and S.M. Fitzgerald (eds.). 1994. NMFS observer programs: minutes and recommendations from a workshop held in Galveston, Texas, November 10-11, 1993. U.S. Department of Commerce, NOAA Tech. Memo. NMFS-OPR-94-1. 96 pp.

DeGange, A.R., A.M. Doroff, and D.H. Monson. 1994. Experimental recovery of sea otter carcasses at Kodiak Island, Alaska, following the *Exxon Valdez* oil spill. Marine Mammal Science 10:492-496.

Dizon, A.E., C. Lockyer, W.F. Perrin, D.P. DeMaster, and J. Sisson. 1992. Rethinking the stock concept: a phylogeographic approach. Conservation Biology 6(1):24-36.

Esslinger, G.G., and Bodkin, J.L. 2009. Status and trends of sea otter populations in Southeast

Alaska, 1969–2003: U.S. Geological Survey Scientific Investigations Report 2009–5045, 18 p.

Esslinger, G.G., J.L. Bodkin, and B. Weitzman. 2013. Sea otter population abundance in Glacier Bay, Alaska, May 2012. U.S. Geological Survey Administrative Report.

Estes, J.A. 1990. Growth and equilibrium in sea otter populations. Journal of Animal Ecology 59:385-401.

Garrott, R.A., L.L. Eberhard, and D.M. Burn. 1993. Mortality of sea otters in Prince William Sound following the *Exxon Valdez* oil spill. Marine Mammal Science 9:343-359.

Garshelis, D.L., and J.A. Garshelis. 1984. Movements and management of sea otters in Alaska. Journal of Wildlife Management 48(3):665-678.

Garshelis, D.L. 1997. Sea otter mortality estimated from carcasses collected after the *Exxon Valdez* oil spill. Conservation Biology 11(4):905-916.

Gill, V.A., and D.M. Burn. 2007. Aerial surveys of sea otters in Yakutat Bay, Alaska, 2005. U.S. Fish and Wildlife Service, Marine Mammals Management Office. Technical Report Marine Mammals Management 2007-01. 18pp.

Gill, V.A., and D.M. Burn. Unpublished data. Available from USFWS, Marine Mammals Management, Anchorage Regional Office, 1011 E Tudor Road, Anchorage, AK 99503.

Gorbics, C.S., and J.L. Bodkin. 2001. Stock structure of sea otters (*Enhydra lutris kenyoni*) in Alaska. Marine Mammal Science 17(3):632-647.

Hatfield, B.B., J.A. Ames, J.A. Estes, M.T. Tinker, A.B. Johnson, M.M. Staedler, M.D. Harris. 2011. Sea otter mortality in fish and shellfish traps: estimating potential impacts and exploring possible solutions. Endangered Species Research 13:219–229.

Jameson, R.J. 1989. Movements, home ranges, and territories of male sea otters off central California. Marine Mammal Science 5:159-172.

Jameson, R.J., K.W. Kenyon, A.M. Johnson, and H.M. Wight. 1982. History and status of translocated sea otter populations in North America. Wildlife Society Bulletin. 10: 100-107.

Johnson, A.M. 1982. Status of Alaska sea otter populations and developing conflicts with fisheries. Pages 293-299 *In*: Transactions of the 47th North American Wildlife and Natural Resources Conference, Washington D.C.

Kenyon, K.W. 1969. The sea otter in the eastern Pacific Ocean. North American Fauna 68. U.S. Department of the Interior, Washington D.C.

Manly, B.F.J. 2009. Incidental Take and Interactions of Marine Mammals and Birds in the Yakutat Salmon Setnet Fishery, 2007 and 2008. Western EcoSystems Technology, Cheyenne, Wyoming.

Mansfield, Bridget. Personal Communication. U.S. Department of Commerce, National Marine Fisheries Service.

Perez, M.A. 2003. Compilation of marine mammal incidental catch data for domestic and joint venture groundfish fisheries in the U.S. EEZ of the North Pacific, 1989-2001. NOAA Technical Memorandum NMFS-AFSC-138. 145 pp.

Perez, M.A. 2006. Analysis of marine mammal bycatch data from the trawl, longline, and pot groundfish fisheries of Alaska, 1998-2004, defined by geographic area, gear type, and target groundfish catch species. NOAA Technical Memorandum NMFS-AFSC-167. 194 pp.

Perez, M.A. 2007. Bycatch of marine mammals in the groundfish fisheries of Alaska, 2006. Alaska Fisheries Science Center Processed Draft Report. 67pp.

Pitcher, K.W. 1989. Studies of Southeastern Alaska sea otter populations: distribution, abundance, structure, range expansion and potential conflicts with shellfisheries. Anchorage, Alaska. Alaska Department of Fish and Game, Cooperative Agreement 14-16-0009-954 with U.S. Fish and Wildlife Service. 24 pp.

Ralls, K., T. Eagle, and D.B. Siniff. 1988. Movement patterns and spatial use of California sea otters, in Siniff, D.B., and Ralls, K., eds. Final Report on Contract No. 14-12-001-3003, Population status of California sea otters: Minerals Management Service, Los Angeles, CA, pp. 33-63.

Riedman, M.L., and J.A. Estes. 1990. The sea otter *Enhydra lutris*: behavior, ecology, and natural history. Biological Report; 90 (14). U.S. Fish and Wildlife Service.

Simenstad, C.A., J.A. Estes, and K.W. Kenyon. 1978. Aleuts, sea otters, and alternate stable-state communities. Science 200:403-411. 127 pp.

Siniff, D.B., T.D. Williams, A.M. Johnson, and D.L. Garshelis. 1982. Experiments on the response of sea otters *Enhydra lutris* to oil contamination. Biological Conservation 23:261-272.

USGS unpublished data. Available from the USGS Alaska Science Center, 4210 University Drive, Anchorage, AK 99508.

Wade, P.R., and R. Angliss. 1997. Guidelines for assessing marine mammal stocks: report of the GAMMS workshop April 3-5, 1996, Seattle, Washington. U.S. Department of Commerce, NOAA Technical Memo. NMFS-OPR-12. 93 pp.

Revised: April 2014

NORTHERN SEA OTTER (*Enhydra lutris kenyoni*):

Southcentral Alaska Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

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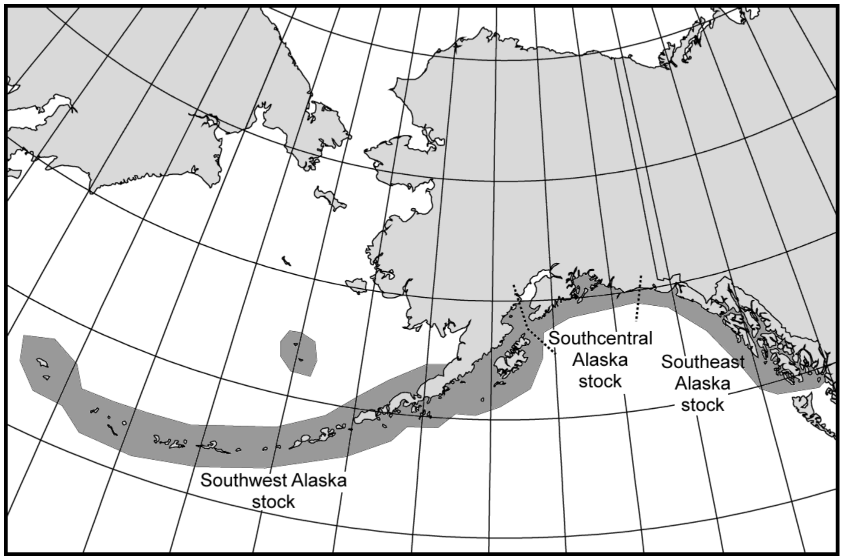

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Alaska stock extends from Dixon Entrance to Cape Yakataga; (2) Southcentral Alaska stock extends from Cape Yakataga to Cook Inlet including Prince William Sound, the Kenai Peninsula coast, and Kachemak Bay; and (3) Southwest Alaska stock includes the Alaska Peninsula and Bristol Bay coasts, and the Aleutian, Barren, Kodiak, and Pribilof Islands (Figure 1). This stock assessment report is focused on the Southcentral stock of sea otters in Alaska.

POPULATION SIZE

Historically, sea otters occurred across the North Pacific Rim, ranging from Hokkaido, Japan, through the Kuril Islands, the Kamchatka Peninsula, the Commander Islands, the Aleutian Islands, peninsular and south coastal Alaska, and south to Baja California, Mexico (Kenyon 1969). In the early 1700s, the worldwide population was estimated to be between 150,000 (Kenyon 1969) and 300,000 individuals (Johnson 1982). Prior to large-scale commercial exploitation, indigenous peoples of the North Pacific hunted sea otters. Although it appears that harvests may have periodically led to local reductions of sea otters (Simenstad *et al.* 1978), the species remained abundant throughout its range until the mid-1700s. Following the arrival in Alaska of Russian explorers in 1741, extensive commercial harvest of sea otters over the next 150 years resulted in the near extirpation of the species. When sea otters were afforded protection by the International Fur Seal Treaty in 1911, probably fewer than 2,000 animals remained in thirteen remnant colonies (Kenyon 1969). Population recovery began following legal protection. As part of efforts to re-establish sea otters in portions of their historical range, otters from Amchitka Island and Prince William Sound were translocated to other areas in the

1960s and 1970s, including to southeast Alaska (Jameson *et al.* 1982). Sea otters have since recolonized much of their historical range in Alaska.

The most recent abundance estimates for survey areas within the Southcentral Alaska stock are presented in Table 1. Estimates for Kenai Fjords and Kachemak Bay have been updated since the previous stock assessment report. In 2008, an aerial survey using the methods described in Bodkin and Udevitz (1999) was conducted within Kachemak Bay, resulting in an estimate of 3,596 sea otters (CV = 0.50; USFWS unpublished data). This method included a survey-specific correction factor to account for undetected animals. A 2010 aerial survey using the Bodkin-Udevitz method in Kenai Fjords National Park resulted in an estimate of 1,322 sea otters (CV = 0.37; Coletti *et al*. 2011). Eastern lower Cook Inlet was surveyed as part of a larger area in 2002, yielding an estimate of 962 sea otters (CV = 0.54; Bodkin *et al*. 2003b) for the areas not covered in 2008 and 2010.

In 2003, an aerial survey of Prince William Sound resulted in an abundance estimate of 11,989 sea otters (CV = 0.18; Bodkin *et al.* 2003a). Finally, an aerial survey of the northern Gulf of Alaska coastline flown in 2000 provided a minimum uncorrected count of 198 sea otters between Cape Hinchinbrook and Cape Yakataga (USGS unpublished data). Applying a correction factor of 2.16 (CV = 0.38) for this observer conducting sea otter aerial surveys produces an adjusted estimate of 428 (CV = 0.38).

The most recent population estimates for survey areas within the Southcentral Alaska stock are presented in Table 1. Combining the adjusted estimates for these areas results in a total estimate of 18,297 sea otters for the Southcentral Alaska stock.

Table 1. Population estimates for the Southcentral Alaska stock of northern sea otters. The previous stock assessment report (SAR) total is from 2008.

Survey Area	Year	Unadjusted Estimate	Adjusted Estimate	CV	N_{MIN}	Reference
Cook Inlet, Kachemak Bay excluded	2002		962	0.54	629	Bodkin *et al.* (2003b)
Kachemak Bay	2008		3,596	0.50	2,416	USFWS unpublished data
Kenai Fjords	2010		1,322	0.37	978	Coletti *et al.* (2011)
Prince William Sound	2003		11,989	0.18	10,324	Bodkin *et al.* (2003a)
North Gulf of Alaska	2000	198	428	0.38	314	USGS unpublished data
Current Total			**18,297**		**14,661**	
Previous SAR Total			15,090		12,774	

Minimum Population Estimate

The minimum population estimate (N_{MIN}) for this stock is calculated using Equation 1 from the Potential Biological Removal Guidelines (Wade and Angliss 1997): $N_{MIN} = N/\exp(0.842 \times [\ln(1+[CV(N)]^2)]^{½})$. The N_{MIN} for each survey area is presented in Table 1. The estimated N_{MIN} for the Southcentral Alaska stock is 14,661 sea otters.

Current Population Trend

All surveys analyzed for trends in abundance used methods described in Bodkin and Udevitz (1999), including use of a survey-specific correction factor to account for undetected animals, with the exception of the survey in the North Gulf of Alaska. Aerial surveys in Kachemak Bay in 2002, 2007, and 2008, indicated that the population is increasing, with an estimated annual rate of increase between 2002 and 2008 of 26% per year (USGS unpublished data, USFWS unpublished data). This rate slightly exceeds the estimated maximum productivity rates (R_{MAX}) for the species (see below). Immigration from other areas (Cook Inlet, Kenai Fjords) may have contributed to the observed increase in sea otter numbers in Kachemak Bay.

Aerial surveys in Kenai Fjords National Park in 2002, 2007, and 2010, had relatively high standard errors, but indicated overall that the population is stable and may be increasing (Coletti *et al.* 2011). Annual aerial surveys of sea otter abundance in western Prince William Sound from 1993 to 2009 (except for 2001 and 2006) identified a significant increase in abundance between 2001 and 2009 at this scale, with an average annual rate of increase from 1993 to 2009 of 2.6% (Bodkin *et al.* 2011). This trend is interpreted as strong evidence of a trajectory toward recovery of sea otter populations in Prince William Sound affected by the 1989 *Exxon Valdez* oil spill (Bodkin *et al.* 2011). Our best assessment is that the overall trend in abundance for this stock appears to be increasing at this time.

MAXIMUM NET PRODUCTIVITY RATE

Estes (1990) estimated a population growth rate of 17 to 20% per year for four northern sea otter populations expanding into unoccupied habitat. Although maximum productivity rates (R_{MAX}) have not been measured throughout much of the sea otter's range in Alaska, in the absence of more detailed information, the rate of 20% calculated by Estes (1990) is considered the best available estimate of R_{MAX}. There is insufficient information available to estimate the current net productivity rate for this population stock.

POTENTIAL BIOLOGICAL REMOVAL

Under the Marine Mammal Protection Act (MMPA), the potential biological removal (PBR) is defined as *the maximum number of animals, not including natural mortalities, that may be removed from a marine mammal stock while allowing that stock to reach or maintain its optimal sustainable population.* Potential biological removal is the product of the minimum population estimate (N_{MIN}), one-half the maximum theoretical net productivity rate, and a recovery factor (F_R): PBR = N_{MIN} x 0.5 R_{MAX} x F_R. The recovery factor for this stock is 1.0 (Wade and Angliss 1997) as population levels have remained stable with a known human take. Thus, for the Southcentral stock of sea otters, PBR = 1,466 animals (14,661 x 0.5 (0.2) x 1.0).

ANNUAL HUMAN CAUSED MORTALITY

Fisheries Information

A complete list of fisheries and marine mammal interactions is published annually by the National Oceanic and Atmospheric Administration (NOAA) Fisheries, the most recent of which was published on August 29, 2013 (78 FR 53336). Numerous fisheries exist within the range of the Southcentral Alaska stock of northern sea otters. Two have been identified as interacting with this stock, the Prince William Sound drift gillnet fishery with an estimated 537 vessels and/or persons participating, and the Cook Inlet salmon set gillnet fishery, with an estimated 738 participants. Additional salmon drift gillnet fisheries occur in Cook Inlet, with 589 vessels; however, with the exception of Kachemak Bay, all of the fishing effort involving salmon drift and set gillnet fisheries in Cook Inlet occurs north of the range of sea otters from the Southcentral

Alaska stock (Manly 2006). Additional salmon set gillnet fisheries occur in Prince William Sound (30 participants).

While much of the salmon set gillnet effort in Cook Inlet occurs north of the range of sea otters, interactions between sea otters and fisheries are reported from the Kachemak Bay region. In July 2009, five sea otters with slashed throats were found dead on a Seldovia beach. They were believed to have been killed after being captured in a set gillnet. In July 2011, a female and pup were successfully released from a set gillnet in the Homer area. Interactions with set gillnet gear also have been observed in the Kodiak and Prince William Sound areas within the ranges of the Southwest and Southcentral Alaska stocks. Available information suggests that fisheries using other types of gear, including trawl, longline, and purse seine, appear to be less likely to have interactions with northern sea otters due to either the areas where such fisheries operate, or the specific gear used, or both.

Although commercial fisheries in Alaska have observer programs that monitor and report injury and mortality of marine mammals incidental to their operations, a reliable estimate of the levels of commercial fisheries incidental mortality and serious injury relative to the Southcentral sea otter stock cannot be made because observer coverage is not sufficient and data are not collected consistently over time. No incidents of sea otter incidental take have been observed in trawl, longline, or pot groundfish fisheries in southcentral Alaska from 1989 to 2010 (NOAA unpublished data). Sea otters are known to interact with pot fisheries in California, however, and it is possible that observer effort for pot fisheries in Alaska has been too low to detect sea otter bycatch (Hatfield *et al.* 2011). In addition to the fisheries listed above, observers monitored the Cook Inlet set gillnet and drift gillnet fisheries from 1999 to 2000 (Manly 2006). The observer coverage during both years was approximately 2 to 5%. No mortalities or injuries of sea otters were reported by fisheries observers for the Cook Inlet set gillnet and drift gillnet fisheries for this period. On several occasions, sea otters were observed within 10 meters (approximately 33 ft) of gillnet gear, but did not become entangled. No other fisheries operating in the region of the Southcentral Alaska stock were monitored by observer programs from 1992 through 2010. Prior to the implementation of the NOAA Fisheries observer program, studies were conducted on sea otter interactions with the drift net fisheries in western Prince William Sound from 1988 to 1990, and no mortalities were observed (Wynne 1990, Wynne *et al.* 1991).

An additional source of information on the number of sea otters killed or injured incidental to commercial fishery operations in Alaska is found in fisher self-reports required of vessel owners by NOAA Fisheries. In 1990, fisher self-report records show one mortality and four injuries due to gear interaction, and three injuries due to deterrence in the Prince William Sound drift gillnet fishery. Self-reports were not available for 1994 and 1995. Credle *et al.* (1994) considered fisher self-reports to be a minimum estimate of incidental take as these data are most likely negatively biased.

In summary, between 2006 and 2010, there were five records of incidental take of sea otters by commercial fisheries within the range of the Southcentral stock, and, therefore, the estimated mean annual mortality and serious injury reported for the 5-year period from 2006 to 2010 is one. Observer coverage for fisheries within the range of the Southcentral stock of sea otters has been absent in some fisheries and low in others, particularly with respect to the set and drift gillnet fisheries that are recognized as interacting with this stock, and current estimates of sea otter bycatch are not available. Self-reporting is not sufficiently reliable to replace observer effort. Additionally, assessment of injury and mortality in sea otters that interact with fisheries is difficult. Information is, therefore, insufficient to determine whether or not the total fishery mortality and serious injury for the Southcentral Alaska stock of the northern sea otter is insignificant and is approaching a zero mortality and serious injury rate.

Oil Spills

Activities associated with exploration, development, and transport of oil and gas resources can adversely impact sea otters and nearshore coastal ecosystems in Alaska. Sea otters rely on air trapped in their fur for warmth and buoyancy. Contamination with oil drastically reduces the insulative value of the pelage, and consequently, sea otters are among the marine mammals most likely to be detrimentally affected by contact with oil. It is believed that sea otters can survive low levels of oil contamination (<10% of body surface), but that greater levels (>25%) will lead to death (Costa and Kooyman 1981, Siniff *et al.* 1982). Vulnerability of sea otters to oiling was demonstrated by the 1989 *Exxon Valdez* oil spill in Prince William Sound. Total estimates of mortality for the Prince William Sound area vary from 750 (range 600 to 1,000; Garshelis 1997) to 2,650 otters (range 500 to 5,000; Garrot *et al.* 1993). Statewide, it is estimated that 3,905 sea otters (range 1,904 to 11,257) died in Alaska as a result of the spill (DeGange *et al.*1994). At present, although abundance of sea otters in some oiled areas of Prince William Sound remains below pre-spill estimates, evidence from ongoing studies suggests that sea otters numbers are increasing, a trend interpreted as evidence of a trajectory toward recovery of spill-affected sea otter populations in western Prince William Sound (Bodkin *et al.* 2002, Stephensen *et al.* 2001, Bodkin *et al.* 2011, Monson *et al.* 2011).

Within the range of the Southcentral Alaska sea otter stock, oil and gas development and production occurs only in Cook Inlet. As of 2011, 16 offshore oil platforms operated in Cook Inlet, and two more are slated to begin operations in 2012. A Federal lease sale in Cook Inlet may be held in 2012 to 2017, if industry interest is sufficient. Tankering of North Slope crude oil occurs regularly through the waters of Prince William Sound with no major oil spills since the *Exxon Valdez*. While the catastrophic release of oil has the potential to take large numbers of sea otters, there is no evidence that other effects (such as disturbance) associated with routine oil and gas development and transport have had a direct impact on the Southcentral Alaska sea otter stock.

Information on oil spills compiled by the Alaska Department of Environmental Conservation from 2006 to 2010 indicates that an average of four spills of crude oil occurred each year in the marine environment within the range of the Southcentral Alaska stock of sea otters. Crude oil spills ranged in size from less than 4 liters to 760 liters (approximately 1 gallon to 200 gallons), with a mean size of about 41.8 liters (approximately 11 gallons). In addition to spills directly associated with the development, production, and transport of crude oil, each year numerous spills of non-crude oil products in the marine environment occur from ships and shore facilities throughout Southcentral Alaska. During the same time period and area, there was an average of about 62 spills of non-crude oil per year, ranging in size from less than 4 to 24,320 liters (approximately 1 to 6,400 gallons). The majority of the non-crude oil spills were small, with a mean size of about 380 liters (100 gallons) and a median size of 4 liters (approximately one gallon). There is no indication that these small-scale spills have an impact on the Southcentral Alaska stock of northern sea otters.

Subsistence/Native Harvest Information

The MMPA exempts Alaska Natives from the prohibition on take of marine mammals, provided such taking is not wasteful and is done for subsistence use or for creating and selling authentic handicrafts or clothing. According to the U.S. Fish and Wildlife Service's (Service) Law Enforcement records from 2006 to 2010, individuals were prosecuted for unlawful possession, transport, or sale of 14 sea otter hides or skulls taken within the range of the Southcentral Alaska stock. Data for subsistence harvest of sea otters in southcentral Alaska are collected by a mandatory Marking, Tagging and Reporting Program administered by the Service since 1988. Figure 2 provides a summary of subsistence harvest information for the Southcentral stock from 1989 to 2010. The mean reported annual subsistence take during the past five complete calendar years (2006 to 2010) was 293 animals. Reported age composition during this period was 93% adults, 6% subadults, and 1% pups. Sex composition during the past five years was 72% males, 23% females, and 5% of unknown sex. The majority of the harvest over the past five years has occurred in northern and eastern Prince William Sound.

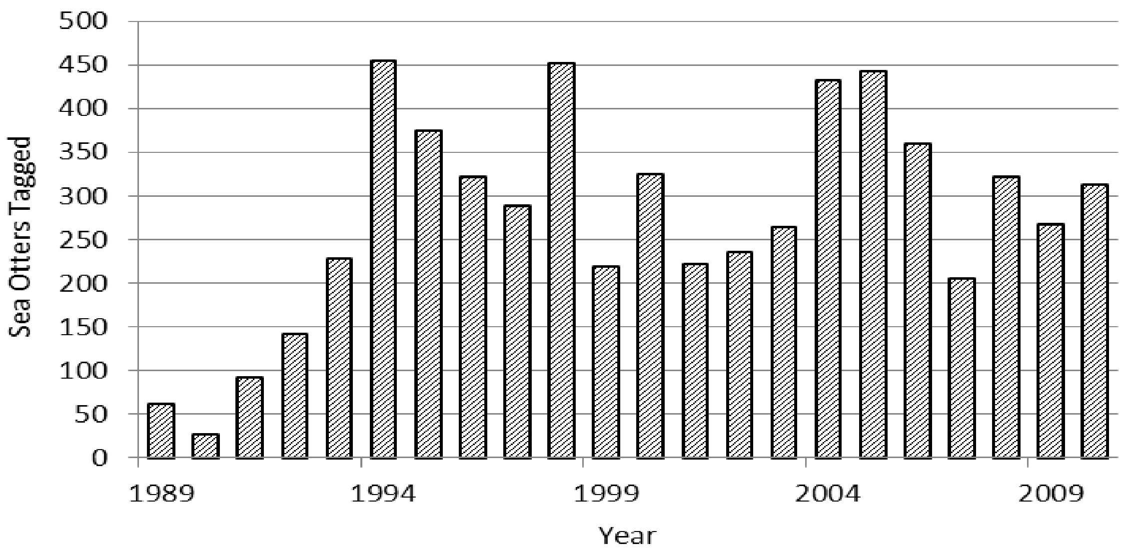

Figure 2. Reported subsistence harvest of northern sea otters from the Southcentral Alaska stock, 1989 to 2010.

Research and Public Display

During 2006 to 2010, four orphaned sea otter pups from the Southcentral Alaska stock were captured, rehabilitated, and placed for public display. During the same time period, 142 sea otters were captured and released for scientific research in Prince William Sound. There were no reported injuries and/or mortalities related to these activities.

Other Factors

In August 2006, the Working Group on Marine Mammal Unusual Mortality Events reviewed information provided by the Service and declared that a dramatic increase in sea otter strandings since 2002 constituted an Unusual Mortality Event (UME) in accordance with Section 404 of the MMPA. The disease complex that typifies this UME is caused by a *Streptococcus infantarius* infection and has been observed over a broad geographic range in Alaska, with the majority of cases identified from Kachemak Bay in the Southcentral Alaska stock. The dramatic increase of sea otter strandings in Kachemak Bay is now thought to be due to a rapidly increasing otter population in the bay combined with more community effort to report strandings. Testing and analysis are still being conducted to pinpoint the cause of this leading source of mortality. However, it is thought that the *Streptococcus infantarius* infection may be the result of immunosuppression due to an emerging virus in the Alaska population. At this time it is unclear what impact this has had, or will have, on the population.

Since 2002, the Service has undertaken a health and disease study of northern sea otters from all three Alaskan stocks. On average, the Service conducts about 100 necropsies a year on sea otter carcasses to determine cause of death, disease incidence, and status of general health parameters. Boat strike is a recurring cause of death across all three stocks. However, it has been determined in most of these cases that although trauma was the ultimate cause of death, there was a contributing factor, such as disease or biotoxin exposure, which incapacitated the animal and made it more vulnerable to boat strike.

STATUS OF STOCK

The known level of direct human-caused mortality within the Southcentral Alaska stock does not exceed the PBR level, and the Southcentral Alaska stock is neither listed as “depleted” under the MMPA nor listed as “threatened” or “endangered” under the U. S. Endangered Species Act of 1973, as amended. The known level of direct human-caused mortality is 293 otters per year. It would require an annual rate of fisheries-associated mortality and serious injury of over 1,170 otters per year for the total amount of direct human-caused mortality to exceed PBR for this stock. Despite uncertainties regarding fisheries mortality and serious injury, we believe that it is unlikely this level of take is occurring at present. Therefore, the Southcentral Alaska stock of the northern sea otter is classified as non-strategic. In addition, although the Service does not currently know the OSP for this stock, based on the known population level and our estimate of growth and considering the known level of human-caused mortality, we have determined that this stock is increasing and that human-caused mortality and serious injury is not likely to cause the stock to be reduced or to decrease its growth rate. Therefore, we would not expect the current level of human-caused mortality and serious injury to cause this stock to be reduced below its plausible OSP.

CITATIONS

Bodkin, J.L. and B.E. Ballachey. 2010. Modeling the effects of mortality on sea otter populations. U.S. Geological Survey Scientific Investigations Report 2010–5096. 12pp.

Bodkin, J.L. and D.H. Monson. 2002. Sea otter population structure and ecology in Alaska. Arctic Research of the United States 16:31-35.

Bodkin, J.L., and M.S. Udevitz. 1999. An aerial survey method to estimate sea otter abundance. Pages 13-26 *In:* G.W. Garner *et al.*, editors. Marine Mammal Survey and Assessment Methods. Balkema, Rotterdam, Netherlands.

Bodkin, J.L., B.E. Ballachey, T.A. Dean, A.K. Fukuyama, S.C. Jewett, L.M. McDonald, D.H. Monson, C.E. O'Clair, and G.R. VanBlaricom. 2002. Sea otter population status and the process of recovery from the *Exxon Valdez* spill. Marine Ecology Progress Series. 241:237-253.

Bodkin, J.L., B.E. Ballachey, T.A. Dean, and D. Esler. 2003a. Patterns and Processes of

Population Change in Selected Nearshore Vertebrate Predators. *Exxon Valdez* Restoration Project //423. Final Report. 83pp.

Bodkin, J.L., B.E. Ballachey, and G.G. Esslinger. 2011. Trends in sea otter population abundance in western Prince William Sound, Alaska: Progress toward recovery following the 1989 *Exxon Valdez* oil spill: U.S. Geological Survey Scientific Investigations Report 2011.

Bodkin, J.L., G.G. Esslinger, and D.H. Monson. 2004. Foraging depths of sea otters and implications to coastal marine communities. Marine Mammal Science 20:305-321.

Bodkin, J.L., D.H. Monson, and G.E. Esslinger. 2003b. A report on the results of the 2002 Kenai Peninsula and Lower Cook Inlet aerial sea otter survey. USGS Report. 10pp.

Coletti, H.A., J.L. Bodkin, and G.G. Esslinger. 2011. Sea otter abundance in Kenai Fjords National Park: Results from the 2010 aerial survey: Southwest Alaska Network Inventory and Monitoring Program. National Resource Technical Report NPS/SWAN/NRTR—21010/417. National Park Service, Fort Collins, Colorado.

Costa, D.P., and G.L. Kooyman. 1981. Effects of oil contamination in the sea otter *Enhydra lutris.* Outer Continental Shelf Environmental Assessment Program. NOAA Final Report. La Jolla, California.

Credle, V.A., D.P. DeMaster, M.M. Merlein, M.B. Hanson, W.A. Karp, and S.M. Fitzgerald (eds.). 1994. NMFS observer programs: minutes and recommendations from a workshop held in Galveston, Texas, November 10-11, 1993. U.S. Department of Commerce, NOAA Tech. Memo. NMFS-OPR-94-1. 96 pp.

DeGange, A.R., A.M. Doroff, and D.H. Monson. 1994. Experimental recovery of sea otter carcasses at Kodiak Island, Alaska, following the *Exxon Valdez* oil spill. Marine Mammal Science 10:492-496.

Dizon, A.E., C. Lockyer, W.F. Perrin, D.P. DeMaster, and J. Sisson. 1992. Rethinking the stock concept: a phylogeographic approach. Conservation Biology 6(1):24-36.

Estes, J.A. 1990. Growth and equilibrium in sea otter populations. Journal of Animal Ecology 59:385-401

Garrott, R.A., L.L. Eberhard, and D.M. Burn. 1993. Mortality of sea otters in Prince William Sound following the *Exxon Valdez* oil spill. Marine Mammal Science 9:343-359.

Garshelis, D.L., and J.A. Garshelis. 1984. Movements and management of sea otters in Alaska. Journal of Wildlife Management 48(3):665-678.

Garshelis, D.L. 1997. Sea otter mortality estimated from carcasses collected after the *Exxon Valdez* oil spill. Conservation Biology 11(4):905-916.

Gorbics, C.S., and J.L. Bodkin. 2001. Stock structure of sea otters (*Enhydra lutris kenyoni*) in Alaska. Marine Mammal Science 17(3):632-647.

Hatfield, B.B., J.A. Ames, J.A. Estes, M.T. Tinker, A.B. Johnson, M.M. Staedler, M.D. Harris. 2011. Sea otter mortality in fish and shellfish traps: estimating potential impacts and exploring possible solutions. Endangered Species Research 13:219–229.

Jameson, R.J., K.W. Kenyon, A.M. Johnson, and H.M. Wight. 1982. History and status of translocated sea otter populations in North America. Wildlife Society Bulletin 10:100-107.

Jameson, R.J. 1989. Movements, home ranges, and territories of male sea otters off central California. Marine Mammal Science 5:159-172.

Johnson, A.M. 1982. Status of Alaska sea otter populations and developing conflicts with fisheries. Pages 293-299 *In*: Transactions of the 47th North American Wildlife and Natural Resources Conference, Washington D.C.

Kenyon, K.W. 1969. The sea otter in the eastern Pacific Ocean. North American Fauna 68. U.S. Department of the Interior, Washington D.C.

Manly, B.F.J. 2006. Incidental catch and interactions of marine mammals and birds in the Cook Inlet salmon driftnet and setnet fisheries. Western EcoSystems Technology Inc. Report. Cheyenne, Wyoming, USA. 98pp.

Monson, D., D. Doak, B.E. Ballachey, and J.L. Bodkin. 2011. Could residual oil from the *Exxon Valdez* spill create a long-term population ‘‘sink’’ for sea otters in Alaska? Ecological Applications 21:2917–2932.

NOAA unpublished data. Available from NOAA, Alaska Fisheries Science Center, National Marine Mammal Laboratory, 7600 Sand Point Way NE, Seattle, WA 98115.

Ralls, K., T. Eagle, and D.B. Siniff. 1988. Movement patterns and spatial use of California sea otters, *In:* Siniff, D.B., and Ralls, K., eds. Final Report on Contract No. 14-12-001-3003, Population status of California sea otters: Minerals Management Service, Los Angeles, CA, pp. 33-63.

Riedman, M.L., and J. A. Estes. 1990. The sea otter *Enhydra lutris*: behavior, ecology, and natural history. Biological Report; 90 (14). U.S. Fish and Wildlife Service.

Simenstad, C.A., J.A. Estes, and K.W. Kenyon. 1978. Aleuts, sea otters, and alternate stable-state communities. Science 200:403-411. 127 pp.

Siniff, D.B., T.D. Williams, A.M. Johnson, and D.L. Garshelis. 1982. Experiments on the response of sea otters *Enhydra lutris* to oil contamination. Biological Conservation 23: 261-272.

Stephensen, S.W., D.B. Irons, S.J. Kendall, B.K. Lance, and L.L. MacDonald. 2001. Marine bird and sea otter population abundance of Prince William Sound, Alaska: trends following the T/V *Exxon Valdez* oil spill, 1989-2000. Restoration Project 00159 Annual Report. USFWS Migratory Bird Management, Anchorage, Alaska. 114 pp.

USFWS unpublished data. Available from USFWS, Marine Mammals Management, Anchorage Regional Office, 1011 E Tudor Road, Anchorage, AK 99503.

USGS unpublished data. Available from the USGS Alaska Science Center, 4210 University Drive, Anchorage, AK 99508.

Wade, P.R., and R. Angliss. 1997. Guidelines for assessing marine mammal stocks: report of the GAMMS workshop April 3-5, 1996, Seattle, Washington. U.S. Department of Commerce, NOAA Technical Memo. NMFS-OPR-12. 93 pp.

Wynne, K.M., D. Hicks, and N. Munro. 1991. 1990 salmon gillnet fisheries observer programs in Prince William Sound and south Unimak Alaska. Final Report, Saltwater, Inc., Anchorage, Alaska. 65 pp.

Wynne, K.M. 1990. Marine mammal interactions with salmon drift gillnet fishery on the Copper River Delta, Alaska: 1988 and 1989. Alaska Sea Grant Technical Report AK-SG-90-05. 36 pp.

Revised: April 2014

NORTHERN SEA OTTER (*Enhydra lutris kenyoni*):

Southwest Alaska Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Sea otters occur in nearshore coastal waters of the U.S. along the North Pacific Rim from the Aleutian Islands to California. The species is most commonly observed within the 40-meter (approximately 12.2 feet) depth contour because the animals require frequent access to benthic foraging habitat in subtidal and intertidal zones (Reidman and Estes 1990). Sea otters are not migratory and generally do not disperse over long distances, although movements of tens of kilometers (tens of miles) are common (Garshelis and Garshelis 1984). Annual home range sizes of adult sea otters are relatively small, with male territories ranging from 4 to 11 square kilometers (km^2) (approximately 10.5 to 28.5 square miles [mi^2]) and adult female home ranges from a few to 24 km^2 (approximately 62 mi^2) (Garshelis and Garshelis 1984; Ralls *et al*. 1988; Jameson 1989). Due to their benthic foraging, sea otter distribution is largely limited by their ability to dive to the sea floor (Bodkin *et al.* 2004).

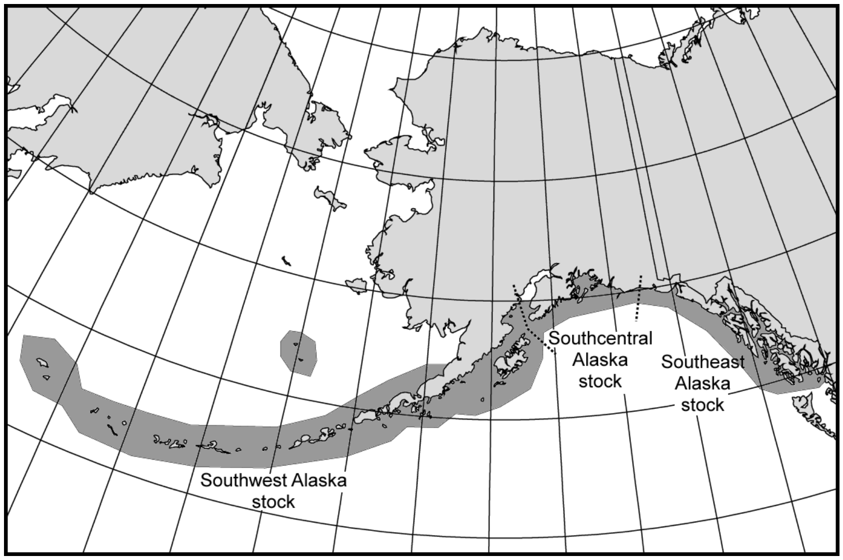

Figure 1. Approximate distribution and stock boundaries of northern sea otters in Alaska waters (shaded area).

The spatial scale at which sea otter populations are managed remains an important, although largely unexplored issue (Bodkin and Ballachey 2010) deserving further study. Bodkin and Ballachey (2010) used models of sea otter mortality to show that range-wide reductions and extirpations during the commercial fur trade of the 18th and 19th centuries occurred not simply because of excessive harvest, but because the harvest was not allocated proportional to the abundance and distribution of sea otters. This process of serial depletion was facilitated by the relatively sedentary nature of sea otters. To reduce the risk of overexploitation, sea otters must be managed on a spatial scale compatible with their well-known behavioral and reproductive biology (Bodkin and Monson 2002), incorporating traits such as home range and movements. These proposed scales for management are much smaller than the currently recognized stocks.

Gorbics and Bodkin (2001) applied the phylogeographic approach of Dizon *et al.* (1992) and used the best available data at the time to identify three sea otter stocks in Alaska: Southeast, Southcentral, and Southwest. The ranges of these stocks are defined as follows: (1) Southeast

Alaska stock extends from Dixon Entrance to Cape Yakataga; (2) Southcentral Alaska stock extends from Cape Yakataga to Cook Inlet including Prince William Sound, the Kenai Peninsula coast, and Kachemak Bay; and (3) Southwest Alaska stock includes the Alaska Peninsula and Bristol Bay coasts, and the Aleutian, Barren, Kodiak, and Pribilof Islands (Figure 1). This stock assessment report is focused on the Southwest stock of sea otters in Alaska.

POPULATION SIZE

Historically, sea otters occurred across the North Pacific Rim, ranging from Hokkaido, Japan, through the Kuril Islands, the Kamchatka Peninsula, the Commander Islands, the Aleutian Islands, peninsular and south coastal Alaska, and south to Baja California, Mexico (Kenyon 1969). In the early 1700s, the worldwide population was estimated to be between 150,000 (Kenyon 1969) and 300,000 individuals (Johnson 1982). Prior to large-scale commercial exploitation, indigenous peoples of the North Pacific hunted sea otters. Although it appears that harvests may have periodically led to local reductions of sea otters (Simenstad *et al.* 1978), the species remained abundant throughout its range until the mid-1700s. Following the arrival in Alaska of Russian explorers in 1741, extensive commercial harvest of sea otters over the next 150 years resulted in the near extirpation of the species. When sea otters were afforded protection by the International Fur Seal Treaty in 1911, probably fewer than 2,000 animals remained in thirteen remnant colonies (Kenyon 1969). Population recovery began following legal protection. As part of efforts to re-establish sea otters in portions of their historical range, otters from Amchitka Island and Prince William Sound were translocated to other areas in the

1960s and 1970s, including to southeast Alaska (Jameson *et al.* 1982). Sea otters have since recolonized much of their historical range in Alaska.

The most recent abundance estimates for survey areas within the Southwest Alaska stock are presented in Table 1. The estimate for the Katmai area has been added since the previous stock assessment report. Aerial surveys along the shorelines of the Aleutian Islands in April 2000 resulted in a count of 2,442 sea otters in the nearshore waters (Doroff *et al.* 2003). Comparison of aerial and skiff survey counts at six islands in 2000 was used to calculate a correction factor of 3.58 for this aerial survey, which resulted in an adjusted population estimate of 8,742 sea otters (CV= 0.22; Doroff *et al.* 2003).

In May 2000, a survey of offshore areas along the north Alaska Peninsula from Unimak Island to Cape Seniavin produced an abundance estimate of 4,728 sea otters (CV= 0.33; Burn and Doroff 2005). A similar survey of offshore areas along the south Alaska Peninsula from False Pass to Pavlov Bay conducted in summer 2001 resulted in a population estimate of 1,005 sea otters (CV= 0.81; Burn and Doroff 2005). Although a correction factor to account for sightability was not calculated during this survey, Evans *et al.* (1997) used a similar twin-engine aircraft flying at the same altitude and air speed to calculate a correction factor of 2.38 (CV = 0.09). Using this correction factor produced adjusted estimates of 11,253 (CV = 0.34) and 2,392 (CV = 0.82) for the north and south Alaska Peninsula offshore areas, respectively.

Table 1. Population estimates for the Southwest Alaska stock of northern sea otters. The previous stock assessment report (SAR) total is from 2008.

Survey Area	Year	Unadjusted Estimate	Adjusted Estimate	CV	N_{min}	Reference
Aleutian Islands	2000	2,442	8,742	0.22	7,309	Doroff *et al*. (2003)
North Alaska Peninsula	2000	4,728	11,253	0.34	8,535	Burn and Doroff (2005)
South Alaska Peninsula - Offshore	2001	1,005	2,392	0.82	1,311	Burn and Doroff (2005)
South Alaska Peninsula - Shoreline	2001	2,651	6,309	0.09	5,865	Burn and Doroff (2005)
South Alaska Peninsula - Islands	2001	402	957	0.09	889	Burn and Doroff (2005)
Unimak Island	2001	42	100	0.09	93	USFWS unpublished data
Kodiak Archipelago	2004		11,005	0.19	9,361	USFWS unpublished data
Katmai	2008		7,095	0.13	6,362	Coletti *et al*. (2009)
Kamishak Bay	2002		6,918	0.32	5,340	Bodkin *et al*. (2003)
Current Total			**54,771**		**45,064**	
Previous SAR Total			47,676		38,703	

In 2001, aerial surveys along the shoreline of the south Alaska Peninsula from Seal Cape to Cape Douglas recorded 2,651 sea otters (Burn and Doroff 2005). Additional aerial surveys of the south Alaska Peninsula island groups (Sanak, Caton, and Deer Islands, and the Shumagin and Pavlov Island groups) and a survey of Unimak Island, recorded 402 otters for the south Alaska Peninsula island groups and 42 animals for Unimak Island. Applying the same correction factor of 2.38 from Evans *et al.* (1997) produced adjusted estimates of 6,309 (CV = 0.09), 957 (CV = 0.09) and 100 (CV = 0.09) for the south Alaska Peninsula shoreline, south Alaska Peninsula islands, and Unimak Island, respectively.

An aerial survey of the Kodiak Archipelago conducted in 2004 resulted in an estimate of 11,005 sea otters (CV = 0.19; USFWS unpublished data). The methods used in this survey follow those of Bodkin and Udevitz (1999), which include the calculation of a survey-specific correction factor for animals undetected by observers. An aerial survey of Katmai National Park in 2009, also using the Bodkin-Udevitz method, resulted in an estimate of 7,095 sea otters (CV = 0.13; Coletti *et al.* 2009). Finally, an aerial survey of Kamishak Bay and western Cook Inlet conducted in June 2002 resulted in an estimate of 6,918 sea otters (CV = 0.32; Bodkin *et al.* 2003). This survey also used the methods of Bodkin and Udevitz (1999).

Combining the adjusted estimates for these areas, as summarized in Table 1, results in a total estimate of 54,771 sea otters for the Southwest Alaska stock. This estimated population size for the Southwest Alaska stock is slightly higher than in the 2008 stock assessment report due to the addition of an estimate for Katmai, which was surveyed in 2009 for the first time.

Minimum Population Estimate

The minimum population estimate (N_{MIN}) for this stock is calculated using Equation 1 from the Potential Biological Removal Guidelines (Wade and Angliss 1997): $N_{MIN} = N/\exp(0.842 \times [\ln(1+[CV(N)]^2)]^{1/2})$. The N_{MIN} for each survey area is presented in Table 1. The estimated N_{MIN} for the entire Southwest Alaska stock is 45,064 sea otters.

Current Population Trend

In spring 2000, the U.S. Fish and Wildlife Service (Service) repeated an aerial survey that had previously been conducted in 1992 and observed widespread declines throughout the Aleutian Islands, with the greatest decreases occurring in the central Aleutians. The uncorrected count for the area was 2,442 animals, indicating that sea otter populations had declined 70% since 1992 (Doroff *et al.* 2003). Burn *et al.* (2003) estimated that the sea otter population in the Aleutians in 2000 may have been reduced to less than 10% of the carrying capacity for the area. With the exception of the Kodiak Archipelago, which was surveyed in 2004, there have been no new large-scale abundance surveys for sea otters in southwest Alaska since the stock assessment report of August 2002.

On-going efforts to monitor trends in abundance include repeated skiff surveys at selected islands (index sites) in the Aleutian Islands. A Bayesian state-space trend analysis (Clark and Bjornstad 2004) developed using all available data compiled from skiff surveys around five islands in the western Aleutian Islands from 1993 to 2003 indicated that the population trends during this time period were strongly negative, with an average rate of decline of approximately 20% per year (USFWS 2013b, USGS unpublished data). Population trends changed during the period 2003 to 2011, with an average growth rate of approximately 0. Some variation in trends was evident but the trends were consistent among islands. These results suggest that population trends have stabilized in the western Aleutian Islands over the last 5 to 8 years, although there is still no evidence of recovery (USFWS 2013a, USFWS 2013b, USGS unpublished data).

Unlike in the Aleutian Islands and along the western Alaska Peninsula, sea otters in other areas within the range of the Southwest stock do not appear to have undergone a population decline over the past 20 years. Sea otter numbers in the Kodiak Archipelago, the Alaska Peninsula coast from Castle Cape to Cape Douglas, and Kamishak Bay in lower western Cook Inlet are stable and may be increasing (Coletti *et al.* 2009, Estes *et al*. 2010, USFWS 2013a, USGS unpublished data).

The estimated population size for the Southwest Alaska stock is slightly higher than in the previous stock assessment report due to the addition of Katmai, which was surveyed in 2009 for the first time. However, the overall sea otter population size in southwest Alaska has declined by more than 50% since the mid-1980s, and there is no evidence of recovery. Although current numbers are well below historical levels, the overall population trend for the Southwest Alaska stock is believed to have stabilized.

MAXIMUM NET PRODUCTIVITY RATE

Estes (1990) estimated a population growth rate of 17 to 20% per year for four northern sea otter populations expanding into unoccupied habitat. Although maximum productivity rates (R_{MAX}) have not been measured throughout much of the sea otter's range in Alaska, in the absence of more detailed information, the rate of 20% calculated by Estes (1990) is considered the best available estimate of R_{MAX}. There is insufficient information available to estimate the current net productivity rate for this population stock.

POTENTIAL BIOLOGICAL REMOVAL

Under the Marine Mammal Protection Act (MMPA), the potential biological removal (PBR) is defined as *the maximum number of animals, not including natural mortalities, that may* *be removed from a marine mammal stock while allowing that stock to reach or maintain its optimal sustainable population.* The potential biological removal is the product of the minimum population estimate (N_{MIN}), one-half the maximum theoretical net productivity rate, and a recovery factor (F_R): PBR = N_{MIN} x 0.5 R_{MAX} x F_R. In August 2005, sea otters in southwest Alaska were listed as a threatened distinct population segment (DPS) under the Endangered Species Act of 1973, as amended (70 FR 46366; August 9, 2005) (ESA). Although Wade and Angliss (1997) provide a default recovery factor of 0.5 as a guideline for threatened species, a lower value may be considered appropriate in the case of a declining population. Therefore, for the Southwest Alaska stock, which has experienced a decline, we are taking a more conservative approach and have set the recovery factor at the default value for an endangered species (0.1). The calculated PBR for this stock is 450 sea otters per year (45,064 x 0.5 (0.2) x 0.1).

ANNUAL HUMAN CAUSED MORTALITY

Fisheries Information

A complete list of fisheries and marine mammal interactions is published annually by the National Oceanic and Atmospheric Administration (NOAA) Fisheries, the most recent of which was published on August 29, 2013 (78 FR 53336). Numerous fisheries exist within the range of the Southwest Alaska stock of northern sea otters, with the only one identified as interacting with this stock being the Kodiak salmon set gillnet fishery, with an estimated 188 vessels and/or persons participating. Additional salmon set gillnet fisheries occur in Bristol Bay (982 participants) and the Alaska Peninsula/Aleutian Islands (114 participants). Although no interactions with salmon drift gillnets have been identified for this stock, interactions have been observed in Prince William Sound with the Southcentral Alaska stock. Salmon drift gillnet fisheries occur in Bristol Bay (1,863 vessels) and the Alaska Peninsula/Aleutian Islands (162 vessels). Although both salmon set and drift gillnet fisheries occur in Cook Inlet, most of the fishing effort for these gillnet fisheries occurs north of the range of sea otters from the Southwest Alaska stock. Available information suggests that fisheries using other types of gear, including trawl, longline, and purse seine, appear to be less likely to have interactions with northern sea otters due to either the areas where such fisheries operate, or the specific gear used, or both.

Although commercial fisheries in Alaska have observer programs that monitor and report injury and mortality of marine mammals incidental to their operations, a reliable estimate of the levels of commercial fisheries incidental mortality and serious injury relative to the Southwest sea otter stock cannot be made because observer coverage is not sufficient and data are not collected consistently over time. Observer data were summarized from 1989 to 2010 (Perez 2003, Perez 2006, Perez 2007, NOAA unpublished data) for Bering Sea, Aleutian Islands, and Gulf of Alaska trawl, longline, and pot groundfish fisheries. During this period, no sea otters were taken in any trawl or longline fisheries. In 1992, a total of eight sea otters were observed caught in the Pacific cod pot fishery in the Aleutian Islands. Observer records indicate that those takes occurred in nearshore waters that had been closed to fishing. This explains why no additional take of sea otters was observed in legal pot fisheries, which took place in other areas, through 2010 (Perez 2006, Perez 2007, NOAA unpublished data). Sea otters are known to interact with pot fisheries in California, and it is possible that observer effort for pot fisheries in Alaska has been too low to detect sea otter bycatch (Hatfield *et al.* 2011).

The NOAA Fisheries conducted a marine mammal observer program for the Kodiak salmon set gillnet fishery during the 2002 and 2005 fishing seasons. This fishery has a seasonal component, occurring only during the summer months. In 2002, four entanglement events were observed in this fishery (Manly *et al.* 2003). Two of these events required intervention to untangle the otter from the net, and the other two were able to escape by themselves. In none of these instances was there any sign of external injuries. The sea otter by-catch in this fishery was estimated at 62 otters during the 2002 fishing season. Although no serious injuries or mortalities were observed in this small sample size of observed entanglements, it is reasonable to assume that some of these otters may have suffered injury as a result of entanglement in set gillnet fisheries. In fact, there was one self-report of an otter killed during the 2002 fishing season. Results from the 2005 Kodiak salmon set gillnet fishery indicate entanglement of one otter that subsequently released itself from the net, although it was not clear if this was a sea otter or river otter (Manly 2007). Assuming that this animal was a sea otter, the total by-catch in this fishery would be estimated at 28 animals during the 2005 season. Based on these results, it would appear that although entanglement of sea otters does occur in this fishery, the rate of mortalityor serious injury is low.

An additional source of information on the number of sea otters killed or injured incidental to commercial fishery operations in Alaska are fisher self-reports required of vessel owners by NOAA Fisheries. In 1997, fisher self-reports indicated one sea otter caught in the Bering Sea and Aleutian Island groundfish trawl fishery; however, it is unclear if the animal was alive when caught. Credle *et al.* (1994) considered fisher self-reports to be a minimum estimate of incidental take as these data are most likely negatively biased. Observer coverage for fisheries within the range of the Southwest stock of sea otters has been absent in some fisheries and low in others, particularly with respect to the set and drift gillnet fisheries that are recognized as interacting with this stock, and current estimates of sea otter bycatch are not available. Self-reporting is not sufficiently reliable to replace observer effort. Additionally, assessment of injury and mortality in sea otters that interact with fisheries is difficult. Information is, therefore, insufficient to determine whether or not the total fishery mortality and serious injury for the Southwest Alaska stock of the northern sea otter is insignificant and is approaching a zero mortality and serious injury rate.

Oil Spills

Activities associated with exploration, development, and transport of oil and gas resources can adversely impact sea otters and nearshore coastal ecosystems in Alaska. Sea otters rely on air trapped in their fur for warmth and buoyancy. Contamination with oil drastically reduces the insulative value of the pelage, and consequently sea otters are among the marine mammals most likely to be detrimentally affected by contact with oil. It is believed that sea otters can survive low levels of oil contamination (<10% of body surface), but that greater levels (>25%) will lead to death (Costa and Kooyman 1981, Siniff *et al.* 1982). Vulnerability of sea otters to oiling was demonstrated by the 1989 *Exxon Valdez* oil spill in Prince William Sound. Estimates of mortality for the Prince William Sound area vary from 750 otters (range 600 to 1,000; Garshelis 1997) to 2,650 otters (range 500 to 5,000; Garrott *et al.* 1993). Statewide, 3,905 sea otters (range 1,904 to 11,257) were estimated to have died in Alaska as a result of the spill (DeGange *et al.* 1994). At present, although abundance of sea otters in some oiled areas of Prince William Sound remains below pre-spill estimates, evidence from ongoing studies suggests that sea otters numbers in this area are increasing, a trend interpreted as strong evidence of a trajectory toward recovery of spill-affected sea otter populations in western Prince William Sound (Bodkin *et al.* 2002, Stephensen *et al.* 2001, Bodkin *et al.* 2011).

Within the range of the Southwest Alaska sea otter stock, oil and gas development and production occurs only in Cook Inlet. As of 2011, 16 offshore oil platforms operated in Cook Inlet, and two more are slated to begin operations in 2012. A Federal lease sale in lower Cook Inlet is planned for the fall of 2013. Although the amount of oil transported in southwest Alaska is relatively small, the *Exxon Valdez* oil spill demonstrated that spilled oil can travel long distances and take large numbers of sea otters far from the point of initial release. The grounding in 2004 of the freighter *Selendang Ayu* on Unalaska Island, within the range of this stock, released 1,219,800 liters (approximately 321,000 gallons) of non-crude oil and caused at least two sea otter mortalities (USFWS unpublished data). While the catastrophic release of oil has the potential to take large numbers of sea otters, there is no evidence that other effects (such as disturbance) associated with routine oil and gas development and transport have had a direct impact on the Southwest Alaska sea otter stock.

Information on oil spills compiled by the Alaska Department of Environmental Conservation from 2006 to 2010 indicates that there were no reported spills of crude oil in southwest Alaska during that time period. In addition to spills that may occur in association with the development, production, and transport of crude oil, each year numerous spills of non-crude oil products in the marine environment occur from ships and shore facilities throughout southwest Alaska. During that same time period, an average of 64 non-crude oil spills occurred each year, ranging in size from less than 4 to 551,000 liters (approximately1 to 145,000 gallons).

The majority of these spills were small, with a mean size of about 3,500 liters (approximately 921 gallons) and a median size of 15 liters (approximately 2 gallons). There is no indication that these small-scale spills have an impact on the Southwest Alaska stock of northern sea otters.

Subsistence/Native Harvest Information

The MMPA exempts Alaska Natives from the prohibition on take of marine mammals, provided such taking is not wasteful and is done for subsistence use or for creating and selling authentic handicrafts or clothing. In addition, section 10(e) of the ESA allows for take of listed species for primarily subsistence purposes under certain circumstances. According to the Service's Law Enforcement records, there were no prosecutions from 2006 to 2010 for unlawful take, possession, transport, or sale of sea otters or sea otter hides taken within the range of the Southwest Alaska stock. Data for subsistence harvest of sea otters in southwest Alaska are collected by a mandatory Marking, Tagging and Reporting Program administered by the Service since 1988. Figure 2 provides a summary of harvest information for the Southwest stock from 1989 through 2010. The mean reported annual subsistence take during the past five complete calendar years (2006-2010) was 76 animals. Reported age composition during this period was 84% adults, 12% subadults, 1% pups, and 3% unknown. Sex composition during the past five years was 77% males, 19% females, and 4% unknown. The majority of this harvest (83%) comes from the Kodiak Archipelago; areas within the stock that show signs of continued population declines have little to no record of subsistence harvest.

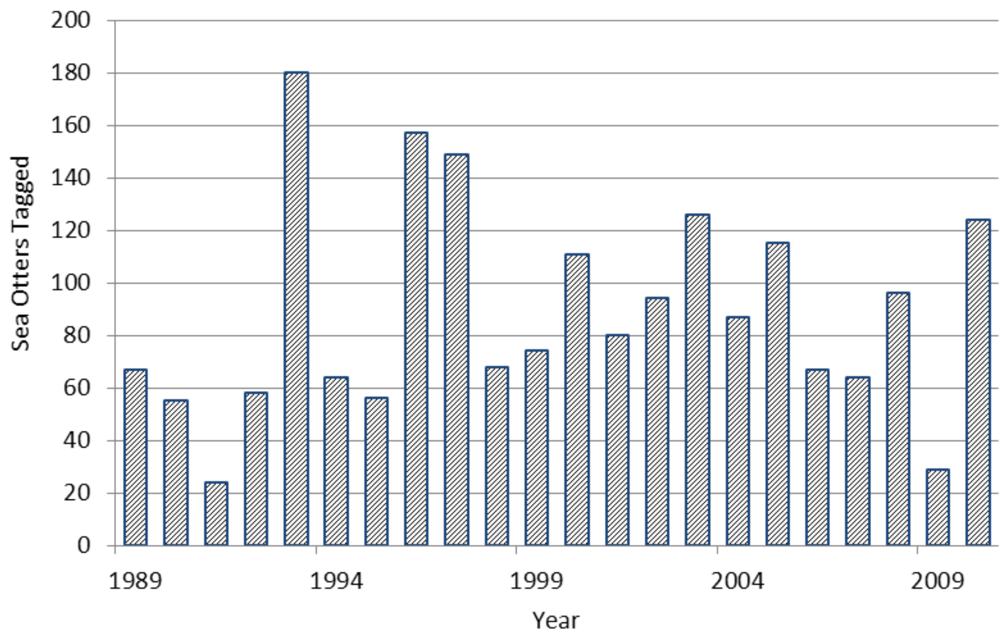

Figure 2. Reported subsistence harvest of northern sea otters from the Southwest Alaska stock, 1989-2010.

Research and Public Display

During 2006 to 2010, one orphaned sea otter pup from the Southwest Alaska stock was captured, rehabilitated, and placed for public display. During this period, a total of 65 otters were live-captured from this stock and released for research purposes. The captures occurred in the vicinities of Kukak Bay (Katmai National Park and Preserve coast), Unga Island (Shumagin Island group), and Dolgoi Island (Pavlov Island group). There were no reported injuries and/or mortalities related to these activities.

Other Factors

Each year, several thousand commercial vessels of varying sizes traverse the North Pacific Great Circle Route between North America and Asia, carrying a variety of cargoes. Vessels generally pass through the Aleutian Islands twice, through Unimak Pass to the east and near Buldir Island to the west. A risk assessment for the area concluded that while a majority of the vessel traffic along the Great Circle Route passes through the region without making any port calls, accidents involving these vessels have the potential to significantly and adversely impact coastal and marine ecosystems, economies, and human activities in the region (Aleutian Islands Risk Assessment Project Management Team 2011). Previous vessel accidents in the Aleutian Islands have resulted in loss of cargo, oil spills, and loss of life. The remoteness, limited infrastructure, and severe weather of the region often limit the potential to mitigate or respond to incidents. Overall, both the total number of accidents and the total risk of a bunker oil spill in the region are predicted to increase (Aleutian Islands Risk Assessment Project Management Team 2011).

Since 2002 the Service has undertaken a health and disease study of northern sea otters from all three Alaskan stocks. On average, the Service conducts about 100 necropsies a year on sea otter carcasses to determine cause of death, disease incidence and status of general health parameters. Boat strike is a recurring cause of death across all three stocks. However, it has been determined in most of these cases that although trauma was the ultimate cause of death, there was a contributing factor, such as disease or biotoxin exposure, which incapacitated the animal and made it more vulnerable to boat strike.

In August 2006, the Working Group on Marine Mammal Unusual Mortality Events reviewed information provided by the Service, and declared that a dramatic increase in sea otter strandings since 2002 constituted an Unusual Mortality Event (UME) in accordance with section 404 of the MMPA. The disease that typifies this UME is caused by a *Streptococcus infantarius* infection and has been observed over a broad geographic range in Alaska, including a few cases from southwest Alaska; however, the majority of cases have come from Kachemak Bay in the Southcentral Alaska stock. It is not clear if the observed stranding pattern is representative of overall sea otter mortality, or an artifact of having a well-developed stranding network in the Kachemak Bay area. The Service will continue to work with NOAA Fisheries and the Alaska Sea Life Center to develop the infrastructure for a State-wide marine mammal stranding network in Alaska.

STATUS OF STOCK

On August 9, 2005, the Southwest Alaska DPS of the northern sea otter was listed as "threatened" under the ESA, and it is, therefore, classified as a strategic stock under the MMPA.

CITATIONS

Aleutian Islands Risk Assessment Management Team. 2011. Aleutian Islands Risk Assessment Project:Phase A Summary Report. *http://www.aleutiansriskassessment.com/documents.htm*.

Bodkin, J.L. and B.E. Ballachey. 2010. Modeling the effects of mortality on sea otter populations. U.S. Geological Survey Scientific Investigations Report 2010–5096. 12pp.

Bodkin, J.L. and D.H. Monson. 2002. Sea otter population structure and ecology in Alaska. Arctic Research of the United States 16:31-35.

Bodkin, J.L. and M.S. Udevitz. 1999. An aerial survey method to estimate sea otter abundance. Pages 13-26 *In:* G.W. Garner et al., editors. Marine Mammal Survey and Assessment Methods. Balkema, Rotterdam, Netherlands.

Bodkin, J.L., B.E. Ballachey, T.A. Dean, A.K. Fukuyama, S.C. Jewett, L.M. McDonald, D.H. Monson, C.E. O'Clair, and G.R. VanBlaricom. 2002. Sea otter population status and the process of recovery from the *Exxon Valdez* spill. Marine Ecology Progress Series 241:237-253.

Bodkin, J.L., B.E. Ballachey, and G.G. Esslinger. 2011. Trends in sea otter population abundance in western Prince William Sound, Alaska: Progress toward recovery following the 1989 *Exxon Valdez* oil spill: U.S. Geological Survey Scientific Investigations Report 2011.

Bodkin, J.L., G.G. Esslinger, and D.H. Monson. 2004. Foraging depths of sea otters and implications to coastal marine communities. Marine Mammal Science 20:305-321.

Bodkin, J.L., D.H. Monson, and G.E. Esslinger. 2003. A report on the results of the 2002 Kenai Peninsula and Lower Cook Inlet aerial sea otter survey. USGS Report. 10pp.

Burn, D.M., A.M. Doroff, and M.T. Tinker. 2003. Estimated carrying capacity and pre-decline abundance of sea otters (*Enhydra lutris kenyoni*) in the Aleutian Islands. Northwestern Naturalist 84:145-148.

Burn, D.M. and A.M. Doroff. 2005. Decline in sea otter (*Enhydra lutris*) populations along the Alaska Peninsula, 1986-2001. Fishery Bulletin 103:270-279.

Clark, J.S. and O.N. Bjornstad. 2004. Population time series: Process variability, observation errors, missing values, lags, and hidden states. Ecology 85:3140-3150.

Coletti, H., J. Bodkin, T. Dean, and K. Kloecker. 2009. Nearshore marine vital signs monitoring in the Southwest Alaska Network of National Parks. Natural Resource Technical Report NPS/SWAN/NRTR-2009/252. National Park Service, Fort Collins, Colorado.

Costa, D.P. and G.L. Kooyman. 1981. Effects of oil contamination in the sea otter *Enhydra lutris.* Outer Continental Shelf Environmental Assessment Program. NOAA Final Report. La Jolla, California.

Credle, V.A., D.P. DeMaster, M.M. Merlein, M.B. Hanson, W.A. Karp, and S.M. Fitzgerald (eds.). 1994. NMFS observer programs: minutes and recommendations from a workshop held in Galveston, Texas, November 10-11, 1993. U.S. Department of Commerce, NOAA Tech. Memo. NMFS-OPR-94-1. 96 pp.

DeGange, A.R., A.M. Doroff, and D.H. Monson. 1994. Experimental recovery of sea otter carcasses at Kodiak Island, Alaska, following the *Exxon Valdez* oil spill. Marine Mammal Science 10:492-496.

Dizon, A.E., C. Lockyer, W.F. Perrin, D.P. DeMaster, and J. Sisson. 1992. Rethinking the stock concept: a phylogeographic approach. Conservation Biology 6:24-36.

Doroff, A.M., J.A. Estes, M.T. Tinker, D.M. Burn, and T.J. Evans. 2003. Sea otter population declines in the Aleutian Archipelago. Journal of Mammalogy 84:55-64.

Estes, J.A. 1990. Growth and equilibrium in sea otter populations. Journal of Animal Ecology 59:385-401.

Estes, J.A., J.L. Bodkin, and M.T. Tinker. 2010. Threatened Southwest Alaska sea otter stock: delineating the causes and constraints to recovery of a keystone predator in the North Pacific Ocean. North Pacific Research Board Final Report 717. 117 p.

Evans, T.J., D.M. Burn and A.R. DeGange. 1997. Distribution and relative abundance of sea otters in the Aleutian Archipelago. USFWS Marine Mammals Management Technical Report, MMM 97-5. 29 pp.

Garrott, R.A., L.L. Eberhard, and D.M. Burn. 1993. Mortality of sea otters in Prince William Sound following the *Exxon Valdez* oil spill. Marine Mammal Science 9:343-359.

Garshelis, D.L. and J.A. Garshelis. 1984. Movements and management of sea otters in Alaska. Journal of Wildlife Management 48:665-678.

Garshelis, D.L. 1997. Sea otter mortality estimated from carcasses collected after the *Exxon Valdez* oil spill. Conservation Biology 11(4):905-916.

Gorbics, C.S. and J.L. Bodkin. 2001. Stock structure of sea otters (*Enhydra lutris kenyoni*) in Alaska. Marine Mammal Science 17:632-647.

Hatfield, B.B., J.A. Ames, J.A. Estes, M.T. Tinker, A.B. Johnson, M.M. Staedler, M.D. Harris. 2011. Sea otter mortality in fish and shellfish traps: estimating potential impacts and exploring possible solutions. Endangered Species Research 13:219–229.

Jameson, R.J., K.W. Kenyon, A.M. Johnson, and H.M. Wight. 1982. History and status of translocated sea otter populations in North America. Wildlife Society Bulletin 10:100-107.

Jameson, R.J. 1989. Movements, home ranges, and territories of male sea otters off central California. Marine Mammal Science 5:159-172.

Johnson, A.M. 1982. Status of Alaska sea otter populations and developing conflicts with fisheries. Pages 293-299 *In*: Transactions of the 47th North American Wildlife and Natural Resources Conference, Washington D.C.

Kenyon, K.W. 1969. The sea otter in the eastern Pacific Ocean. North American Fauna 68. U.S. Department of the Interior, Washington D.C.

Manly, B.F.J., A.S. Van Atten, K.J. Kuletz, and C. Nations. 2003. Incidental catch of marine mammals and birds in the Kodiak Island set gillnet fishery in 2002. Western EcoSystems Technology Inc. Report. Cheyenne, Wyoming, USA. 91pp.

Manly, B.F.J. 2007. Incidental take and interactions of marine mammals and birds in the Kodiak Island salmon set gillnet fishery, 2002 and 2005. Western EcoSystems Technology Inc. Report. Cheyenne, Wyoming, USA. 221pp.

NOAA unpublished data. Available from NOAA, Alaska Fisheries Science Center, National Marine Mammal Laboratory, 7600 Sand Point Way NE, Seattle, WA 98115.

Perez, M.A. 2003. Compilation of marine mammal incidental catch data for domestic and joint venture groundfish fisheries in the U.S. EEZ of the North Pacific, 1989-2001. NOAA Technical Memorandum NMFS-AFSC-138. 145 pp.

Perez, M.A. 2006. Analysis of marine mammal by-catch data from the trawl, longline, and pot groundfish fisheries of Alaska, 1998-2004, defined by geographic area, gear type, and target groundfish catch species. NOAA Technical Memorandum NMFS-AFSC-167. 194 pp.

Perez, M.A. 2007. By-catch of marine mammals in the groundfish fisheries of Alaska, 2006. Alaska Fisheries Science Center Processed Draft Report. 67pp.

Ralls, K., T. Eagle, and D.B. Siniff. 1988. Movement patterns and spatial use of California sea otters, *In:* Siniff, D.B., and Ralls, K., eds. Final Report on Contract No. 14-12-001-3003, Population status of California sea otters: Minerals Management Service, Los Angeles, CA, pp. 33-63.

Riedman, M.L. and J.A. Estes. 1990. The sea otter *Enhydra lutris*: behavior, ecology, and natural history. U.S. Fish and Wildlife Service Biological Report 90(14).

Simenstad, C.A., J.A. Estes, and K.W. Kenyon. 1978. Aleuts, sea otters, and alternate stable-state communities. Science 200:403-411. 127 pp.

Siniff, D.B., T.D. Williams, A.M. Johnson, and D.L. Garshelis. 1982. Experiments on the response of sea otters *Enhydra lutris* to oil contamination. Biological Conservation 23:261-272.

Stephensen, S.W., D.B. Irons, S.J. Kendall, B.K. Lance, and L.L. MacDonald. 2001. Marine bird and sea otter population abundance of Prince William Sound, Alaska: trends following the T/V *Exxon Valdez* oil spill, 1989-2000. Restoration Project 00159 Annual Report. USFWS Migratory Bird Management, Anchorage, Alaska. 114 pp.

USFWS 2013a. Southwest Alaska Distinct Population Segment of the Northern Sea Otter (*Enhydra lutris kenyoni*) Recovery Plan. U.S. Fish and Wildlife Service, Region 7, Alaska. 171 pp.

USFWS 2013b. Southwest Alaska DPS of the Northern Sea Otter *(Enhydra lutris kenyoni)* 5-Year Review: Summary and Evaluation. U.S. Fish and Wildlife Service, Region 7, Alaska. 18 pp.

USFWS unpublished data. Available from USFWS, Marine Mammals Management, Anchorage Regional Office, 1011 E Tudor Road, MS-341, Anchorage, AK 99503.

USGS unpublished data. Available from the USGS Alaska Science Center, 4210 University Drive, Anchorage, AK 99508.

Wade, P.R. and R. Angliss. 1997. Guidelines for assessing marine mammal stocks: report of the GAMMS workshop April 3-5, 1996, Seattle, Washington. U.S. Department of Commerce, NOAA Technical Memo. NMFS-OPR-12. 93 pp.

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July 2021

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