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

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

PINNIPEDS

CETACEANS - U.S. WEST COAST

CETACEANS – HAWAI‘I & WESTERN PACIFIC

APPENDICES

PREFACE

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

Pacific region stock assessments include those studied by the Southwest Fisheries Science Center (SWFSC, La Jolla, CA), the Pacific Islands Fisheries Science Center (PIFSC, Honolulu, HI), the Marine Mammal Laboratory (NMML, Seattle, WA), and the Northwest Fisheries Science Center (NWFSC, Seattle, WA). The 2023 Draft Pacific marine mammal stock assessments include revised reports for 30 stocks under NMFS jurisdiction, including 8 "strategic" stocks: Hawaiian monk seal, Southern Resident killer whale, CA-OR-WA Sperm Whale, Eastern North Pacific blue whale, CA-OR-WA fin whale, Eastern North Pacific sei whale, Hawai‘i pelagic false killer whale, and Hawaiian Islands Insular false killer whale. Information on sea otters, manatees, walrus, and polar bears are published separately by the US Fish and Wildlife Service.

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

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

Revised ~~7/15/2014~~ 01/11/2023

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

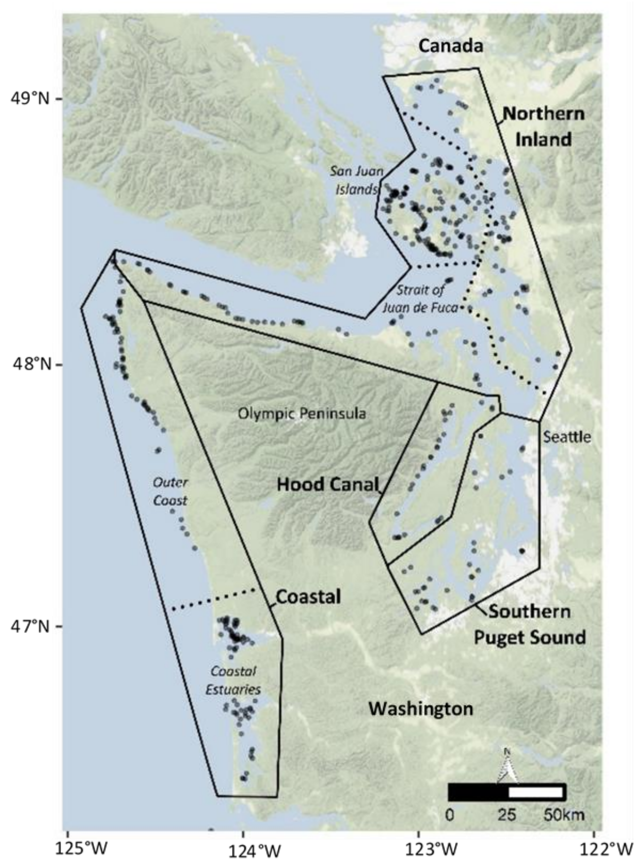

Figure 1. Census locations of harbor seals in Washington. Stock boundaries are indicated by solid lines separating the four stocks. The dotted lines indicate sub-regions within the stocks but are not estimated separately. The Coastal stock is the Washington portion of the Oregon/Washington Coast stock.

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

POPULATION SIZE

Aerial surveys of harbor seals in Washington were conducted during the pupping season in ~~1999~~2019, during which time the total numbers of hauled-out seals (including pups) were counted. ~~In 1999, the~~The ~~mean~~ count of harbor seals from photographic images ~~occurring in Washington's inland waters~~ was ~~7,213~~10,752 ~~(CV=0.14)~~ in Washington Northern Inland Waters, ~~1,025~~1,580 ~~(CV=0.14)~~ in Southern Puget Sound, and ~~711~~ 731~~(CV=0.14)~~ in Hood Canal, (~~Jeffries~~ Pearson *et al*. ~~2003~~in review).

Radio-tagging studies conducted at six locations (three Washington inland waters sites and three Oregon and Washington coastal sites) collected information on haulout patterns from 63 harbor seals in 1991 and 61 harbor seals in 1992. Data from coastal and inland sites were not significantly different and were thus pooled, resulting in a correction factor of 1.53 (CV=0.065) to account for animals in the water which are missed during the aerial surveys (Huber et al. 2001). Using this correction factor results in a population estimates of ~~11,036~~16,451 (~~7,213~~10,752 x 1.53; CV=0.~~15~~074) for the Washington Northern Inland Waters stock~~;~~ ~~1,088 (711 x 1.53; CV=0.15) for the Hood Canal stock;~~ and ~~1,568~~2,417 (~~1,025~~1,580 x 1.53; CV=0.~~15~~083) for the Southern Puget Sound stock of harbor seals ~~(Jeffries et al. 2003~~Pearson *et al.* in review). A separate haulout probability model based on London et al. (2012) was used to estimate abundance from counts in Hood Canal due to different behavior and haulout patterns for this stock compared to other inland waters stocks. This model produced an estimate of 3,363 (CV=0.16) harbor seals in Hood Canal in 2019 (Pearson *et al.* in review). ~~However, because the most recent abundance estimates are >8 years old, there are no current estimates of abundance for these stocks. Surveys of harbor seals in Washington inland waters are planned for 2013.~~

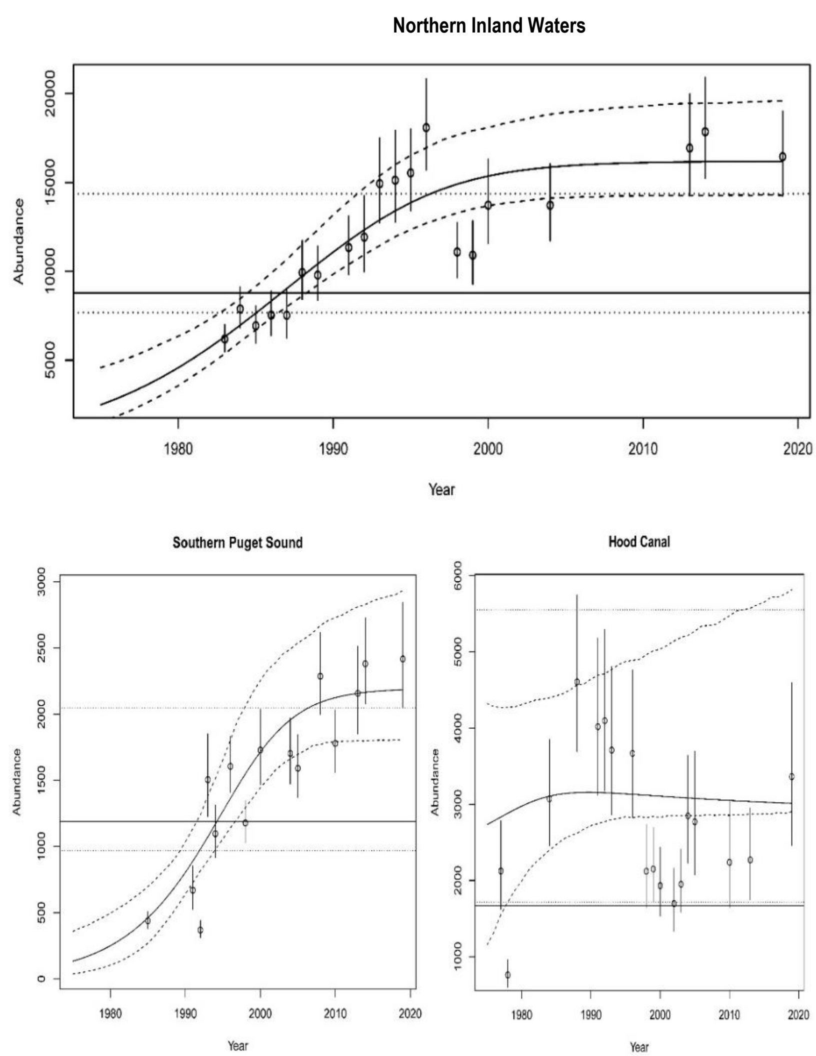

Figure 2. Generalized logistic population growth curves for the three Washington Inland Waters stocks of harbor seals: Northern Inland Waters (1983-2019), Southern Puget Sound (1985-2019), and Hood Canal (1977-2019) (Pearson et al. in review)~~, 1978-1999 (Jeffries et al. 2003)~~. Circles represent each count with 95% CI; solid curve is estimated abundance from logistic model and dashed curves are 95% CI; solid horizontal line is estimated MNPL and dashed horizontal lines are 95% CI for MNPL. Vertical scale is different in each panel.

Minimum Population Estimate

Minimum population estimates from 2019 were based the 20th percentile of seal counts that were corrected for haulout probability (see "Population Size" above). Resulting minimum population estimates (N_{MIN}) for harbor seal stocks in Washington inland waters are: 15,462 for Washington Northern Inland Waters; 2,253 for Southern Puget

Sound; and 2,940 for Hood Canal.~~No current information on abundance is available to obtain a minimum population estimate for the Washington Inland Waters stock of harbor seals.~~

Current Population Trend

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

~~Between 1983 and 1996, the annual rate of increase for this stock was 6% (Jeffries et al. 1997). The peak count occurred in 1996 and, b~~Based on ~~a~~ fitted generalized logistic models (Fig. 2), the population trends are ~~is thought to be~~ stable for the Washington Northern Inland Waters and Southern Puget Sound stocks (Pearson *et al*. in review). The trend for Hood Canal is too variable, due to limited data, to determine the current status (Pearson *et al*. in review). ~~(Jeffries et al. 2003). In the absence of recent abundance estimates, the current population trend is unknown.~~

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

~~From 1991 to 1996, counts of harbor seals in Washington State have increased at an annual rate of 10% (Jeffries et al. 1997). Because the population was not at a very low level by 1991, the observed rate of increase may underestimate the maximum net productivity rate (R_{MAX}). When a logistic model was fit to the 1978-1999 abundance data, the resulting estimate of R_{MAX} was 12.6% (95% CI = 9.4-18.7%) (Jeffries et al. 2003).~~ Model-averaged maximum net productivity rate (R_{MAX}) estimates were highly variable for each of the stocks (Pearson *et al*. in review). However the confidence intervals included 12%, the default R_{MAX} value for pinnipeds. Because we cannot conclude that the true R_{MAX} is different from 12%, the default value will be used for all three stocks (NMFS 2023)~~This value of R_{MAX} is very close to the default pinniped maximum theoretical net productivity rate of 12% (R_{MAX}), therefore, 12% will be employed for this harbor seal stock (Wade and Angliss 1997)~~.

POTENTIAL BIOLOGICAL REMOVAL

~~Because there is no current estimate of minimum abundance, a potential biological removal (PBR) cannot be calculated for this stock.~~The Potential Biological Removal (PBR) is defined as the product of the minimum population size (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 a stock that is within its optimum sustainable population level (OSP) is 1.0 (NMFS 2016). Using this formula, the PBRs are 928 (15,462 × 0.06 × 1.0) and 135 (2,253 × 0.06 × 1.0) for the Northern Inland Waters and Southern Puget Sound harbor seal stocks, respectively. Due to the uncertainty of the status of the Hood Canal stock, we use the recovery factor of 0.5 for a stock of unknown status relative to OSP (NMFS 2016), resulting in a PBR of 88 harbor seals (2,940 × 0.06 × 0.5).

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

Historically, harbor seals were killed in several commercial salmon gillnet fisheries in Puget Sound (Erstad *et al*. 1996; Pierce *et al*. 1994, 1996). These fisheries, now collectively referred to as the "WA Puget Sound Region salmon drift gillnet" Category II fishery (NMFS List of Fisheries; 87 FR 23122, April 19, 2022) were last observed in 1993 (NMFS 2022). The first harbor seal stock assessment report for this region (Barlow *et al*. 1998) included a single harbor seal stock in Puget Sound ("Inland Washington Stock") for which total commercial gillnet mortality over several fisheries was estimated as ≥ 35.8 seals annually, with no estimate of precision and reflecting generally low (<10%) observer coverage (Erstad *et al*. 1996; Pierce *et al*. 1994, 1996). Current levels of harbor seal mortality and serious injury in commercial salmon gillnets in this region are unknown, although NMFS receives information annually for tribal fisheries (see below). All other Washington commercial salmon fisheries are classified as Category III, with a remote likelihood of, or no known interactions. ~~Fishing effort in the northern Washington marine gillnet tribal fishery is conducted within the range of the Oregon/Washington Coast and Washington Northern Inland Waters stocks of harbor seals. Some movement of animals between Washington's coastal and inland waters is likely, although data from tagging studies have not shown movement of harbor seals between the two locations (Huber et al. 2001). For the purposes of this stock assessment report, the animals taken in waters east of Cape Flattery, WA, are assumed to have belonged to the Washington Northern Inland Waters stock, and Table 1 includes data only from that portion of the fishery. There was no observer coverage in the northern Washington marine set gillnet tribal fishery in inland waters in 2007-2011; however, there were two fishermen self-reports of harbor seal deaths in this fishery in 2008 and five in 2009 (Makah Fisheries Management, unpublished data). The mean annual mortality for this fishery in 2007-2011 is 1.4 harbor seals from self-reports. Fishing effort in the northern Washington marine drift gillnet tribal fishery in inland waters is also conducted within the range of the Washington Northern Inland Waters stock of harbor seals. This fishery is not observed; however, there was one self-report of a harbor seal death in 2008 (Makah Fisheries Management, unpublished data). The mean annual mortality for this fishery in 2007-2011 is 0.2 harbor seals from self-reports. Commercial salmon drift gillnet fisheries in Washington inland waters were last observed in 1993 and 1994, with observer coverage levels typically less than 10% (Erstad et al. 1996, Pierce et al. 1994, Pierce et al. 1996, NWIFC 1995). Drift gillnet fishing effort in the inland waters has declined considerably since 1994 because far fewer vessels participate today (NMFS NW Region, unpublished data), but entanglements of harbor seals likely continue to occur. The most recent data on harbor seal mortality from commercial gillnet fisheries is included in Table 1.~~

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

~~**Fishery name**~~	~~**Years**~~	~~**Data type**~~	~~**Percent observer coverage**~~	~~**Observed mortality**~~	~~**Estimated mortality**~~	~~**Mean annual takes (CV in parentheses)**~~
~~Northern WA marine set gillnet (tribal fishery in inland waters)~~	~~2008~~ ~~2009~~	~~fisherman self-reports~~	-	~~2~~ ~~5~~	~~n/a~~ ~~n/a~~	~~1.4 (n/a)~~
~~Northern WA marine drift gillnet (tribal fishery in inland waters)~~	~~2008~~	~~fisherman self-reports~~	-	~~1~~	~~n/a~~	~~>0.2 (n/a)~~
~~WA Puget Sound Region salmon set/drift gillnet (observer programs listed below covered segments of this fishery):~~	-	-	-	-	-	-
~~Puget Sound non-treaty salmon gillnet (all areas and species)~~	~~1993~~	~~observer data~~	~~1.3%~~	~~2~~	~~n/a~~	~~see text~~
~~Puget Sound non-treaty chum salmon gillnet (areas 10/11 and 12/12B)~~[1]	~~1994~~	~~observer data~~	~~11%~~	~~1~~	~~10~~	~~see text~~[1]
~~Puget Sound treaty chum salmon gillnet (areas 12, 12B, and 12C)~~[1]	~~1994~~	~~observer data~~	~~2.2%~~	~~0~~	~~0~~	~~see text~~[1]
~~Puget Sound treaty chum and sockeye salmon gillnet (areas 4B, 5, and 6C)~~[1]	~~1994~~	~~observer data~~	~~7.5%~~	~~0~~	~~0~~	~~see text~~[1]
~~Puget Sound treaty and non-treaty sockeye salmon gillnet (areas 7 and 7A)~~[1]	~~1994~~	~~observer data~~	~~7%~~	~~1~~	~~15~~	~~see text~~[1]
~~Unknown Washington Northern Inland Waters fisheries~~	~~2007-2011~~	~~stranding data~~	~~n/a~~	~~1, 1, 1, 1, 2~~	~~n/a~~	~~≥1.2 (n/a)~~
~~Unknown Hood Canal fisheries~~	~~2007-2011~~	~~stranding data~~	~~n/a~~	~~0, 0, 0, 0, 1~~	~~n/a~~	~~≥0.2 (n/a)~~

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

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

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

Other Mortality

Non-commercial fisheries sources of harbor seal human-caused mortality and serious injury include harassment, shootings, hook and line fisheries, vehicle collisions, dog attacks, vessel strikes, marine debris entanglements, Washington state river otter fur traps, unidentified fisheries, and interactions with tribal fisheries (Carretta et al. 2023). Interactions with tribal fisheries are reported by the Northwest Indian Fisheries Commission (NWIFC) via annual reports to NMFS. Takes reported by the NWIFC lack details on animal condition, but primarily involve gillnet fisheries, a gear type where survival of pinnipeds and cetaceans is rarely observed (Carretta 2022). In this report, we assume that reported NWIFC interactions represent serious injuries or deaths with a mortality or serious injury value equal to one for purposes of assessment relative to PBR (Carretta et al. 2023). A summary of non-commercial fishery sources of mortality and serious injury for the most-recent 5-year period of 2017-2021, based on cases published in Carretta et al. (2023), and summarized by stock, is shown in Table 1.

Table 1. Summary of non-commercial sources of human-caused mortality and serious injury by source, 2017-2021. Totals for tribal fisheries are based on self-reported takes and include identification of stock areas based on the old stock boundary for Southern Puget Sound being placed at the Tacoma Narrows Bridge. This boundary has moved north to a line roughly between Apple Cove Point and Edwards Point. As a result, values for Southern Puget Sound may be underrepresented. Non-tribal sources are derived from strandings data.

	Hood Canal 5-yr total (Annual Mean)	**Southern Puget Sound** 5-yr total (Annual Mean)	**Washington Northern Inland Waters** 5-yr total (Annual Mean)
Source			
Tribal Gillnet Fisheries	5 (1.00)	21 (4.2)	140 (28)
Harassment	4 (0.8)	11 (2.2)	29 (5.8)
Shootings	0	3 (0.6)	4 (0.8)
Unidentified Gillnet Fishery	0	16 (3.2)	8 (1.6)
Vessel Strike	0	11 (2.2)	5 (1.0)
Hook and Line Fishery	0	2 (0.4)	1 (0.2)
Unidentified Fishery Interaction	0	1 (0.2)	0
Marine Debris	0	2 (0.4)	8 (1.6)
Dog Attack	0	1 (0.2)	2 (0.4)
Unidentified human interaction	1 (0.2)	0	1 (0.2)
Tribal Pot Fisheries	0	0	1 (0.2)
Vehicle collision	0	1 (0.2)	0
WA State fur trap	0	0	1 (0.2)
Totals	**10 (2.0)**	**69 (13.8)**	**200 (40.0)**

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

Subsistence Harvests by Northwest Treaty Indian Tribes

Tribal subsistence takes of harbor seals in Washington ~~this stock may occur, but no data on recent takes are available~~ have not been reported.

STATUS OF STOCK

Harbor seals are not considered to be "depleted" under the MMPA or listed as "threatened" or "endangered" under the Endangered Species Act. Based on currently available data, the minimum level of human-caused mortality and serious injury is ~~9.8~~ 40 harbor seals per year for the Washington Northern Inland Waters stock (~~2.8 from fishery sources in Table 1 + 0.6 from recreational hook and line fisheries + 6.4 from non-fishery sources~~). Annual human-caused serious injury and mortality for the Southern Puget Sound stock is 13.8 per year. Annual human-caused serious injury and mortality for the Hood Canal stock is 2.0 per year. ~~0.2 from unknown fishery sources.~~ ~~3.4,~~ ~~including 1.0 from fishery sources listed in Table 1, 0.4 from recreational hook and line fisheries, and 2.0 from non-fishery sources. PBRs cannot be calculated for these stocks because there are no current abundance estimates. Human-caused mortality relative to PBR is unknown for these stocks, but is considered to be small relative to stock size.~~ For all three harbor seal stocks, mean annual human-caused mortality and serious injury is less than PBR, ~~T~~therefore, the Washington Northern Inland Waters, Southern Puget Sound, and Hood Canal stocks of harbor seals are not classified as "strategic" stocks. ~~At present, the minimum annual fishery mortality and serious injury for these stocks (based on stranding data) are 1.2 for the Washington Northern Inland Waters stock, 0.2 for the Hood Canal stock, and 1.0 for the Southern Puget Sound stock.~~ ~~Since a PBR cannot be calculated for these stocks, fishery mortality relative to PBR is unknown.~~ The Northern Inland Waters and Southern Puget Sound stocks are ~~was previously~~ reported to be within ~~its~~ their Optimum Sustainable Population (OSP) range (~~Jeffries et al. 2003~~Pearson *et al*. in review), but ~~in the absence of recent abundance estimates, this stock's status relative to OSP is unknown~~the status of the Hood Canal stock relative to its OSP remains unknown.

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HAWAIIAN MONK SEAL (*Neomonachus schauinslandi*)

STOCK DEFINITION AND GEOGRAPHIC RANGE

Hawaiian monk seals are distributed throughout the Northwestern Hawaiian Islands (NWHI), with subpopulations at French Frigate Shoals, Laysan Island, Lisianski Island, Pearl and Hermes Reef, Midway Atoll, Kure Atoll, and Necker and Nihoa Islands. They also occur throughout the main Hawaiian Islands (MHI). Genetic variation among monk seals is extremely low and may reflect a long-term history at low population levels and more recent human influences (Kretzmann *et al*. 1997, 2001, Schultz *et al*. 2009). Though monk seal subpopulations often exhibit asynchronous variation in demographic parameters (such as abundance trends and survival rates), they are connected by animal movement throughout the species' range (Johanos *et al*. 2013). Genetic analysis (Schultz *et al*. 2011) indicates the species is a single panmictic population. The Hawaiian monk seal is therefore considered a single stock. Scheel *et al*. (2014) established a new genus, *Neomonachus*, comprising the Caribbean and Hawaiian monk seals, based upon molecular and skull morphology evidence.

POPULATION SIZE

The best estimate of the total population size is ~~1,465~~1,564 (95% confidence interval ~~1,388~~1,475 – ~~1,688~~1,719); CV = 0.~~03~~05), (Table 1, Johanos 2023~~2022~~a, b, c). In 2016, new approaches were developed to estimate Hawaiian monk seal abundance, both range-wide and at individual subpopulations (Baker *et al.* 2016, Harting *et al.* 2017). Obtaining abundance estimates for all NWHI subpopulations requires sea-going vessel support for approximately 56 days. In brief, methods for abundance estimation vary by site and year depending on the type and quantity of data available. Total enumeration is the favored method, but requires sufficient field presence to convincingly identify all the seals present, which is typically not achieved at most sites (Baker *et al*. 2006). When total enumeration is not possible, capture-recapture estimates (using Program CAPTURE) are conducted (Baker 2004; Otis *et al*. 1978, Rexstad & Burnham 1991, White *et al*. 1982). When no reliable estimator is obtainable in Program CAPTURE (i.e., the model selection criterion is < 0.75, following Otis *et al*. 1978), total non-pup abundance is estimated using pre-existing information on the relationship between proportion of the population identified and field effort hours expended (referred to as discovery curve analysis). At rarely visited sites (Necker, Nihoa, Niihau and Lehua Islands) where data are insufficient to use any of the above methods, beach counts are corrected for the proportion of seals at sea. In the MHI other than Niihau and Lehua Islands, abundance is estimated as the minimum tally of all individuals identified by an established sighting network during the calendar year. At all sites, pups are tallied. Finally, site-specific abundance estimates and their uncertainty are combined using Monte Carlo methods to obtain a range-wide abundance estimate distribution. All the above methods are described or referenced in Baker *et al*. (2016) and Harting *et al*. (2017). Note that because some of the abundance estimation methods utilize empirical distributions which are updated as new data accrue, previous years' estimates can change slightly when recalculated using these updated distributions.

~~In 2020, NMFS did not conduct field surveys in the NWHI due to the COVID pandemic. NMFS partners, including the USFWS, the State of Hawaii, the Papahānaumokuākea Marine Debris Project (PMDP), and Friends of Hawaiian Islands National Wildlife Refuges, conducted limited monk seal surveys. The most thorough monitoring in the NWHI in 2020 occurred at Midway and Kure Atolls. Total enumeration was not achieved at these sites, and because the amount and timing of survey effort was not comparable to typical years, standard abundance estimation methods (see above) could not be applied. Consequently, minimum tallies were used to represent Midway and Kure Atoll abundance in 2020.. A single count was conducted at Nihoa Island in 2020.~~ In 2021, total enumeration was not achieved at any subpopulation. Consequently, capture-recapture estimates were obtained at French Frigate Shoals, Laysan and Lisianski Islands, and at Pearl and Hermes Reef. Discovery curve analysis was used to generate abundance estimates at Midway and Kure Atolls (Table 1). Counts at Necker and Nihoa Islands are typically conducted from zero to a few times per year. Pups are born over the course of many months and have very different haulout patterns compared to older animals. Therefore, pup production at Necker and Nihoa Islands is estimated as the mean of the total pups observed in the past 5 years, excluding counts occurring early in the pupping season when most have yet to be born. ~~For the purposes of estimating total and minimum range wide abundance in 2020 for this report, 2019 values were used for subpopulations other than Nihoa Island and Kure and Midway Atolls.~~

In the MHI, NMFS collects information on seal sightings reported throughout the year by a variety of sources, including a volunteer network, the public, and directed NMFS observation effort. A small number of surveys of Ni'ihau and nearby Lehua Islands are conducted through a collaboration between NMFS, Ni'ihau residents and the

US Navy. Total MHI monk seal abundance is estimated by adding the number of individually identifiable seals documented during a calendar year on all MHI other than Ni'ihau and Lehua to an estimate for these latter two islands based on counts expanded by a haulout correction factor. A telemetry study (Wilson *et al*., 2017) found that MHI monk seals (N=23) spent a greater proportion of time ashore than Harting *et al*. (2017) estimated for NWHI seals. Therefore, the total non-pup estimate for Ni'ihau and Lehua Islands was the total beach count at those sites (less individual seals already counted at other MHI) divided by the mean proportion of time hauled out in the MHI (Wilson *et al*., 2017). The total pups observed at Ni'ihau and Lehua Islands were added to obtain the total (Table 1). ~~While NMFS surveys in 2020 were very limited, information from partners and the public were typical, such that MHI estimates were obtained.~~

Table 1. Total and minimum estimated abundance (N_{min}) of Hawaiian monk seals by location in 2021. ~~Estimates from 2020 data were available for Kure and Midway Atolls, Nihoa Island, and the MHI. Estimates from 2019 were used for all remaining subpopulations~~. The estimation method is indicated for each site. Methods used include DC: discovery curve analysis, EN: total enumeration; CR: capture-recapture; CC: counts corrected for the proportion of seals at sea; Min: minimum tally. Median values are presented. Note that the median range-wide abundance is not equal to the total of the individual sites' medians, because the median of sums may differ from the sum of medians for non-symmetrical distributions. N_{min} for individual sites are either the minimum number of individuals identified or the 20th percentile of the abundance distribution (the latter applies to Necker, Nihoa, Ni'ihau/Lehua, and range-wide).

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

Minimum Population Estimate

The total numbers of seals identified at the NWHI subpopulations other than Necker and Nihoa, and in the MHI other than Ni'ihau and Lehua, are the best estimates of minimum population size at those sites. Minimum population sizes for Necker, Nihoa, Ni'ihau, and Lehua Islands are estimated as the lower 20th percentiles of the non-pup abundance distributions generated using haulout corrections as described above, plus the pup estimates. The minimum abundance estimates for each site and for all sites combined (1,444~~1,431~~) are presented in Table 1.

Current Population Trend

Range-wide abundance estimates are available from 2013 to ~~2019~~2021~~, and a value for 2020 was generated using 2020 data where available and 2019 values elsewhere~~ (Table 1, Figure 1). While these estimates remain somewhat negatively-biased for reasons explained in Baker et al. (2016), they provided a much more comprehensive assessment of status and trends than has been previously available. A Monte Carlo approximation of the annual multiplicative rate of realized population growth during 2013-~~2020~~2021 was generated by fitting 10,000 log-linear regressions to randomly selected values from each year's abundance distributions. The median rate (and 95% confidence limits) is 1.02 (1.01, 1.03). Thus, the best estimate is that the population grew at an average rate of about 2% per year from 2013 to ~~2020~~2021. ~~Because there were no new estimates for most of the NWHI subpopulations in 2020, true uncertainty is greater than indicated by the nominal confidence intervals above.~~

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Mean non-pup beach counts are used as a long-term index of abundance for years when data are insufficient to estimate total abundance as described above. Prior to 1999, beach count increases of up to 7% annually were observed at Pearl and Hermes Reef, and this is the highest estimate of the maximum net productivity rate (R_{max}) observed for this species (Johanos ~~2022a~~2023a). Consistent with this value, a life table analysis representing a time when the MHI monk seal population was apparently expanding, yielded an estimated intrinsic population growth rate of 1.07 (Baker *et al*. 2011).

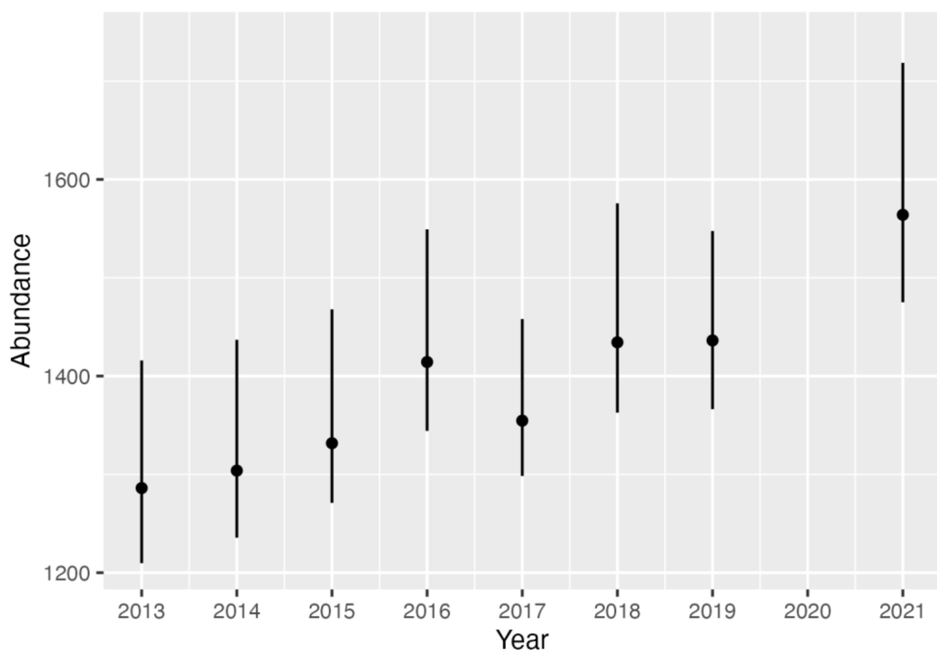

Figure 1. Range-wide abundance of Hawaiian monk seals, 2013-~~2020~~2021. Medians and 95% confidence limits are shown. Estimates prior to ~~2020~~2021 are re-estimated based on new data and represent negligible changes compared with values reported in the previous final stock assessments. ~~Note that 2019 estimates were used to represent abundance at most of the NWHI subpopulations where no information was collected in 2020~~ (Table 1).

POTENTIAL BIOLOGICAL REMOVAL

Using current minimum population size (~~1,431~~1,444), R_{max} (0.07) and a recovery factor (F_r) for ESA endangered stocks (0.1), yields a Potential Biological Removal (PBR) of ~~5.0~~5.1.

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Human-related mortality has caused two major declines of the Hawaiian monk seal (Ragen 1999). In the 1800s, this species was decimated by sealers, crews of wrecked vessels, and guano and feather hunters (Dill and Bryan 1912; Wetmore 1925; Bailey 1952; Clapp and Woodward 1972). Following a period of at least partial recovery in the first half of the 20th century (Rice 1960), most subpopulations again declined. This second decline has not been fully explained, but long-term trends at several sites appear to have been driven both by variable oceanic productivity (represented by the Pacific Decadal Oscillation) and by human disturbance (Baker *et al*. 2012, Ragen 1999, Kenyon 1972, Gerrodette and Gilmartin 1990). Currently, human activities in the NWHI are limited and human disturbance is relatively rare, but human-seal interactions, have become an important issue in the MHI. Intentional killing of seals in the MHI is an ongoing and serious concern (Table 2). In 2021, three seals were bludgeoned or shot to death, all on Molokai.

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

Year	Age/sex	Island	Cause of Death	Comments
2017	Adult female	Kauai	Trauma	Suspect intentional
2017	Juvenile female	Molokai	Blunt force trauma	Suspect intentional
2018	Juvenile female	Molokai	Blunt force trauma	Intentional
2021	Subadult male	Molokai	Blunt force trauma	Intentional
2021	Subadult male	Molokai	Blunt force trauma	Intentional
2021	Juvenile female	Molokai	Gunshot	Intentional

Harting et al. (2021) found that the 46% of carcasses of monk seals which died in the MHI during 2004-2019 were detected. Consequently, the cases in Table 2 must be considered a minimum representation of intentional killings.

Fishery Information

Fishery interactions with monk seals can include direct interaction with gear (hooking or entanglement), seal consumption of discarded or depredated catch, and competition for prey. Entanglement of monk seals in derelict fishing gear, which is believed to originate outside the Hawaiian archipelago, is described in a separate section. Fishery interactions are a serious concern in the MHI, especially involving nearshore fisheries managed by the State of Hawaii (Gobush et al. 2016). There are no fisheries operating in or near the NWHI. In ~~2020~~2021, 29 seal hookings were documented, ~~one~~two of which ~~resulted in death, another~~ were classified as serious, and 27 as non-serious, injuries. Of the non-serious injuries, ~~four~~ two would have been deemed serious had they not been mitigated (Henderson 2019a, Mercer ~~2022~~2023). Monk seals also interact with nearshore gillnets, and several confirmed deaths have resulted. In ~~2020~~2021, ~~the deaths of~~ two seals became entangled in gillnets and were released alive, and were consequently classified as non-serious injuries.~~were deemed most likely due to net drowning based on available information.~~ One adult seal was discovered swimming inside a mariculture pen and was displaced outside the pen through an existing hole. No mortality or injuries have been attributed to the MHI bottomfish handline fishery, and no interactions with longline fisheries have occurred since 1991. Consequently, these fisheries are no longer included in Table 3. Published studies on monk seal prey selection based upon scat/spew analysis and video from seal-mounted cameras revealed evidence that monk seals fed on families of bottomfish which contain commercial species (many prey items recovered from scats and spews were identified only to the level of family; Goodman-Lowe 1998, Longenecker *et al*. 2006, Parrish *et al*. 2000). Quantitative fatty acid signature analysis (QFASA) results support previous studies illustrating that monk seals consume a wide range of species (Iverson *et al*. 2011). However, deepwater-slope species, including two commercially targeted bottomfishes and other species not caught in the fishery, were estimated to comprise a large portion of the diet for some individuals. Similar species were estimated to be consumed by seals regardless of location, age or gender, but the relative importance of each species varied. Diets differed considerably between individual seals. These results highlight the need to better understand potential ecological interactions with the MHI bottomfish handline fishery.

Table 3. Summary of mortality, serious and non-serious injury of Hawaiian monk seals due to fisheries and calculation of annual mortality rate. n/a indicates that sufficient data are not available Total non-serious injuries are presented as well as, in parentheses, the number of those injuries that would have been deemed serious had they not been mitigated (e.g., by de-hooking or disentangling). Nearshore fisheries injuries and mortalities include seals entangled/drowned in nearshore gillnets and hooked/entangled in hook-and-line gear, recognizing that it is not possible to determine whether the nets or hook-and-line gear involved were being used for commercial purposes.

Fishery Name	Year	Data	% Obs.	Observed/Reported	Estimated	Non-serious	Mean

		Type	Coverage	Mortality/Serious Injury	Mortality/ Serious Injury	(Mitigated serious)	Takes (CV)
Nearshore	~~2016~~ 2017 2018 2019 2020 2021	Incidental observations of seals	None	~~0~~ 3 0 3 4 2	n/a	~~11 (6)~~ 19(6) 11(3) 17(5) 29(4) 30(4)	≥ 2.4~~2.0~~
Mariculture	~~2016~~ 2017 2018 2019 2020 2021	Incidental Observation	None	~~0~~ 1 0 0 0 0	n/a	~~0~~ 0 0 0 0 1	0.2 (2.2)
Minimum total annual takes							≥ 2.6~~2.2~~

Fishery Mortality Rate

Total fishery mortality and serious injury is not considered to be insignificant and approaching a rate of zero. Monk seals are regularly hooked and entangled in the MHI and the resulting deaths have substantially reduced the population growth rate (Harting et al. 2021). Monk seals also die from entanglement in fishing gear and other debris throughout their range (likely originating from various sources outside of Hawaii), and NMFS along with partner agencies is actively working to mitigate entanglement (see below).

Entanglement in Marine Debris

Hawaiian monk seals become entangled in fishing and other marine debris at rates higher than reported for other pinnipeds (Henderson 2001). Several hundred cases of debris entanglement have been documented in monk seals (nearly all in the NWHI), including ten documented deaths (Henderson 2001; Henderson 2019b, Mercer ~~2022~~2023). The number of marine debris entanglements documented in the past five years (Table 4) is an underestimate of the total impact of this threat because no people are present to document nor mitigate entanglements at most of the NWHI for the majority of the year. The low number of entanglements documented in 2020 is due to limited or no surveillance conducted at NWHI subpopulations due to the COVID pandemic. The fishing gear fouling the reefs and beaches of the NWHI and entangling monk seals only rarely includes types used in Hawaii fisheries. For example, trawl net and monofilament gillnet accounted for approximately 35% and 34%, respectively, of the debris removed from reefs in the NWHI by weight, and trawl net alone accounted for 88% of the debris by frequency (Donohue *et al*. 2001), despite the fact that trawl fisheries have been prohibited in Hawaii since the 1980s.

Table 4. Summary of documented marine debris entanglements of Hawaiian monk seals during the most recent five years. Total non-serious injuries are presented as well as, in parentheses, the number of those injuries that would have been deemed serious had the seals not been disentangled.

Year	Observed/Reported Mortality/Serious Injury	Non-serious (Mitigated serious)
~~2016~~	~~0~~	~~3(2)~~
2017	0	~~11~~15(8)
2018	1	15(6)
2019	0	16(10)
2020	0	5(1)
2021	0	11(6)
Minimum total annual takes	≥ 0.2	

The NMFS and partner agencies continue to mitigate impacts of marine debris on monk seals as well as turtles, coral reefs and other wildlife. Marine debris is removed from beaches and seals are disentangled during population assessment activities in the NWHI. Since 1996, annual debris survey and removal efforts in the NWHI coral reef habitat have been ongoing (Donohue *et al*. 2000, Donohue *et al*. 2001, Dameron *et al*. 2007).

Toxoplasmosis

Land-to-sea transfer of *Toxoplasma gondii*, a protozoal parasite shed in the feces of cats, is of growing concern. Although the parasite can infect many species, felids are the definitive host, meaning it can only reproduce in cats. There are no native felids in Hawaii, but several hundred thousand feral and domestic cats occur throughout the MHI. As such, all monk seal deaths attributable to toxoplasmosis are considered human caused. A case definition for toxoplasmosis and other protozoal-related mortalities was developed and retrospectively applied to 306 cases of monk seal mortality from 1982-2015 (Barbieri et al. 2016). During the past five years (~~2016~~2017-~~2020~~2021) ~~five~~ seven monk seal deaths (representing a minimum average of ~~one~~ 1.4 deaths per year) have been directly attributed to toxoplasmosis (Mercer 2021). ~~Four of the five~~Five of the seven deaths involved female seals. The number of deaths from this pathogen are likely underrepresented, given that more seals disappear each year than are found dead and examined (Harting et al. 2021), and the potential for chronic infections remains poorly understood in this species. Furthermore, *T. gondii* can be transmitted vertically from dam to fetus, and failed pregnancies are difficult to detect in wild, free-ranging animals. Unlike threats such as hook ingestion or malnutrition, which can often be mitigated through rehabilitation, options for treating seals with toxoplasmosis are challenging and two attempts have not been successful ~~(n = 2)~~. The accumulating number of monk seal deaths from toxoplasmosis in recent years is a growing concern given the increasing geographic overlap between humans, cats, and Hawaiian monk seals in the MHI.

Other Mortality

Sources of mortality that impede recovery include food limitation (see Habitat Issues), single and multiple-male intra-species aggression (mobbing), shark predation, and disease. Male seal aggression has caused episodes of mortality and injury. Past interventions to remove aggressive males greatly mitigated, but have not eliminated, this source of mortality (Johanos *et al*. 2010). Galapagos shark predation on monk seal pups has been a chronic and significant source of mortality at French Frigate Shoals since the late 1990s, despite mitigation efforts by NMFS (Gobush 2010). Besides toxoplasmosis, infectious disease effects on monk seal demographic trends are low relative to other stressors. However, a disease outbreak could be catastrophic to the immunologically naïve monk seal population. Key disease threats include West Nile virus, morbillivirus and influenza.

STATUS OF STOCK

In 1976, the Hawaiian monk seal was designated depleted under the Marine Mammal Protection Act of 1972 and as endangered under the Endangered Species Act of 1973 (NMFS 2007). Therefore, the Hawaiian monk seal is a strategic stock. The species is well below its optimum sustainable population and has not recovered from past declines. Annual human-caused mortality for the most recent 5-year period (~~2015-2019~~2017-2021) was at least ~~4.0~~5.4 animals, including fishery-related mortality in nearshore gillnets, hook-and-line gear, and mariculture (≥ ~~2.2~~2.6/yr, Table 3), intentional killings and other human-caused mortalities (≥ ~~0.6~~1.2/yr, Table 2), entanglement in marine debris (≥ 0.2/yr, Table 4), and deaths due to toxoplasmosis (≥ ~~1.0~~1.4/yr). ~~Because 4.6 is a~~The minimum rate of annual human-caused mortality~~, the true value almost certainly~~ exceeds PBR (~~5.0~~5.1).

OTHER FACTORS THAT MAY BE CAUSING A DECLINE OR IMPEDING RECOVERY

~~**Habitat Issues**~~

Poor juvenile survival rates and variability in the relationship between weaning size and survival suggest that prey availability has limited recovery of NWHI monk seals (Baker and Thompson 2007, Baker *et al*. 2007, Baker 2008). Multiple strategies for improving juvenile survival, including translocation and captive care are being implemented (Baker and Littnan 2008, Baker *et al*. 2013, Norris 2013). A testament to the effectiveness of past actions to improve survival, Harting *et al.* (2014) demonstrated that approximately one-third of the monk seal population alive in 2012 was made up of seals that either had been intervened with to mitigate life-threatening situations, or were descendants of such seals. In 2014, NMFS produced a final Programmatic Environmental Impact Statement (PEIS) on current and future anticipated research and enhancement activities and issued a permit covering the activities described in the PEIS preferred alternative. Loss of terrestrial habitat at French Frigate Shoals is a serious threat to the viability of the resident monk seal population (Baker et al. 2020). Prior to 2018, pupping and resting islets had shrunk or virtually disappeared (Antonelis *et al*. 2006). In 2018, the two remaining primary islands where pups were born at French Frigate Shoals (Trig and East Islands) were obliterated due to progressive erosion and hurricane Walaka (in September 2018). Projected increases in global average sea level are expected to further significantly reduce terrestrial habitat for monk seals in the NWHI (Baker *et al*. 2006, Reynolds *et al*. 2012).

The seawall at Tern Island, French Frigate Shoals, continues to degrade and poses an increasing entrapment hazard for monk seals and other fauna. The situation has worsened since 2012, when the USFWS ceased operations on Tern Island, thus leaving the island unmanned for most of the year. Previously, daily surveys were conducted throughout the year to remove entrapped animals. Now this only occurs when NMFS staff are on site. Furthermore, sea wall breaches are allowing sections of the island to erode and undermine buildings and other infrastructure. Several large water tanks have collapsed, exposing pipes and wiring that may entangle or entrap seals. In September 2018, hurricane Walaka exacerbated this situation by largely destroying remaining structures and strewing the resulting debris around the island. Strategies to mitigate these threats are currently under consideration. In 2020, the Papahānaumokuākea Marine Debris Project (PMDP), a non-profit organization, conducted an extensive cleanup operation at Tern Island, removing over 80,000 lb of debris and cutting multiple gaps in the seawall to provide escape routes for seals.

Goodman-Lowe (1998) provided information on prey selection using hard parts in scats and spewings. Information on at-sea movement and diving is available for seals at all six main subpopulations in the NWHI using satellite telemetry (Stewart *et al*. 2006). Cahoon (2011) and Cahoon *et al*. (2013) described diet and foraging behavior of MHI monk seals, and found no striking difference in prey selection between the NWHI and MHI.

Monk seal juvenile survival rates are favorable in the MHI (Baker *et al*. 2011). Further, the excellent condition of pups weaned on these islands suggests that there are ample prey resources available, perhaps in part due to fishing pressure that has reduced monk seal competition with large fish predators (sharks and jacks) (Baker and Johanos 2004). Yet, there are many challenges that may limit the potential for growth in this region. The human population in the MHI is approximately 1.4 million compared to fewer than 100 in the NWHI, such that anthropogenic threats in the MHI are considerable. Intentional killing of seals is a very serious concern. Also, the same fishing pressure that may have reduced the monk seal's competitors is a source of injury and mortality. Vessel traffic in the populated islands entails risk of collision with seals and impacts from oil spills. A mortality in 2015 was deemed most likely due to boat strike. Finally, as noted above, toxoplasmosis is now recognized as a serious anthropogenic threat to seals in the MHI.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

In the Pacific, harbor porpoise are found in coastal and inland waters from Point Conception, California to Alaska and across to Kamchatka and Japan (Gaskin 1984). Harbor porpoise appear to have more restricted movements along the western coast of the continental U.S. than along the eastern coast. Regional differences in pollutant residues in harbor porpoise indicate that they do not move extensively between California, Oregon, and Washington (Calambokidis and Barlow 1991). That study also showed regional differences within California (although the sample size was small). This pattern stands as a sharp contrast to the eastern coast of the U.S. and Canada where harbor porpoise are believed to migrate seasonally from as far south as the Carolinas to the Gulf of Maine and Bay of Fundy (Polacheck *et al*. 1995). A phylogeographic analysis of genetic data from northeast Pacific harbor porpoise did not show complete concordance between DNA sequence types and geographic location (Rosel 1992). However, an analysis of molecular variance (AMOVA) of the same data with additional samples found significant genetic differences for four of the six pair-wise comparisons between the four areas investigated: California, Washington, British Columbia, and Alaska (Rosel *et al*. 1995). These results demonstrated that harbor porpoise along the west coast of North America are not panmictic or migratory, and movement is sufficiently restricted that genetic differences have evolved. ~~Subsequent genetic analyses of samples ranging from Monterey Bay, California to Vancouver Island, British Columbia indicate that there is small-scale subdivision within the U.S. portion of this range Chivers *et al*., 2002, 2007).~~

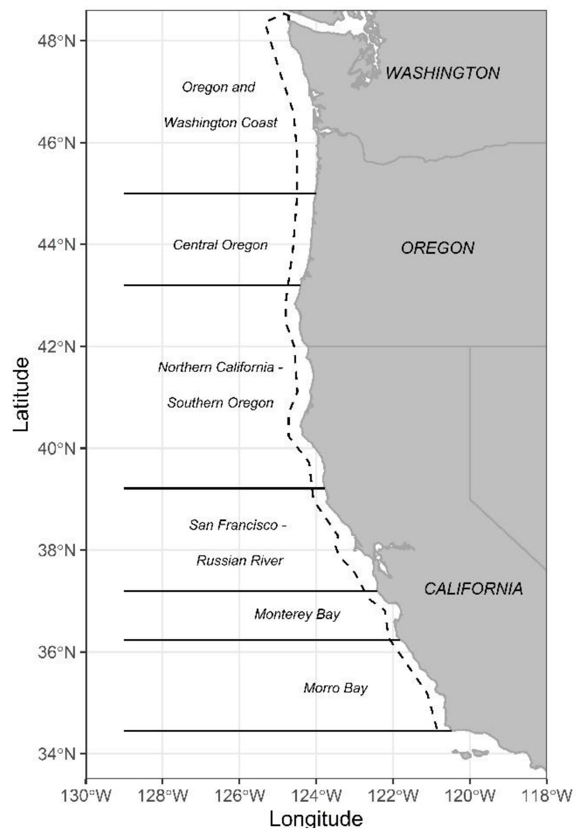

Figure 1. Stock boundaries and distributional range of harbor porpoise along the ~~California/southern Oregon coasts~~ U.S. West Coast. Dashed line represents an approximate boundary for harbor porpoise habitat (0-200m water depth) ~~along the U.S. west coast~~.

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

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

For the Marine Mammal Protection Act (MMPA) Stock Assessment Reports, Pacific coast harbor porpoise stocks include: 1) a Morro Bay stock, 2) a Monterey Bay stock, 3) a San Francisco-Russian River stock, 4) a northern California/southern Oregon stock, 5) a central Oregon stock, 6) a northern Oregon/Washington coast stock, ~~5)~~7) an Inland Washington stock, ~~6)~~8) a northern Southeast Alaska Inland Waters stock, 9) a southern Southeast Alaska Inland Waters stock, 10) a Yakutat/Southeast Alaska Offshore Waters stock, ~~a Southeast Alaska stock, 7)~~11) a Gulf of Alaska stock, and ~~8)~~12) a Bering Sea stock. The stock assessment reports for harbor porpoise stocks within waters of California, Oregon, and Washington appear in this volume. The ~~three~~ five Alaska harbor porpoise stocks are reported separately in the Stock Assessment Reports for the Alaska Region.

POPULATION SIZE

Previous estimates of abundance for California harbor porpoise were based on aerial surveys conducted between the coast and the 50-fm isobath during 1988-95 (Barlow and Forney 1994, Forney 1999). These estimates did not include an unknown number of animals found in deeper waters. Barlow (1988) found that the vast majority of harbor porpoise in California were within the 0-50-fm depth range; however, Green et al. (1992) found that 24% of harbor porpoise seen during aerial surveys of Oregon and Washington were between the 100m and 200m isobaths (55 to 109 fathoms). A systematic ship survey of depth strata out to 90 m in northern California showed that porpoise abundance declined significantly in waters deeper than 60 m (Carretta et al. 2001). Since 1999, aerial surveys extended farther offshore (to the 200m depth contour or 15 nmi distance, whichever is farther) to provide a more complete abundance estimate (Forney *et al.* 2014). ~~A recent~~ An analysis of long-term trends in the northern California portion of this harbor porpoise stock between 1989 and 2016 (Forney *et al.* ~~2019~~2020) estimated a northern California population size of ~~11,670~~12,160 (CV=~~0.659~~0.663) porpoises during 2016. More recently, Forney *et al.* (2023) estimated the abundance of harbor porpoise within the Oregon range of this stock (south of 43.2°N) to be 3,143 (CV = 0.464) based on a habitat-based density model developed from 2021-2022 aerial surveys off Oregon and Washington. Both of these ~~These~~ estimates include a correction factor of 3.42 (1/g(0); g(0)=0.292, CV=0.366) (Laake *et al*. 1997) to adjust for groups missed by aerial observers. Combining these two abundance estimates yields an overall abundance estimate of 15,303 (CV = 0.575) for the entire northern California/southern Oregon stock (Forney *et al.* 2023). ~~The most recent estimate available for the entire northern California / southern Oregon stock is the sum of the 2016 California estimate of 11,670 (Forney *et al.* 2019), plus the 2007-2011 southern Oregon estimate of 12,525 (CV = 0.48; Forney *et al.* 2014), totaling 24,195 (CV = 0.40).~~

Minimum Population Estimate

The minimum population estimate for harbor porpoise in the northern California/southern Oregon stock is ~~taken~~ calculated as the lower 20th percentile of the log-normal distribution of the abundance estimate given above, or ~~17,447~~ 9,759 animals.

Current Population Trend

A hierarchical Bayesian analysis of harbor porpoise trends for the northern California portion of this stock between 1989 and 2016 (Forney *et al.* ~~2019~~ 2020) suggests largely stable population during this period, although there is considerable uncertainty in the estimates because of limited survey coverage (Figure 2). No trend estimates are available for the entire northern California/southern Oregon range of this stock.

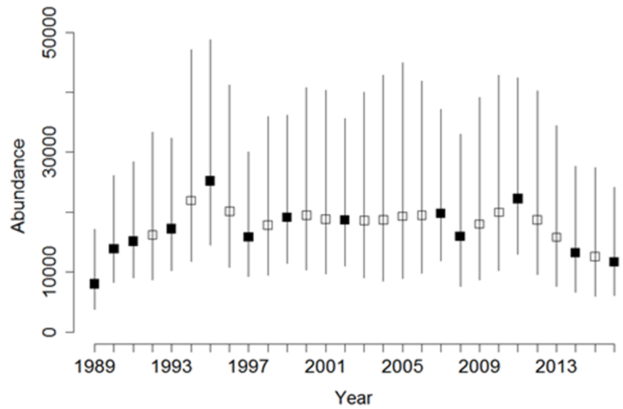

Figure 2. Population trends for the northern California portion of the Northern California / Southern Oregon harbor porpoise stock, 1989-2016 (from Forney *et al.* ~~2019~~ 2020). Estimates represent median abundance (with 95% credible intervals) for years with survey effort (solid symbols) and without survey effort (open symbols).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Based on what are argued to be biological limits of the species (i.e. females give birth first at age 4 and produce one calf per year until death), the theoretical, maximum-conceivable growth rate of a closed harbor porpoise population was estimated as 9.4% per year based on a human survivorship curve (Barlow and Boveng 1991). ~~This maximum theoretical rate represents maximum survival in a protected environment and may not be achievable for any wild population (Barlow and Boveng 1991). Woodley and Read (1991) calculate a maximum growth rate of approximately 5% per year, but their argument for this being a maximum (i.e. that porpoise survival rates cannot exceed those of Himalayan thar) is not well justified.~~ This is very similar to the growth rate of 9.6% per year (95% credible interval: 6.2% - 13.0%) estimated by Forney *et al.* (2020) for the Morro Bay harbor porpoise stock between 1991 and 2012, based on long-term aerial surveys. Because a reliable estimate of the maximum net productivity rate is not available for ~~this~~ the Northern California / Southern Oregon harbor porpoise stock, we use the default maximum net productivity rate (R_{MAX}) of 4% for cetaceans (Wade and Angliss 1997).

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (9,759~~17,447~~) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 1.0 (for a species within its Optimal Sustainable Population; see Status of Stock section; Wade and Angliss 1997), resulting in a PBR of ~~349~~195.

HUMAN-CAUSED MORTALITY

Fishery Information

There were no harbor porpoise strandings in this stock's range with evidence of fishery interactions during ~~2013-2017~~2017-2021 (Carretta *et al.* 2023).

Table 1. Summary of available information on incidental mortality and injury of harbor porpoise (northern California/southern Oregon stock) in commercial fisheries that might take this species during ~~2013-2017~~2017-2021 (Carretta *et al.* ~~2019~~2023). n/a indicates that data are not available.

Fishery Name	Year(s)	Data Type	Percent Observer Coverage	Observed Mortality	Estimated Mortality (CV in parentheses)	Mean Annual Takes (CV in parentheses)
Unknown fishery	2017-2021~~2013-2017~~	Stranding	~~n/a~~	none	n/a	0 (n/a)
Minimum total annual takes						0 (n/a)

~~**Other Mortality**~~

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

STATUS OF STOCK

Harbor porpoise in northern California/southern Oregon are not listed as threatened or endangered under the Endangered Species Act nor as depleted under the Marine Mammal Protection Act. The northern California portion of this harbor porpoise stock was determined to be within their Optimum Sustainable Population (OSP) level in the mid-1990s (Barlow and Forney 1994), based on a lack of significant anthropogenic mortality. Because there is no known human-caused mortality or serious ~~injury (≥ 0.2 harbor porpoise per year) is less than the PBR (349)~~, this stock is not considered a "strategic" stock under the MMPA, and fishery mortality can be considered insignificant and approaching zero mortality and serious injury rate. ~~Harbor porpoises are sensitive to disturbance by a variety of anthropogenic sound sources, and the limited range of several U.S. West Coast harbor porpoise stocks makes them particularly vulnerable to potential impacts (see overview in Forney *et al*. 2017). A recent habitat concern along the U.S. West coast includes the use of acoustic deterrent devices ('seal bombs') that are used in commercial fishing activities off California (Simonis *et al*. 2020), especially in the Monterey Bay region.~~

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

~~Habitat Issues~~

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

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Revised 09/26/2023 (new stock)

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

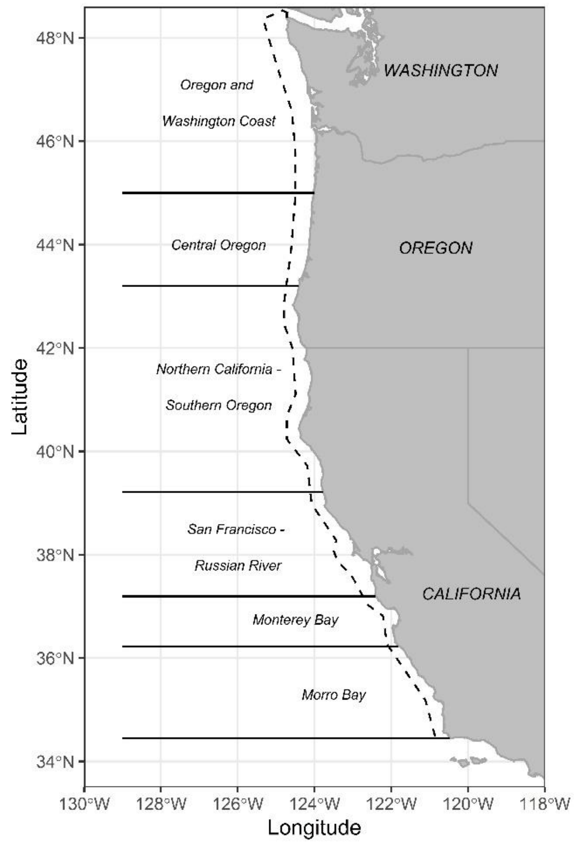

Figure 1. Stock boundaries and distributional range of harbor porpoise along the outer U.S. West Coast. Dashed line represents an approximate boundary for harbor porpoise habitat (0-200m water depth).

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

For the Marine Mammal Protection Act (MMPA) Stock Assessment Reports, Pacific coast harbor porpoise stocks include: 1) a Morro Bay stock, 2) a Monterey Bay stock, 3) a San Francisco-Russian River stock, 4) a northern California/southern Oregon stock, 5) a central Oregon stock, 6) a northern Oregon/Washington coast stock, 7) an Inland Washington stock, 8) a northern Southeast Alaska Inland Waters stock, 9) a southern Southeast Alaska Inland Waters stock, 10) a Yakutat/Southeast Alaska Offshore Waters stock, 11) a Gulf of Alaska stock, and 12) a Bering Sea stock. The stock assessment reports for harbor porpoise stocks within waters of California, Oregon, and Washington appear in this volume. The five Alaska harbor porpoise stocks are reported separately in the Stock Assessment Reports for the Alaska Region.

POPULATION SIZE

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

Minimum Population Estimate

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

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (5,303) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.5 (for a stock of unknown status, Wade and Angliss 1997), resulting in a PBR of 53.

HUMAN-CAUSED MORTALITY

Fishery Information

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

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

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

STATUS OF STOCK

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

OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

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

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

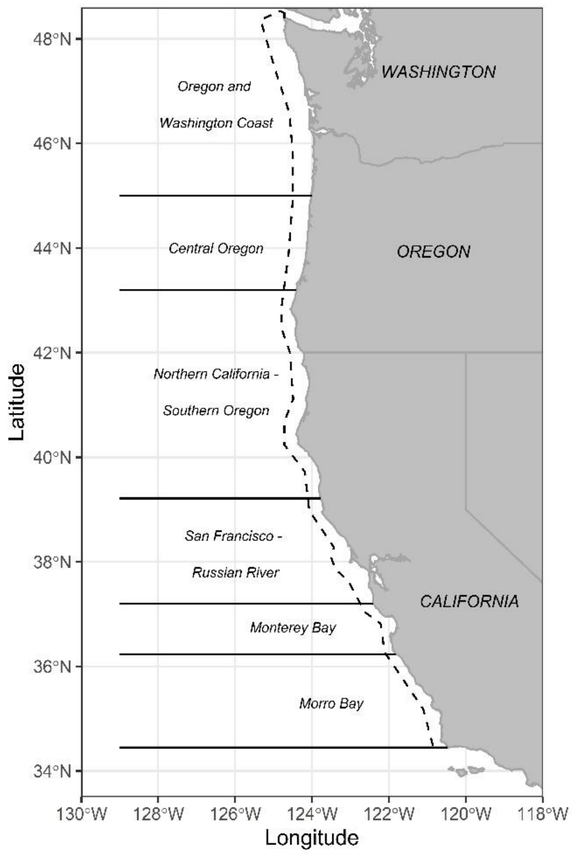

Figure 1. Stock boundaries and distributional range of harbor porpoise along the outer U.S. West Coast. Dashed line represents an approximate boundary for harbor porpoise habitat (0-200m water depth).

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

For the Marine Mammal Protection Act (MMPA) Stock Assessment Reports, Pacific coast harbor porpoise stocks include: 1) a Morro Bay stock, 2) a Monterey Bay stock, 3) a San Francisco-Russian River stock, 4) a northern California/southern Oregon stock, 5) a central Oregon stock, 6) a northern Oregon/Washington coast stock, 7) an Inland Washington stock, 8) a northern Southeast Alaska Inland Waters stock, 9) a southern Southeast Alaska Inland Waters stock, 10) a Yakutat/Southeast Alaska Offshore Waters stock, 11) a Gulf of Alaska stock, and 12) a Bering Sea stock. The stock assessment reports for harbor porpoise stocks within waters of California, Oregon, and Washington appear in this volume. The five Alaska harbor porpoise stocks are reported separately in the Stock Assessment Reports for the Alaska Region.

~~In the eastern North Pacific Ocean, harbor porpoise are found in coastal and inland waters from Point Barrow, along the Alaskan coast, and down the west coast of North America to Point Conception,~~

~~California (Gaskin 1984). Harbor porpoise are known to occur year-round in the inland trans-boundary waters of Washington and British Columbia, Canada (Osborne *et al*. 1988) and along the Oregon/Washington coast (Barlow 1988, Barlow *et al*. 1988, Green *et al*. 1992). Aerial survey data from coastal Oregon and Washington, collected during all seasons, suggest that harbor porpoise distribution varies by depth (Green *et al*. 1992). Although distinct seasonal changes in abundance along the west coast have been noted, and attributed to possible shifts in distribution to deeper offshore waters during late winter (Dohl *et al*. 1983, Barlow 1988), seasonal movement patterns are not fully understood. Investigation of pollutant loads in harbor porpoise ranging from California to the Canadian border suggests restricted harbor porpoise movements (Calambokidis and Barlow 1991). Stock discreteness in the eastern North Pacific was analyzed using mitochondrial DNA from samples collected along the west coast (Rosel 1992) and is summarized in Osmek *et al*. (1994). Two distinct mtDNA groupings or clades exist. One clade is present in California, Washington, British Columbia, and Alaska (no samples were available from Oregon), while the other is found only in California and Washington. Although these two clades are not geographically distinct by latitude, the results may indicate a low mixing rate for harbor porpoise along the west coast of North America. Further genetic testing of the same data, along with additional samples, found significant genetic differences for four of the six pair-wise comparisons between the four areas investigated: California, Washington, British Columbia, and Alaska (Rosel *et al*. 1995). These results demonstrate that harbor porpoise along the west coast of North America are not panmictic or migratory and that movement is sufficiently restricted that genetic differences have evolved. Recent preliminary genetic analyses of samples ranging from Monterey Bay, California, to Vancouver Island, British Columbia, indicate that there is small-scale subdivision within the U.S. portion of this range (Chivers *et al*. 2002, 2007). This is consistent with low movement suggested by genetic analysis of harbor porpoise specimens from the North Atlantic, where numerous stocks have been delineated with clinal differences over areas as small as the waters surrounding the British Isles.~~

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

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

POPULATION SIZE

~~Two separate aerial surveys for leatherback turtles were conducted during 2010 and 2011 from the coast approximately to the 2,000 m isobath between Cape Blanco, Oregon, and Cape Flattery, Washington. Some additional adaptive surveys were conducted in areas of special interest for leatherback turtles; although~~

~~these transects were not included in the analysis, the corresponding harbor porpoise sightings were included for estimation of the detection function in this study. Using a correction factor of 3.42 (1/g(0); g(0)=0.292, CV=0.366) (Laake *et al*. 1997a), to adjust for groups missed by aerial observers, the corrected estimate of abundance for harbor porpoise in the coastal waters of northern Oregon (north of Lincoln City) and Washington in 2010-2011 is 21,487 (CV = 0.44) (Forney et al. 2013).~~

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

Minimum Population Estimate

The minimum population estimate for this stock is calculated as the lower 20th percentile of the log-normal distribution ~~(Wade and Angliss 1997)~~ of the ~~2010-2011~~ 2021-2022 ~~population~~ abundance estimate, or 16,068 animals ~~of 21,487, which is 15,123 harbor porpoise~~.

Current Population Trend

There are no reliable data on population trends of harbor porpoise for coastal Oregon and ~~,~~ Washington~~, or British Columbia waters~~; however, the 2010-2011 (Forney *et al.* 2014) and 2021-22 (Forney et al. 2023) abundance estimates are very similar. ~~uncorrected estimates of abundance for the Northern Oregon/Washington Coast stock in 1997 (6,406; SE=826.5) and 2002 (4,583) were not significantly different (Z=-1.73, P=0.08), although the survey area in 1997 (Regions I-S through III) was slightly larger than in 2002 (Strata D-G) (Laake *et al*. 1998a; J. Laake, unpublished data). The 2010-2011 Northern Oregon/Washington Coast stock estimate (21,487, CV = 0.44) is greater than the previous 2002 estimate of 15,674 (CV = 0.39), but the previous estimate is within the confidence limit of the current abundance estimate (Forney et al. 2013).~~

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Based on what are argued to be biological limits of the species (i.e. females give birth first at age 4 and produce one calf per year until death), the theoretical, maximum-conceivable growth rate of a closed harbor porpoise population was estimated as 9.4% per year based on a human survivorship curve (Barlow and Boveng 1991). ~~This maximum theoretical rate represents maximum survival in a protected environment and may not be achievable for any wild population (Barlow and Boveng 1991). Woodley and Read (1991) calculate a maximum growth rate of approximately 5% per year, but their argument for this being a maximum (i.e. that porpoise survival rates cannot exceed those of Himalayan thar) is not well justified. Population growth rates have not actually been measured for any harbor porpoise population.~~ This is very similar to the growth rate of 9.6% per year (95% credible interval: 6.2% - 13.0%) estimated by Forney *et al.* (2020) for the Morro Bay harbor porpoise stock between 1991 and 2012, based on long-term aerial surveys. Because a reliable estimate of the maximum net productivity rate is not available for the Northern Oregon / Washington Coast harbor porpoise stock, we use the default maximum net productivity rate (R_{MAX}) of 4% for cetaceans (Wade and Angliss 1997).

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (~~15,123~~16,068) times one-half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.5 (for a stock of unknown status, Wade and Angliss 1997), resulting in a PBR of ~~151~~161 ~~harbor porpoise per year~~.

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Fisheries Information

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

~~Within the EEZ boundaries of the coastal waters of northern Oregon and Washington, harbor porpoise deaths are known to occur in the northern Washington marine set gillnet tribal fishery. Total fishing effort in this fishery is conducted within the range of both harbor porpoise stocks (Northern Oregon/Washington Coast and Washington Inland Waters) occurring in Washington State waters (Gearin *et al*. 1994). Some movement of harbor porpoise between Washington's coastal and inland waters is likely, but it is currently not possible to quantify the extent of such movements. For the purposes of this stock assessment report, the animals taken in waters south and west of Cape Flattery, WA, are assumed to have belonged to the Northern Oregon/Washington Coast stock, and Table 1 includes data only from that portion of the fishery. Fishing effort in the coastal marine set gillnet tribal fishery has declined since 2004. A test set gillnet fishery, with 100% observer coverage, was conducted in coastal waters in 2008 and 2011. This test fishery required the use of nets equipped with acoustic alarms, and no harbor porpoise deaths were reported (Makah Fisheries Management, unpublished data). The mean estimated mortality for this fishery in 2007-2011 is 0 (CV=0) harbor porpoise per year from observer data.~~

Table 1. Summary of incidental mortality and serious injury of harbor porpoise (Northern Oregon/Washington Coast stock) in commercial and tribal fisheries that might take this species and calculation of the mean annual mortality rate; n/a indicates that data are not available. Mean annual takes are based on ~~2007-2011~~ 2017-2021 data unless noted otherwise.

Fishery name	Years	Data type	Percent observer coverage	Observed mortality	Estimated mortality	Mean annual takes (CV in parentheses)
~~Northern WA marine set gillnet (tribal test fishery in coastal waters)[1]~~	~~2007~~ ~~2008~~ ~~2009~~ ~~2010~~ ~~2011~~	~~observer~~	~~no fishery~~ ~~100%~~ ~~no fishery~~ ~~100%~~ ~~no fishery~~	~~0~~ ~~0~~ ~~0~~ ~~0~~ ~~0~~	~~0 (0)~~ ~~0 (0)~~ ~~0 (0)~~ ~~0 (0)~~ ~~0 (0)~~	~~0 (0)~~
Gillnet fishery (tribal)[1]	2017-2021	Fishery self-report	n/a	0, 0, 2, 0, 0		>0.4 (n/a)
Unidentified gillnet fishery	2017-2021	stranding	n/a	0, 3, 1, 0, 1		>1.0 (n/a)
~~Unknown West Coast fisheries~~ Unidentified fishery	2007-2011	stranding	~~n/a~~	~~2, 1, 3, 3, 6~~ 8, 0, 0, 0, 1	n/a	>~~3.0~~1.8 (n/a)
Minimum total annual takes						>~~3.0~~3.2 (n/a)

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

~~In 1995-1997, data were collected for the coastal portions (areas 4 and 4A) of the northern Washington marine set gillnet fishery as part of an experiment, conducted in cooperation with the Makah Tribe, designed to explore the merits of using acoustic alarms to reduce bycatch of harbor porpoise in salmon gillnets. Results in 1995-1996 indicated that the nets equipped with acoustic alarms had significantly lower entanglement rates, as only 2 of the 49 deaths occurred in alarmed nets (Gearin *et al*. 1996, 2000; Laake *et al*. 1997b). In 1997, 96% of the sets were equipped with acoustic alarms and 13 deaths were observed (Gearin *et al*. 2000; P. Gearin, unpublished data). Harbor porpoise were displaced by an acoustic buffer around the alarmed nets, but it is unclear whether the porpoise or their prey were repelled by the alarms (Kraus *et al*. 1997, Laake *et al*. 1998b). However, the acoustic alarms did not appear to affect the target catch (chinook salmon and sturgeon) in the fishery (Gearin *et al*. 2000). For the past decade, Makah tribal regulations have required nets set in coastal waters (areas 4 and 4A) to be equipped with acoustic alarms.~~

~~According to Northwest Marine Mammal Stranding Network records, maintained by the NMFS Northwest Region (NMFS, Northwest Regional Office, unpublished data), there were 15 fishery-related strandings of harbor porpoise from this stock reported on the northern Oregon/Washington coast in 2007-2011 (2 in 2007, 1 in 2008, 3 in 2009, 3 in 2010, and 6 in 2011), resulting in a mean annual mortality of~~

~~3.0 harbor porpoise in 2007-2011. Evidence of fishery interactions included net marks, rope marks, and knife cuts (Carretta et al. 2013). Since these deaths could not be attributed to a particular fishery, and were the only confirmed fishery-related deaths in this area in 2007-2011, they are listed in Table 1 as occurring in unknown West Coast fisheries. Seven additional strandings reported in 2007-2011 (2 in 2007, 1 in 2008, 1 in 2009 , and 3 in 2011) were considered possible fishery-related strandings but were not included in the estimate of mean annual mortality. This estimate is considered a minimum because not all stranded animals are found, reported, or examined for cause of death (via necropsy by trained personnel).~~

~~Other Mortality~~

~~A significant increase in the number of harbor porpoise strandings reported throughout Oregon and Washington in 2006 prompted the Working Group on Marine Mammal Unusual Mortality Events to declare an Unusual Mortality Event (UME) on 3 November 2006 (Huggins 2008). A total of 114 harbor porpoise strandings were reported and confirmed throughout Oregon/Washington coast and Washington inland waters in 2006 and 2007 (Huggins 2008). The cause of the UME has not been determined, and several factors, including contaminants, genetics, and environmental conditions, are still being investigated. Cause of death, determined for 48 of 81 porpoise that were examined in detail, was attributed mainly to trauma and infectious disease. Suspected or confirmed fishery interactions were the primary cause of adult/subadult traumatic injuries, while birth-related trauma was responsible for the neonate deaths. Although six of the Northern Oregon/Washington Coast harbor porpoise deaths examined as part of the UME were suspected to have been caused by fishery interactions, only two could be confirmed as fishery-related deaths; these two deaths are listed in Table 1 as occurring in unknown West Coast fisheries in 2007.~~

STATUS OF STOCK

Harbor porpoise along the outer coast of northern Oregon and Washington are not listed as threatened or endangered under the Endangered Species Act nor as depleted under the Marine Mammal Protection Act. The status of this stock relative to its Optimum Sustainable Population (OSP) level and population trends is unknown. ~~are not listed as "depleted" under the MMPA or listed as "threatened" or "endangered" under the Endangered Species Act.~~ Based on currently available data, the minimum annual level of total human-caused mortality and serious injury (~~3.0~~ 3.2 per year) does not exceed the PBR (~~151~~161). Therefore, the Northern Oregon/Washington Coast stock of harbor porpoise is not classified as a "strategic" ~~."~~ stock under the MMPA. The minimum annual fishery mortality and serious injury for this stock (~~3.0~~3.2) is not known to exceed 10% of the calculated PBR (~~15.1~~16.1) and, therefore, can be considered to be insignificant and approaching zero mortality and serious injury rate. ~~The status of this stock relative to its Optimum Sustainable Population (OSP) level and population trends is unknown.~~

OTHER FACTORS THAT MAY AFFECTING THE STOCK

~~Habitat Issues~~

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

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

Killer whales occur in all oceans and seas (Leatherwood and Dahlheim 1978). Although they occur in tropical and offshore waters, killer whales prefer the colder waters of both hemispheres, with greatest abundances found within 800 km of major continents (Forney and Wade 2006). Along the west coast of North America, killer whales occur along the entire Alaskan 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. Seasonal and year-round occurrence is documented for killer whales throughout Alaska (Braham and Dahlheim 1982) and in the intracoastal waterways of British Columbia and Washington, where three ecotypes have been recognized: 'resident', 'transient' and 'offshore' (Bigg *et al.* 1990, Ford *et al.* 1994), based on aspects of morphology, ecology, genetics and behavior (Ford and Fisher 1982; Baird and Stacey 1988; Baird *et al.* 1992, Hoelzel *et al.* 1998, Morin *et al.* 2010, Ford *et al.* 2014). Genetic studies of killer whales globally suggest that residents and transient ecotypes warrant subspecies recognition (Morin *et al.* 2010) and each are currently listed as unnamed subspecies of *Orcinus orca* (Committee on Taxonomy 2018).

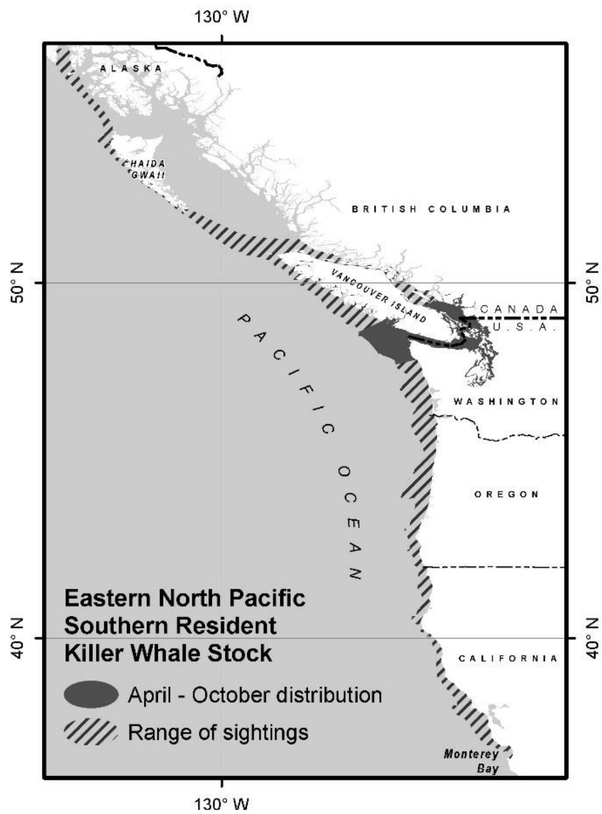

Figure 1. Approximate April - October distribution of the Eastern North Pacific Southern Resident killer whale stock (shaded area) and range of sightings (diagonal lines).

The range of southern resident killer whales is described in the biological report for the Revision of the Critical Habitat Designation for Southern Resident Killer Whales (NMFS 2021a, 2021b): "The three pods of the Southern Resident DPS, identified as J, K, and L pods, reside for part of the year in the inland waterways of Washington State and British Columbia known as the Salish Sea (Strait of Georgia, Strait of Juan de Fuca, and Puget Sound), principally during the late spring, summer, and fall (Ford et al. 2000, Krahn et al. ~~2002~~ 2004). The whales also occur in outer coastal waters, primarily in winter, off Washington and Vancouver Island, especially in the area between Grays Harbor and the Columbia River, and off Westport, WA (Ford et al. 2000, Hanson et al. 2017), but have been documented as far south as central California and as far north as the Southeast Alaska. Although less is known about the whales' movements in outer coastal waters, satellite tagging, opportunistic sighting, and acoustic recording data suggest that Southern Residents spend nearly all of their time on the continental shelf, within 34 km (21.1 mi) of shore in water less than 200 m (656.2 ft) deep (Hanson *et al*. 2017)." Details of their winter range from satellite-tagging reveal whales use the entire Salish Sea (northern end of the Strait of Georgia and Puget Sound) in addition to coastal waters from the central west coast of Vancouver Island, British Columbia to Pt. Reyes in northern California. Animals from J pod were documented moving between the northern Strait of Georgia and the western entrance of the Strait of Juan de Fuca, with limited movement into coastal waters. In contrast, K and L pod movements were characterized by a coastal distribution from the western entrance to the Strait of Juan de Fuca to Pt. Reyes California (Hanson *et al*. 2017). Of the three pods comprising this stock, one (J) is commonly sighted in inshore waters in winter, while the other two (K and L) apparently spend more time offshore (Ford *et al.* 2000). Krahn *et al.* (2009) described sample pollutant ratios from K and L pod whales that were consistent with a hypothesis of time spent foraging in California waters, which is consistent with sightings of K and L pods as far south as Monterey Bay. In June 2007, whales from L-pod were sighted off Chatham Strait, Alaska, the farthest north they have ever been documented (J. Ford, pers. comm.). Southern resident killer whale attendance in their core summer habitat in the Salish Sea appears to be declining, with occurrence well-below average since 2017 (Center for Whale Research 2019). Passive autonomous acoustic recorders have provided more information on the seasonal occurrence of these pods along the west coast of the U.S. (Hanson *et al.* 2013). In addition, satellite-linked tags were deployed in winter months on members of J, K, and L pods. Results were consistent with previous data, but provided much greater detail, showing wide-ranging use of inland waters by J Pod whales and extensive movements in U.S. coastal waters by K and L Pods.

Based on data regarding association patterns, acoustics, movements, genetic differences and potential fishery interactions, eight killer whale stocks are recognized within the Pacific U.S. EEZ: 1) the Eastern North Pacific Alaska Resident stock - occurring from Southeast Alaska to the Bering Sea, 2) the Eastern North Pacific Northern Resident stock - occurring from British Columbia through Alaska, 3) the Eastern North Pacific Southern Resident stock - occurring mainly within the inland waters of Washington State and southern British Columbia but extending from central California into southern Southeast Alaska (see Fig. 1), 4) the West Coast Transient stock - occurring from Alaska through California, 5) the Gulf of Alaska, Aleutian Islands, and Bering Sea Transient stock - occurring from southeast Alaska to the Bering Sea, 6) the AT1 Stock – found only in Prince William Sound, 7) the Eastern North Pacific Offshore stock - occurring from Southeast Alaska through California, 8) the Hawaiian stock. The Stock Assessment Reports for the Alaska Region contain information concerning the Eastern North Pacific Alaska Resident, Eastern North Pacific Northern Resident and the Gulf of Alaska, Aleutian Islands, and Bering Sea, AT1, and Eastern North Pacific Transient stocks.

POPULATION SIZE

The Eastern North Pacific Southern Resident stock is a trans-boundary stock including killer whales in inland Washington and southern British Columbia waters. In 1993, the three pods comprising this stock totaled 96 killer whales (Ford *et al.* 1994). The population increased to 99 whales in 1995, then declined to 79 whales in 2001, and most recently numbered ~~74~~ 73 whales in ~~2021~~ 2022 (Fig. 2; Ford *et al.* 2000; Center for Whale Research ~~2021~~ 2022). The most recent census spanning 1 July ~~2020~~ 2021 through 1 July ~~2021~~ 2022 includes ~~three~~ two new calves (~~J57, J58, L125~~ J59, K45), and the death of ~~a post-reproductive female, but does not include the death of an~~ three adult males (K21, K44, and L89). No other births or mortalities have been documented since completion of the census ~~in late summer of 2021, or two calves born in early 2022~~.

Minimum Population Estimate

The abundance estimate for this stock of killer whales is a direct count of individually identifiable animals. It is thought that the entire population is censused every year. This estimate therefore serves as both a best estimate of abundance and a minimum estimate of abundance. Thus, the minimum population estimate (N_{min}) for the Eastern North Pacific Southern Resident stock of killer whales is ~~74~~ 73 animals.

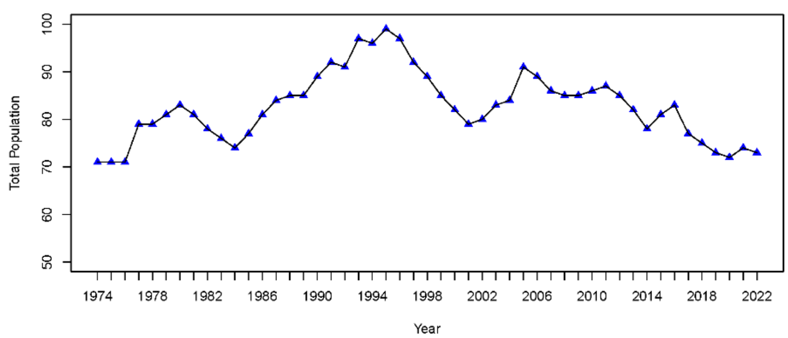

Figure 2. Population of Eastern North Pacific Southern Resident stock of killer whales, 1974-~~2021~~ 2022. Each year's count includes animals first seen and first missed; a whale is considered first missed the year after it was last seen alive (Ford *et al.* 2000; Center for Whale Research ~~2021~~ 2022).

Current Population Trend

During the live-capture fishery that existed from 1967 to 1973, it is estimated that 47 killer whales, mostly immature, were taken out of this stock (Ford *et al.* 1994). Since the first complete census of this stock in 1974 when 71 animals were identified, the number of southern resident killer whales has fluctuated. Between 1974 and the mid-1990s, the Southern Resident stock increased approximately 35% (Ford *et al.* 1994), representing a net annual growth rate of 1.8% during those years. Following the peak census count of 99 animals in 1995, the population size has declined approximately 1% annually and currently stands at ~~74~~ 73 animals as of the ~~2021~~ 2022 census (Ford *et al.* 2000; Center for Whale Research ~~2021~~ 2022). Recent population models, based on an analysis of the entire SRKW genome, suggest that inbreeding depression is limiting population growth, and predicts further decline if the population remains genetically isolated and typical environmental conditions continue (Kardos *et al.* in press).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate is currently unavailable for this stock of killer whales. Matkin *et al.* (2014) estimated a maximum population annual growth rate of 1.035 for southern Alaska resident killer whales. The authors noted that the 3.5% annual rate estimated for southern Alaska residents is higher than previously measured rates for British Columbia northern residents (2.9%, Olesiuk *et al.* 1990) and "probably represents a population at r-max (maximum rate of growth)." In the absence of published estimates of R_{max} for southern resident killer whales, the maximum annual rate of 3.5% found for southern Alaska residents is used for this stock of southern resident killer whales. This reflects more information about the known life history of resident killer whales than the default R_{max} of 4% and results in a more conservative estimate of potential biological removal (PBR).

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (~~74~~ 73) times one-half the maximum net growth rate for *Alaska* resident killer whales (½ of 3.5%) times a recovery factor of 0.1 (for an endangered stock, Wade and Angliss 1997), resulting in a PBR of 0.13 whales per year, or approximately 1 animal every 7 years.

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Fisheries Information

The only known case of southern resident killer whale mortality due to fisheries is an adult male, L8, who entangled in gillnet fishing gear and drowned in 1977 (Center for Whale Research 2015). The entanglement occurred near southeastern Vancouver Island (Ford *et al*. 1998), and upon necropsy two pounds of recreational fishing lures and lines were found in the stomach. It was noted that some of the fishing gear found did not appear to be used locally at the time and the ingestion of the gear did not cause the death of the animal. Salmon drift gillnet fisheries in Washington inland waters were last observed in 1993 and 1994 and no killer whale entanglements were documented, though observer coverage levels were less than 10% (Erstad *et al.* 1996, Pierce *et al.* 1994, Pierce *et al.* 1996, NWIFC 1995). Fishing effort in the inland waters drift gillnet fishery has declined considerably since 1994 because far fewer vessels participate today. Past marine mammal entanglements in this fishery included harbor porpoise, Dall's porpoise, and harbor seals. Coastal marine tribal set gillnets also occur along the outer Washington coast and no killer whale interactions have been reported in this fishery since the inception of the observer program in 1988, though the fishery is not active every year (Gearin *et al.* 1994, Gearin *et al.* 2000, Makah Fisheries Management). No fishery-related mortality from gillnet fisheries in California waters was documented between ~~2015-2020~~ 2017-2021 (Carretta ~~2021~~2022, ~~Carretta *et al.* 2021,~~ Carretta *et al*. ~~2022~~ 2023).

An additional source of information on killer whale mortality and injury incidental to commercial fishery operations is the self-reported fisheries information required of vessel operators by the MMPA. No self-report records of killer whale mortality have been reported.

In 2015, J39, a young male southern resident killer whale, was found near False Bay, WA, with a recreational salmon flasher dangling from its mouth (Center for Whale Research, 2015). The whale was seen five days later without the gear attached and appeared energetic. The whale was monitored over the following weeks and there was no evidence of injury or behavioral changes (Center for Whale Research, 2015).

Due to a lack of observer programs, there are few data concerning the mortality of marine mammals incidental to Canadian commercial fisheries. Since 1990, there have been no reported fishery-related strandings of killer whales in Canadian waters. However, in 1994 one killer whale was reported to have contacted a salmon gillnet but did not entangle (Guenther *et al.* 1995). In 2014 a northern resident killer whale became entangled in a gillnet, was released from the net, but died the next winter (Fisheries and Oceans Canada 2018). Data regarding the level of killer whale mortality related to commercial fisheries in Canadian waters are not available.

The known total fishery mortality and serious injury for the southern resident stock of killer whales is zero, but undetected mortality and serious injury may occur.

Other Mortality

In 2012, a moderately decomposed juvenile female southern resident killer whale (L-112) was found dead near Long Beach, WA. A full necropsy was performed and the cause of death was determined to be blunt force trauma to the head, however the source of the trauma (vessel strike, intraspecific aggression, or other unknown source) could not be established (NOAA 2014). There was documentation of a whale-boat collision in Haro Strait in 2005 which resulted in a minor injury to a whale. In 2006, whale L98 was killed during a vessel interaction. It is important to note that L98 had become habituated to regularly interacting with vessels during its isolation in Nootka Sound. In spring 2016, a young adult male, L95, was found to have died of a fungal infection related to a satellite tag deployment approximately 5 weeks prior to its death. The expert panel reviewing the stranding noted that "the tag loss, tag petal retention with biofilm formation or direct pathogen implantation, and development of a fungal infection at the tag site contributed to the illness, stranding, and death of this whale." (NMFS 2016). In fall 2016 another young adult male, J34, was found dead in the northern Georgia Strait. The necropsy indicated that "the animal had injuries consistent with blunt trauma to the dorsal side, and a hematoma indicating that it was alive at the time of injury and would have survived the initial trauma for a period of time prior to death" (Fisheries and Oceans Canada 2019). The injuries are consistent with those incurred during a vessel strike. A recent summary of killer whale strandings in the northeastern Pacific Ocean and Hawaii noted the occurrence of human interactions across all age classes (Raverty *et al*. 2020).

STATUS OF STOCK

Total documented annual fishery mortality and serious injury for this stock from ~~2015-2020~~ 2017-2021 (zero) is not known to exceed 10% of the calculated PBR (0.13). Given the low PBR level, a single undetected / undocumented fishery mortality or serious injury would exceed 10% of the PBR, thus it is unknown if fishery mortality and serious injury is approaching zero mortality and serious injury rate. The documented annual level of human-caused mortality and serious injury for the most-recent 5-year period of 2017-2021 is zero. ~~includes the death of L95 (fungal infection related to a satellite tag) and J34 (vessel strike), or 0.4 whales annually, which exceeds the PBR (0.13).~~ Southern Resident killer whales were formally listed as "endangered" under the ESA in 2005 and consequently the stock is automatically considered as a "strategic" stock under the MMPA. This stock was considered "depleted" (68 FR 31980, May 29, 2003) prior to its 2005 listing under the ESA (70 FR 69903, November 18, 2005).

OTHER FACTORS THAT MAY BE CAUSING A DECLINE OR IMPEDING RECOVERY

~~**Habitat Issues**~~

A population viability analysis identified several risk factors to this population, including limitation of preferred Chinook salmon prey, anthropogenic noise and disturbance resulting in decreased foraging efficiency, and high levels of contaminants, including PCBs and DDT (Ebre 2002, Clark *et al.* 2009, Krahn *et al.* 2007, 2009, Lacy *et al.* 2017). The summer range of this population, the inland waters of Washington and British Columbia, are home to a large commercial whale watch industry, and high levels of recreational boating and commercial shipping. Potential for acoustic masking effects on the whales' communication and foraging due to vessel traffic remains a concern (Erbe 2002, Clark *et al.* 2009, Lacy *et al.* 2017, Holt *et al*. 2021a, b). In 2011, vessel approach regulations were implemented to restrict vessels from approaching closer than 200m. A genetic study of diet of southern resident killer whales from fecal remains collected during 2006-2011 noted that salmonids accounted for >98.6% of genetic sequences (Ford *et* al. 2016). Of six salmonid species documented, Chinook salmon accounted for 79.5% of the sequences, followed by coho salmon (15%). Chinook salmon dominate the diet in early summer, with coho salmon averaging >40% of the diet in late summer. Sockeye salmon were also found to be occasionally important (>18% in some samples). Non-salmonids were rarely observed. These results are consistent with those obtained from surface prey remains, and confirm the importance of Chinook salmon in this population's diet. These authors also noted the absence of pink salmon in the fecal samples. Prior studies note the prevalence of Chinook salmon in the killer whale diet, despite the relatively low abundance of this species in the region, supporting the thesis that southern resident killer whales are Chinook salmon specialists (Ford and Ellis 2006, Hanson *et al.* 2010). Recent studies of diet in other seasons and regions of their range indicate that although Chinook represent a major component of their diet almost year-round, other species also make potentially important contributions, likely when Chinook are less available (Hanson et al. 2021). There is evidence that reduced abundance of Chinook salmon has negatively affected this population via reduced fecundity (Ayres *et al*. 2012, Ford *et al.* 2009, Ward *et al.* 2009, Wasser *et al.* 2017). Studies on body condition and sizes of southern resident killer whales using aerial photogrammetry (Fearnbach *et al*. 2011, Fearnbach *et al*. 2018, Stewart *et al*. 2021) reflect hypotheses between Chinook salmon abundance and killer whale body condition and overall body size. In some cases (J-Pod), Chinook abundance was found to have the greatest predictive power on southern resident body condition, while this relationship was absent for K-Pod (Stewart *et al*. 2021). In other studies (Fearnbach *et al*. 2011), authors suggest that nutritional stress is linked to a longer term decrease in body size in the population. In addition, the high trophic level and longevity of the population has predisposed them to accumulate high levels of contaminants that potentially impact health (Krahn *et al.* 2007, 2009). In particular, there is evidence of high levels of flame retardants in young animals (Krahn *et al.* 2007, 2009). High DDT/PCB ratios have been found in Southern Resident killer whales, especially in K and L pods (Krahn *et al.* 2007, NMFS 2019b), which spend more time in California waters where DDTs still persist in the marine ecosystem (Sericano *et al*. 2014).

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

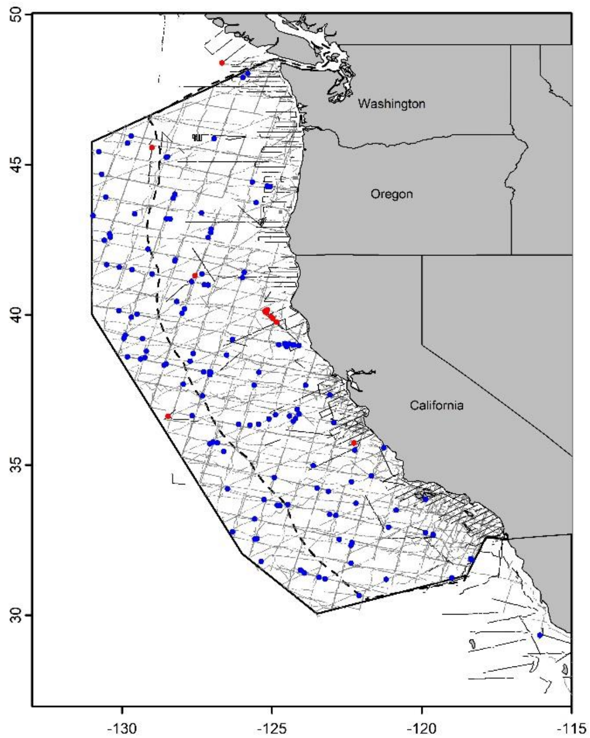

Figure 1. Sperm whale sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018. Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

POPULATION SIZE

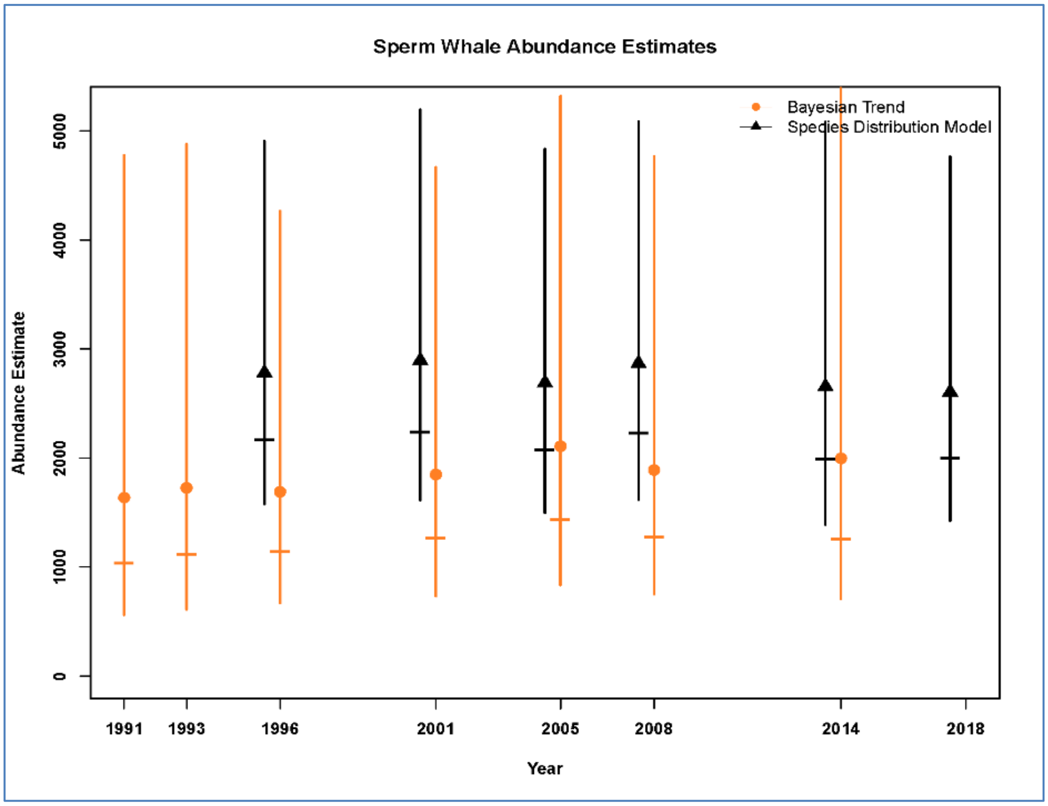

Figure 2. Trend-based and habitat-based abundance estimates of sperm whale abundance in the California Current, 1991-~~2014~~ 2018 (Moore and Barlow 2017, Becker *et al.* 2020). Abundance estimates (posterior medians [●] and 95% CRIs) from the trend model and mean estimated abundance [▲] and 95% confidence limits from the habitat model are shown. Horizontal hatch marks represent lower 80% percentiles of each estimate, corresponding to the minimum population estimate.

~~Previous estimates of sperm whale abundance from 2005 (3,140, CV=0.40, Forney 2007) and 2008 (300, CV=0.51, Barlow 2010) show a 10-fold difference that cannot be attributed to human-caused or natural population declines and likely reflect sampling variance and inter-annual variability in movement of animals into and out of the study area. New estimates of sperm whale abundance in California, Oregon, and Washington waters out to 300 nmi are available from a trend-model analysis of line-transect data collected from seven surveys conducted from 1991 to 2014 (Moore and Barlow 2017). Abundance trend models incorporate information from the entire 1991 to 2014 time series to obtain each annual abundance estimate, yielding estimates with less inter-annual variability. The trend model also uses improved estimates of group size and trackline detection probability (Moore and Barlow 2014, Barlow 2015). Sperm whale abundance estimates based on the trend-model range between 2,000 and 3,000 animals for the 1991 to 2014 time series (Moore and Barlow 2014). The best estimate of sperm whale abundance in the California Current is the trend-based estimate corresponding to the most recent 2014 survey, or 1,997 (CV= 0.57) whales. This estimate is corrected for diving animals not seen during surveys.~~ A series of abundance estimates are available from Bayesian trend models derived from line-transect surveys between 1991-2014 (Moore and Barlow 2017) and habitat-based density models from 1991-2018 (Becker *et al.* 2020) (Figure 2). Estimates from the two methods largely-overlap, though estimates from habitat models are, on average, higher. The most-recent estimate of sperm whale abundance for this stock is based on a 2018 survey and a habitat density model that is informed by 1991-2018 data, or 2,606 (CV = 0.135) whales (Becker *et al.* 2020).

Minimum Population Estimate

The minimum population estimate for sperm whales is taken as the lower 20th percentile of the ~~posterior distribution of the 2014~~ 2018 abundance estimate, or ~~1,270~~ 2,011 whales (~~Moore and Barlow 2017~~ Becker *et al*. 2020).

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (~~1,270~~ 2,011) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.1 (for an endangered stock with N_{min} ~~<1,500~~ >1,500 and stable trend in abundance; Taylor *et al.* 2003), resulting in a PBR of ~~2.5~~ 4 animals per year.

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Fishery Information

The fishery most likely to injure or kill sperm whales from this stock is the California thresher shark/swordfish drift gillnet fishery (Julian and Beeson 1998, Carretta ~~*et al.* 2019a, 2019b~~ 2022). Observed serious injury and mortality is rarely observed in the fishery (10 animals from 6 events observed during ~~8,956~~>9,246 fishing sets between 1990 and ~~2017~~ 2021, Carretta ~~*et al.* 2019b~~ 2022). While there has not been an observed entanglement of sperm whales in this fishery since 2010, there is a positive estimate of sperm whale bycatch in the fishery for the most-recent 5-year period of 2017-2021, based on a data model that uses 1990-2021 data (Carretta 2022). This estimate is 1.58 (CV=2.8) whales, or 0.32 whales annually (Carretta 2022). ~~Previous ratio estimates of drift gillnet bycatch for this stock suffered from inter-annual volatility and estimation bias because estimates were based on intra-annual data where observed entanglements were rare and observer coverage was low (Julian and Beeson 1998, Carretta *et al.* 2004, Carretta and Moore, 2014,). The prescribed strategy of pooling 5 years of annual bycatch estimates in stock assessments (Wade and Angliss 1997) is insufficient to overcome these biases when events are rare and when estimates are based on within-year data (Carretta and Moore 2014). However, model-based bycatch estimates that incorporate all available data for annual estimates allow for the robust pooling of data over 5-year time periods. New model-based estimates of sperm whale bycatch based on random forest regression trees were generated for the 28-year period 1990-2017, where annual estimates incorporate data from all years (Carretta *et al.* 2019b). Additionally, estimates were derived for the most recent 5-year period of 2013 to 2017, and because the last observation of sperm whale entanglement occurred >5 years ago, Table 1 also includes bycatch estimates for the most recent 10-year period (2008-2017) for additional context. Estimated entanglements for the period 2013-2017 in the California drift gillnet fishery are 2.9 (CV=1.3) sperm whales, however, not all of these represent deaths or serious injuries (Carretta *et al.* 2019b). Based on a review of sperm whale entanglements in the fishery, 7 of the 10 entanglements resulted in serious injury (n=2) or death (n=5), with the remaining 3 cases resulting in non-serious injuries because animals were released from nets uninjured and were expected to live. The estimated number of sperm whales seriously injured or killed from 2013-2017 is therefore 2.0 (CV=1.4) whales (Carretta *et al.* 2019b), or 0.4 whales annually (Table 1). The 5-year annual mean (0.4 whales, CV=1.4) is similar to the 10-year annual mean of 0.56 whales (Table 1). Two notable differences~~

~~between intra-annual ratio estimates and model-based estimates of bycatch are: 1) annual model-based estimates can be positive, even when no entanglements are observed and 2) estimates can take on fractional values (<1 whale) (Carretta *et al.* 2019b, Table 1). As some estimates of serious injury and mortality are < 0.5 of a whale, resulting coefficients of variation (CVs) can be quite large due to the extremely small mean estimates. Of particular note is that the regression tree bycatch estimate for 2010 is 2.0 sperm whales entangled (Carretta *et al.* 2019b). The ratio estimate of bycatch for the same year is 16.7 whales and is considered positively-biased (Carretta *et al.* 2019b). The estimate of serious injury and death in the fishery in 2017 is also 2.0 whales, even though there were no entanglements observed that year. The 2017 estimate (equal to the 2010 estimate) is among the highest in the previous 10 years because observed fishing depths and locations in 2017 are similar to fishing conditions associated with observed sperm whale entanglements and predictions of unobserved bycatch are based on these fishing set characteristics (Carretta *et al.* 2019b).~~

~~Estimates of sperm whale bycatch in the limited-entry sablefish hook and line fishery are also available for 2012 to 2016, based on a single observed interaction in 2007 (Jannot *et al.* 2018). Estimates are based on a Bayesian model for years without observed bycatch and are approximately 0.25 whales annually (Jannot *et al.* 2018, Table 1).~~

Strandings of sperm whales are rare and it is expected that documented anthropogenic deaths and injuries due to entanglements within unknown fisheries or ingestion of marine debris represent a small fraction of the true number of cases, due to the low probability that the carcass of a highly-pelagic species washes ashore (Williams *et al.* 2011, Carretta *et al.* 2016a). ~~Published summaries of human-caused mortality and serious injury of sperm whales from unidentified fisheries and marine debris on the U.S. west coast include records inclusive from 2007 to 2015 (Jacobsen *et al.* 2010, Carretta *et al.* 2013, 2014, 2015, 2016b, 2017a, 2019a). Three separate sperm whale strandings in 2008 (all dead animals) showed evidence of fishery interactions (Jacobsen *et al.* 2010). Two whales died from gastric impaction from ingesting multiple types of floating polyethylene netting (Jacobsen *et al.* 2010). The variability in size and age of the ingested net material suggests that it was ingested as surface debris and was not the result of fishery depredation (Jacobsen *et al.* 2010). Net types recovered from the whales' stomachs included portions of gillnet, bait nets, and fish/shrimp trawl nets. A third whale in 2008 showed evidence of entanglement scars (Carretta *et al.* 2013).~~ Prior cases of observed mortality and serious injury of sperm whales due to interactions with unidentified fisheries and marine debris have been reported by Jacobsen *et al.* (2010) and Carretta *et al.* (2013, 2014). In the most recent 5-year period (~~2013 to 2017~~ 2017 to 2021), there was one observation of a seriously-injured sperm whale in unidentified fishing gear (large gauge line) (Carretta *et al.* 2023).~~were no observations of sperm whale serious injuries or mortalities in commercial fisheries for this stock of sperm whales (Carretta et al. 2019a,2019b~~). There were 3 reports of sperm whales feeding on catch in the limited entry sablefish hook and line fishery, but there was no evidence of entanglement or hooking (Carretta *et al.* 2023). Total mean annual commercial fishery-related serious injury and mortality of sperm whales from 2017-2021 is ~~therefore~~ the sum of mean annual California drift gillnet fishery serious injury and mortality ~~from 2013-2017~~ (~~0.4~~ 0.32 whales) and unidentified fisheries (0.2 whales), or 0.52 whales per year ~~and limited-entry sablefish hook and line estimates (0.24 whales) or 0.64 whales per year~~. (Table 1).

Table 1. Summary of available information on the incidental mortality and serious injury of sperm whales (CA/OR/WA stock) for commercial fisheries that might take this species. n/a indicates that data are not available. Mean annual serious injury and mortality for the California swordfish drift gillnet fishery are based on ~~2013-2017~~ 2017-2021 data unless stated otherwise ~~and annual estimates for the most recent 10-year period are provided for additional context~~(Carretta 2022, Jannot *et al.* 2022).

Fishery Name	Year(s)	Data Type	~~Percent~~ Observer Coverage	Observed mortality (and serious injury in parentheses)	Estimated mortality and serious injury (CV)	Mean annual mortality and serious injury (CV)
CA thresher shark/swordfish drift gillnet fishery	2017	observer	0.186	0	1.58 (2.8)	
	2018		0.251	0	0 (n/a)	
	2019		0.226	0	0 (n/a)	
	2020		0.222	0	0 (n/a)	0.32 (2.8)
	2021		0.228	0	0 (n/a)	
					0 (n/a)	
	~~2008~~					~~0.56 (0.78)~~
	~~2009~~		~~14%~~	~~0~~		
	~~2010~~		~~13%~~	~~0~~	~~0.2 (1.7)~~	
	~~2011~~		~~12%~~	~~1 (1)~~	~~0.3 (2.5)~~	

	~~2012~~ ~~2013~~ ~~2014~~ ~~2015~~ ~~2016~~ ~~2017~~ ~~2013-2017~~	 ~~observer~~	~~20%~~ ~~19%~~ ~~37%~~ ~~24%~~ ~~20%~~ ~~18%~~ ~~19%~~ ~~23%~~	~~0~~ ~~0~~ ~~0~~ ~~0~~ ~~0~~ ~~0~~ ~~0~~ ~~0~~	~~2 (n/a)~~ ~~0.6 (2.7)~~ ~~0.1 (2.1)~~ ~~0.1 (1.5)~~ ~~0.2 (2.7)~~ ~~<0.1 (2.7)~~ ~~0.1 (2.0)~~ ~~2 (1.7)~~ ~~2 (1.4)~~	~~0.4 (1.4)~~
CA/OR/WA limited entry sablefish hook and line	~~2013-2017~~ 2015-2019	observer	n/a	0	0	~~0.25~~ 0
Unidentified fishery	2020	Sighting	n/a	0 (1)	1 (n/a)	0.2
Total annual takes						~~≥ 0.65 (1.4)~~ 0.52 (n/a)

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

Vessel Strikes

~~One sperm whale died as the result of a ship strike in Oregon in 2007 (NMFS Northwest Regional Stranding data, unpublished). Another sperm whale was struck by a 58-foot sablefish longline vessel in 2007 while at idle speed (Jannot *et al.* 2011). The observer noted no apparent injuries to the whale. Based on the size and speed of the vessel relative to the size of a sperm whale, this incident was categorized as a non-serious injury (Carretta *et al.* 2013).~~ For the most recent 5-year period of ~~2013 to 2017~~ 2017-2021, no ~~ship~~ vessel strike deaths or serious injuries were observed, though one was recorded in 2007 (Carretta *et al.* 2013). Due to the low probability of a sperm whale carcass washing ashore, estimated vessel strike deaths are likely underestimated. ~~Ship strikes are assessed over the most recent 5-year period to reflect the degree of shipping risk to large whales since ship traffic routes changed in response to new ship pollution rules implemented in 2009 (McKenna *et al.* 2012, Redfern *et al.* 2013).~~

Other removals

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

STATUS OF STOCK

Sperm whales are listed as "endangered" under the U.S. Endangered Species Act (ESA), and consequently this stock is automatically considered as "depleted" and "strategic" under the MMPA. The status of sperm whales with respect to carrying capacity and optimum sustainable population (OSP) is unknown. The observed annual rate of documented mortality and serious injury (≥ ~~0.4~~ 0.52 per year) is less than the calculated PBR (~~2.5~~ 4.0) for this stock, but anthropogenic mortality and serious injury is likely underestimated due to incomplete detection of carcasses and injured whales. Total human-caused mortality from commercial fisheries is greater than 10% of the calculated PBR and, therefore, is not insignificant and approaching zero mortality and serious injury rate. Increasing levels of anthropogenic sound in the world's oceans has been suggested to be a habitat concern for whales, particularly for deep-diving whales like sperm whales that feed in the ocean's sound channel.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

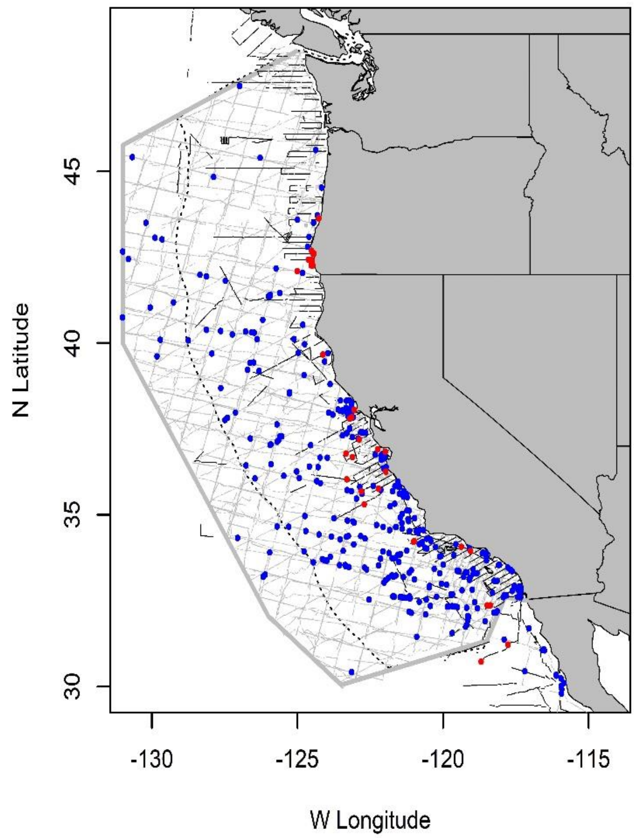

Figure 1. Blue whale sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018. Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

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

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

POPULATION SIZE

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

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

Minimum Population Estimate

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

Current Population Trend

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

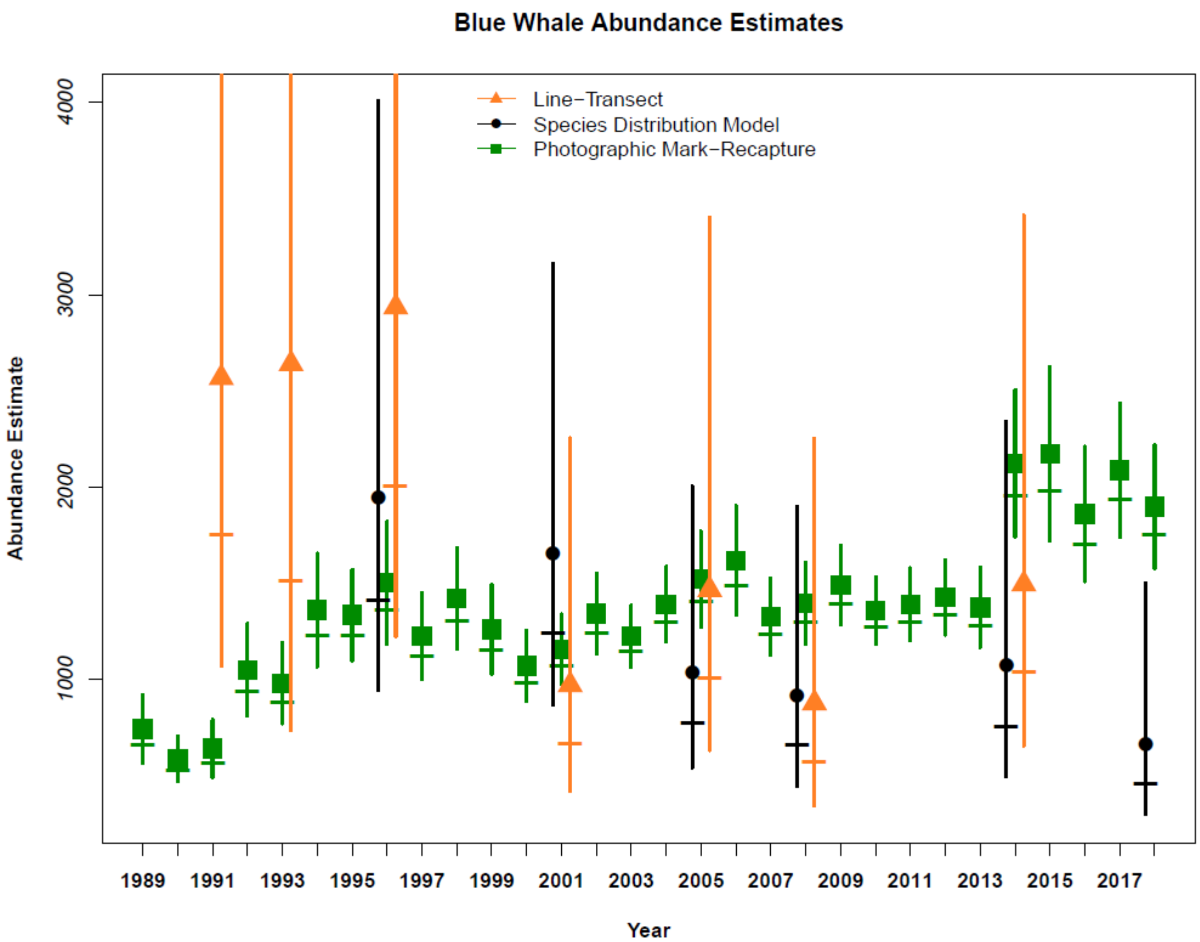

Figure 2. Estimated abundance of blue whales based on three methods (standard vessel-based line transect surveys, habitat-based species distribution models, and a photographic mark-recapture model). The line-transect estimates are based on surveys reported by Barlow (2016). Species distribution model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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Based on mark-recapture estimates from the U.S. West Coast and Baja California, Mexico, Calambokidis *et al*. (2009b) estimated an approximate rate of increase of 3% per year. This estimate is not considered a maximum net productivity rate because it does not account for the effects of anthropogenic mortality and serious injury on the population and therefore likely represents an underestimate of the maximum net productivity rate. For this reason and because an estimate of maximum net productivity is lacking for any blue whale population, the default rate of 4% is used for all blue whale stocks, based on NMFS guidelines for preparing stock assessments (NMFS ~~2016~~2023).

POTENTIAL BIOLOGICAL REMOVAL

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

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Fisheries Information

Blue whales are occasionally documented entangled in pot/trap fisheries and other unidentified fishery gear on the U.S. West Coast (Table 1). The annual entanglement rate of blue whales (observed) during ~~2015-2019~~ 2017-2021 is the sum of observed annual entanglements (~~1.5~~ 0.60/yr), plus species probability assignments (Carretta 2018) from ~~16~~ 4 unidentified whale entanglements (0.014/yr), totaling ~~1.54~~ 0.61 blue whales annually (Table 1). Observed totals represent a negatively-biased accounting of the serious injury and mortality of blue whales in the region, because not all cases are detected and there is no correction factor available to account for undetected events.

Table 1. Summary of available information on observed incidental mortality and injury of blue whales (Eastern North Pacific stock) from commercial fisheries (Carretta *et al.* ~~2021~~2023, Carretta ~~2021~~ 2022). Values in this table represent observed deaths and serious injuries and totals are negatively-biased because not all cases are detected.

Fishery Name	Year(s)	Data Type	~~Percent~~ Observer Coverage	Observed Mortality + Serious Injury	Estimated mortality and/or serious injury (CV in parentheses)	Mean Annual Mortality and Serious Injury (CV in parentheses)
CA Dungeness crab pot	~~2015-2019~~ 2017-2021	Strandings + sightings	n/a	0 + ~~2.75~~ 0.75	n/a	≥ ~~0.55~~ 0.15 (n/a)
~~**Unidentified pot/trap fishery**~~	~~2015-2019~~	~~Strandings + sightings~~	~~n/a~~	~~0 + 1~~	~~n/a~~	~~≥ 0.2 (n/a)~~
Unidentified fishery interactions involving identified blue whales	~~2015-2019~~ 2017-2021	Strandings + sightings	n/a	0 + ~~3.75~~ 2.25	n/a	≥ ~~0.75~~ 0.45 (n/a)
Unidentified fishery interactions involving *unidentified whales* prorated to blue whale	~~2015-2019~~ 2017-2021	Strandings + Sightings	n/a	n/a	~~0.2~~ 0.07	≥~~0.04~~ 0.014
CA/OR thresher shark/swordfish drift gillnet fishery	~~2015-2019~~ 2017 2018 2019 2020 2021	observer	~~21%~~ 0.186 0.251 0.226 0.222 0.228	0	0	0 (n/a)
Total Annual Takes						≥ ~~1.54~~ 0.61 (n/a)

Vessel Strikes

~~Four~~ Three blue whale vessel strike deaths were observed during ~~2015-2019~~ 2017-2021 (Carretta *et al.* ~~2021~~2023), resulting in an observed annual average of ~~0.8~~ 0.6 vessel strike deaths. Observations of blue whale vessel strikes have been highly-variable in previous 5-year periods, with as many as 10 observed (9 deaths + 1 serious injury) during 2007-2011 (Carretta *et al*. 2013). The highest number of blue whale vessel strikes observed in a single year (2007) was 5 whales (Carretta *et al*. 2013). Since 2007, documented vessel strikes have totaled 14 blue whales and ~~7~~ 10 unidentified whales (Carretta *et al*. 2013, ~~2021~~2023). Methods to prorate the number of unidentified whale vessel strike cases to species are not available, because observed sample sizes are small and identified cases are likely biased towards species that are large, easy to identify, and more likely to be detected, such as blue and fin whales. Most observed blue whale vessel strikes have been in southern California or off San Francisco, CA, where blue whales seasonally occur close to shipping ports (Berman-Kowalewski *et al*. 2010). Documented vessel strike deaths and serious injuries are derived from observed whale carcasses and at-sea sightings and are considered minimum values.

Where evaluated, estimates of detection rates of cetacean carcasses are consistently quite low across different regions and species (<1% to 36%), highlighting that observed numbers are unrepresentative of true impacts (Kraus *et al*. 2005, Pace *et al*. 2021, Perrin *et al*. 2011, Williams *et al*. 2011, Prado *et al*. 2013). Due to this negative bias, Redfern *et al*. (2013) noted that the number of observed vessel strike deaths of blue whales in the U.S. West Coast EEZ likely exceeds PBR.

Vessel strike mortality was estimated for blue whales in the U.S. West Coast EEZ (Rockwood *et al*. 2017), using an encounter theory model (Martin *et al.* 2016) that combined species distribution models of whale density (Becker *et al*. 2016), vessel traffic characteristics (size + speed + spatial use), along with whale movement patterns obtained from satellite-tagged whales in the region to estimate encounters that would result in mortality. The estimated number of annual vessel strike deaths was 18 blue whales, though this includes only the period July – November when whales are most likely to be present in the U.S. West Coast EEZ and was based on cetacean habitat models generated from line-transect surveys (Becker *et al*. 2016, Rockwood *et al*. 2017). This estimate was also based on an assumption of a moderate level of vessel avoidance (55%) by blue whales, as measured by the behavior of satellite-tagged whales in the presence of vessels (McKenna *et al*. 2015). The estimated mortality of 18 blue whales annually due to vessel strikes represents approximately 1% of the most recent estimated population size of the stock (18 deaths / 1,898 whales). The results of Rockwood *et al*. (2017) also include a no-avoidance encounter model that results in a worst-case estimate of 40 annual blue whale vessel strike deaths, which represents 2.1% of the estimated population size. The authors ~~also~~ note that 74% of blue whale vessel strike mortalities occur within 10% of the study area, implying that vessel avoidance mitigation measures can be effective if applied over relatively small regions. Using the moderate level of avoidance model from Rockwood *et al*. (2017), estimated vessel strike deaths of blue whales are 18 annually. A comparison of average annual vessel strikes observed over the period ~~2015-2019~~ 2017-2021 (~~0.8~~ 0.6/yr) versus estimated vessel strikes (18/yr) indicates that the rate of detection for blue whale vessel strikes is approximately 4 3%. Comparing the highest number of vessel strikes observed in a single year (5 in 2007) with the estimated annual number (18) implies that vessel strike detection rates have not exceeded 28% (5/18) in any single year.

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

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

~~Habitat Issues~~

~~Increasing levels of anthropogenic sound in the world's oceans is a habitat concern for blue whales (Reeves *et al*. 1998, Andrew *et al*. 2002). Tagged blue whales exposed to simulated mid-frequency sonar and pseudo-random noise demonstrated a variety of behavioral responses, including no change in behavior, termination of deep dives, directed travel away from sound sources, and cessation of feeding (Goldbogen *et al*. 2013, Southall *et al.* 2019). Behavioral responses were highly dependent upon the type of sound source, distance from sound sources, and the behavioral state of the animal at the time of exposure. Deep-feeding and non-feeding whales reacted more strongly to experimental sound sources than surface-feeding whales that typically showed no change in behavior (Goldbogen *et al*. 2013, Southall *et al.* 2019). Both studies noted that behavioral responses to such sounds are influenced by a complex interaction of behavioral state, environmental context, and prior exposure of individuals to such sound sources. One concern expressed in both studies is if blue whales did not habituate to such sounds near feeding areas, that chronic~~

~~cessation of feeding behavior could affect the fitness of individual whales, which could impact population fitness (Goldbogen *et al*. 2013, Southall *et al.* 2019). Currently, no evidence indicates that such reduced population health exists, but such evidence would be difficult to differentiate from natural sources of reduced fitness or mortality in the population. Nine blue whale feeding areas identified off the California coast by Calambokidis *et al*. (2015) represent a diversity of nearshore and offshore habitats that overlap with a variety of anthropogenic activities, including shipping, oil and gas extraction, and military activities.~~

STATUS OF STOCK

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

The sum of observed and assigned annual incidental mortality and serious injury due to commercial fisheries (≥~~1.54~~ 0.61 /yr), plus estimated vessel strike deaths (18/yr), is ~~19.5~~ 18.6 whales annually for ~~2015-2019~~ 2017-2021. This exceeds the calculated PBR of 4.1 for this stock. Monnahan *et al*. (2015) proposed that estimated vessel strike levels of 10 – 35 whales annually did not pose a threat to the status of this stock, but estimates of carrying capacity of this blue whale stock differed depending on the level of vessel strikes: 97% of K with 10 annual strikes and 91% of K with 35 annual strikes. The highest estimates of blue whale vessel strike mortality (35/yr; Monnahan *et al*. (2015) and 40/yr; Rockwood *et al*. (2017) are similar, and annually represent approximately 2% of the estimated population size. Observed and assigned levels of serious injury and mortality due to commercial fisheries (≥ ~~1.54~~ 0.61) exceed 10% of the stock's PBR (4.1), thus, commercial fishery take levels are not approaching zero mortality and serious injury rate.

OTHER FACTORS THAT MAY BE CAUSING A DECLINE OR IMPEDING RECOVERY

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

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

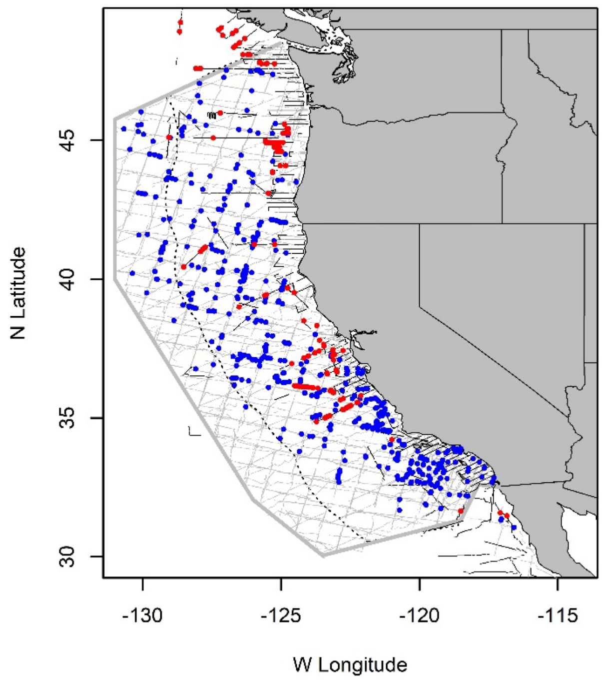

Figure 1. Fin whale sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018 (Barlow 2016, Henry *et al*. 2020). Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

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

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

POPULATION SIZE

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

Minimum Population Estimate

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

Current Population Trend

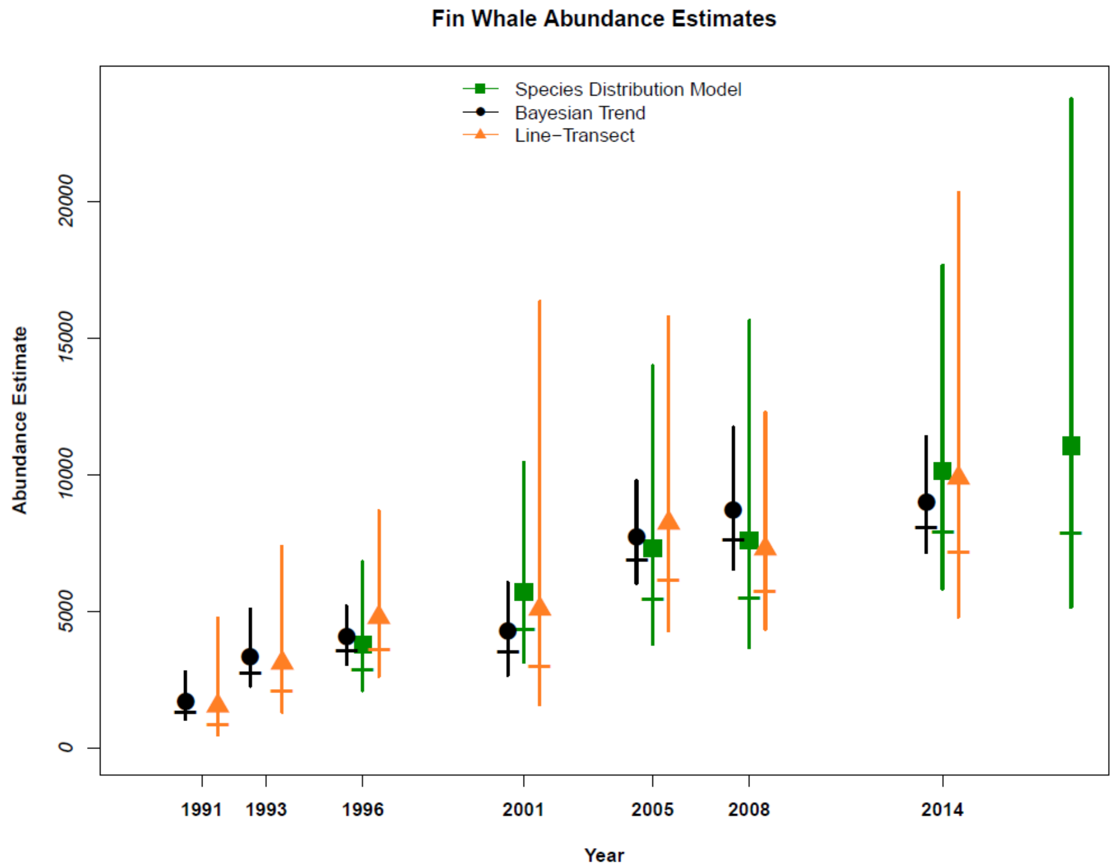

Figure 2. Fin whale abundance estimated from three methods (standard vessel-based line transect surveys (Barlow 2016), habitat-based species distribution models (Becker *et al*. 2020), and a Bayesian trend analysis (Nadeem *et al*. 2016). Vertical bars indicate approximate 95% log-normal confidence limits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

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

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

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Fisheries Information

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

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

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

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

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

Vessel Strikes

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

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

STATUS OF STOCK

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

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

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

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

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Revised ~~2/15/2019~~ 10/16/2023

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

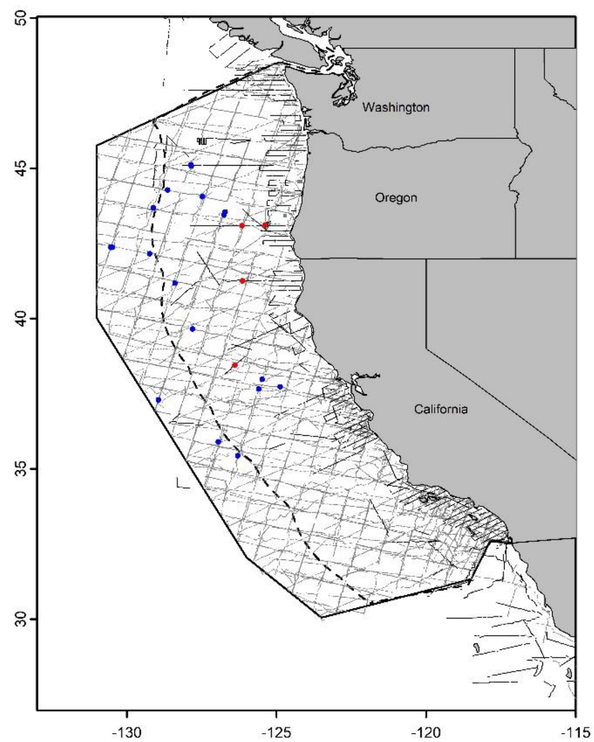

Figure 1. Sei whale sighting locations from shipboard surveys off California, Oregon, and Washington, 1991-~~2014~~2018. Dashed line represents the U.S. EEZ; thin lines indicate completed transect effort of all surveys combined. Sightings from the 2018 survey are shown in red.

POPULATION SIZE

Ohsumi and Wada (1974) estimated the pre-whaling abundance of sei whales to be 58,000-62,000 in the North Pacific. Tillman (1977) estimated sei whale abundance in the North Pacific and revised this pre-whaling estimate to 42,000. His estimates for the year 1974 ranged from 7,260 to 12,620. These previous studies depended on using the history of catches and trends in CPUE or sighting rates. Hakamada *et al*. (2017) estimated sei whale abundance at 29,632 sei whales (CV = 0.242, 95% CI 18,576–47,267) in the central and eastern North Pacific based on visual line-transect surveys between 2010 and 2012. This estimate corresponds with the first systematic sighting survey abundance estimate for this species over a pelagic high-seas region. However, while the study area of Hakamada *et al*. (2017) included waters north of 40ºN latitude and west of

135°W longitude, it excluded waters of the California Current, where sightings are rare (Barlow 2016). The ~~estimated number~~ most-recent estimate of sei ~~whales~~ whale abundance in the California Current is based on a 2014 survey (864, CV=0.40), however, this estimate is now 9 years old. Although there was no formal assessment of an abundance trend for sei whales, estimates reported in Barlow (2016) showed an increasing trend from 1991-2014, with the 2014 estimate being the highest estimated. ~~ship line-transect surveys between 1991-2014 within 300 nmi of the U.S. West Coast, where sightings are relatively rare (Fig. 1, Barlow 2016). Abundance estimates for the two most recent line transect surveys of California, Oregon, and Washington waters in 2008 and 2014 are 311 (CV=0.76) and 864 (CV=0.40) sei whales, respectively (Barlow 2016). The best estimate of abundance for California, Oregon, and Washington waters is the unweighted geometric mean of the 2008 and 2014 estimates, or 519 (CV=0.40) sei whales (Barlow 2016).~~

Minimum Population Estimate

~~The minimum population estimate for sei whales is taken as the lower 20th percentile of the log-normal distribution of abundance estimated from 2008 and 2014 vessel line-transect surveys, or 374 whales.~~ Although the most-recent abundance estimate for this stock is from 2014, an estimate of minimum population size may be inferred by assuming the population size has at least been stable over the period 1991-2014 (Barlow 2016), based on increasing estimates of abundance. Thus, the minimum population size is calculated from the most-recent estimate (864, CV=0.40), resulting in a minimum population size of 625 whales.

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (~~374~~ 625) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.1 (for an endangered species), resulting in a PBR of ~~0.75~~ 1.25 whales.

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Fishery Information

The California swordfish drift gillnet fishery is the most likely U.S. fishery to interact with sei whales from this stock, but no entanglements have been observed from ~~8,845~~ 9,246 ~~monitored~~ observed fishing sets from 1990-~~2016~~ 2021 (Carretta ~~*et al.* 2018a~~ 2022, Table 1). Mean annual takes for this fishery (Table 1) are based on ~~2012-2016~~ 2017-2021 data and are zero whales annually. However, some gillnet mortality of large whales may go unobserved because whales swim away with a portion of the net.

Table 1. Summary of available information on the incidental mortality and injury of sei whales (eastern North Pacific stock) for commercial fisheries that might take this species. n/a indicates that data are not available. Mean annual takes are based on ~~2012-2016~~ 2017-2021 data unless noted otherwise.

Fishery Name	Year(s)	Data Type	~~Percent~~ Observer Coverage	Observed mortality (and injury in parentheses)	Estimated mortality (CV in parentheses)	Mean annual takes (CV in parentheses)
CA/OR thresher shark/swordfish drift gillnet fishery	2017 2018 2019 2020 2021 ~~2012~~ ~~2013~~ ~~2014~~ ~~2015~~	observer	0.186 0.251 0.226 0.222 0.228 ~~19%~~ ~~37%~~ ~~24%~~ ~~20%~~	0	0	0 (n/a)

	~~2016~~		~~18%~~			

Vessel Strikes

No vessel strikes of sei whales have been documented during the most-recent 5-yr period (2017-2021) (Carretta *et al*. 2023). ~~One documented ship strike of a sei whale occurred in the most recent 5-year period, 2012-2016 (Carretta *et al.* 2018b), although uncertainty over whether the strike occurred pre- or post-mortem exists. For purposes of this stock assessment report, the ship strike is considered as the probable cause of death. During 2012-2016, there was one additional serious injury of an unidentified large whale attributed to a ship strike. Additional ship strike mortality probably goes unreported because the whales do not strand or, if they do, they may not have obvious signs of trauma. The average observed annual mortality due to ship strikes is 0.2 sei whales per year for the period 2012-2016.~~

STATUS OF STOCK

The NMFS sei whale recovery plan notes that basic data such as distribution, abundance, trends and stock structure is of poor quality or largely unknown, owing to the rarity of sightings of this species (NMFS 2011). Sei whales were estimated to have been reduced to 20% (8,600 out of 42,000) of their pre-whaling abundance in the North Pacific (Tillman 1977). The initial abundance has never been reported separately for the eastern North Pacific stock, but this stock was also depleted by whaling. Kanda *et al*. (2006) found a high level of genetic variation among sei whale samples in the western North Pacific and hypothesized that the population did not suffer from a genetic bottleneck due to commercial whaling. Sei whales are formally listed as "endangered" under the Endangered Species Act (ESA), and consequently the eastern North Pacific stock is automatically considered a "depleted" and "strategic" stock under the MMPA. Total observed fishery mortality is zero and therefore is considered to be approaching zero mortality and serious injury rate. ~~The current known rate of ship strike deaths and serious injuries is 0.2 annually, but most sei whale ship strikes are likely unreported.~~ Risks to sei whales include vessel strikes, though none were recorded in the most-recent 5-yr period (Carretta *et al*. 2023). Increasing levels of anthropogenic sound in the world's oceans is a habitat concern for whales, particularly for baleen whales that may communicate using low-frequency sound (Croll *et al.* 2002). Behavioral changes associated with exposure to simulated mid-frequency sonar, including no change in behavior, cessation of feeding, increased swimming speeds, and movement away from simulated sound sources has been documented in tagged blue whales (Goldbogen *et al.* 2013), but it is unknown if sei whales respond in the same manner to such sounds.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

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

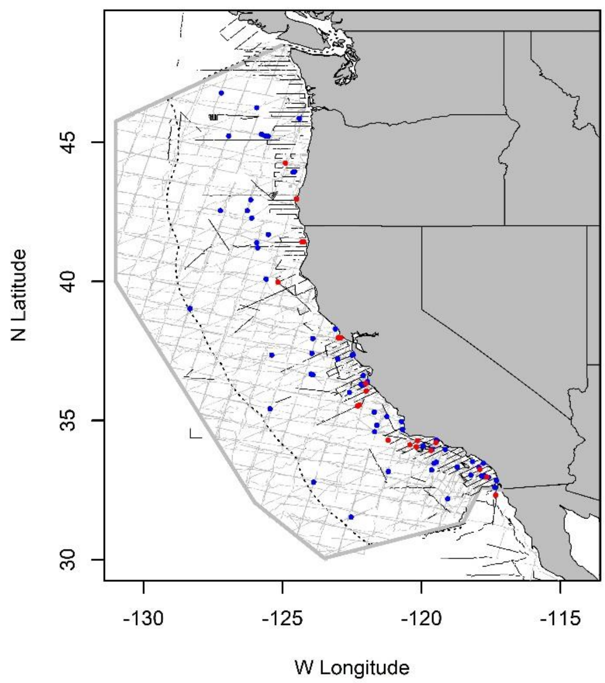

Figure 1. Minke whale sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018. Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

POPULATION SIZE

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

Minimum Population Estimate

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

Current Population Trend

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

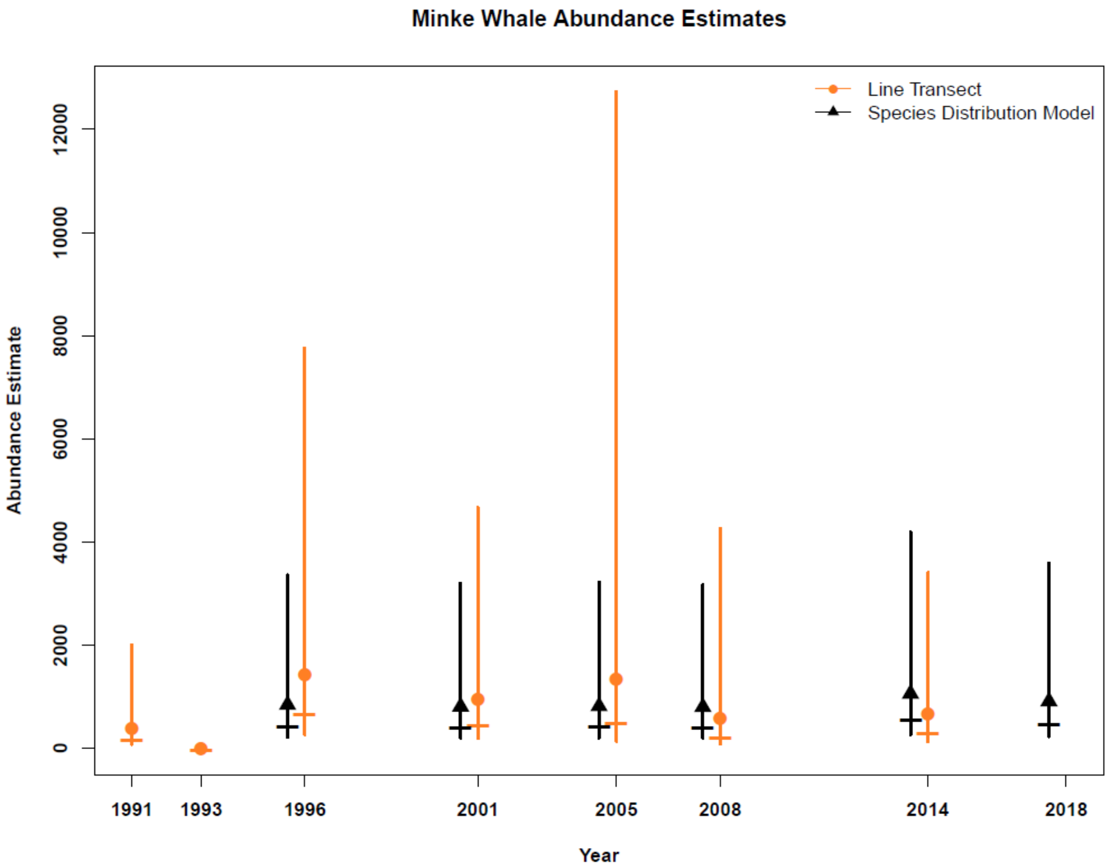

Figure 2. Minke whale abundance estimated from vessel-based line transect surveys (Barlow 2016) and habitat-based species distribution models based on 1991-2018 line-transect surveys (Becker *et al*. 2020). Vertical bars indicate approximate 95% log-normal confidence limits for line-transect estimates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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

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

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Table 1. Summary of available information on the incidental mortality and injury of minke whales (CA/OR/WA stock) for commercial fisheries that might take this species (Carretta ~~2020~~2022, Carretta *et al.* ~~2021~~2023). Mean annual takes are based on ~~2015-2019~~2017-2021 data.

Fishery Name	Years	Data Type	~~Percent~~ Observer Coverage	Observed mortality (and serious injury)	Estimated Mortality (CV)	Mean Annual Takes (CV)
CA/OR thresher shark/swordfish gillnet fishery	2017 2018 2019 2020 2021 ~~2015-2019~~	Observer	0.186 0.251 0.226 0.222 0.228 ~~21%~~	0	~~1.2 (0.99)~~ 0.1 (3.9)	~~0.24 (0.99)~~ 0.02 (3.9)
CA halibut and other species large mesh (>3.5") set gillnet fishery	2017	Observer	~10%	0	0	0 (n/a)
Dungeness Crab Pot Fishery (Oregon)	2021	Sighting	n/a	0 (0)	0	0 (n/a)
Unidentified fisheries	~~2015-2019~~ 2017-2021	Sightings and strandings	n/a	~~1~~ 0 (0.75)	~~1.75~~ 0.75 (n/a)	≥ ~~0.35~~ 0.15 (n/a)
Total annual takes						≥ ~~0.59~~ 0.17 (~~0.99~~n/a)

Fishery Information

Minke whales may occasionally be caught in coastal set gillnets off California, in salmon drift gillnet in Puget Sound, Washington, and in offshore drift gillnets off California. The most-recent estimate of bycatch in the California swordfish drift gillnet fishery is ~~1.2 (CV=0.99)~~ 0.1 (CV=3.9) whales for the 5-year period ~~2015-2019~~2017-2021, or ~~0.24~~ 0.02 whales annually (Carretta ~~2021~~2022, Table 1). This is a model-based estimate based on a total of four minke whales observed entangled (2 dead, 2 released alive) between 1990-~~2019~~ 2021 from ~~9,158~~ 9,246 observed fishing sets (Carretta ~~2021~~2022). Two additional unidentified fishery interactions with minke whales were recorded during 2015-2019, totaling ~~1.75~~ 0.75 serious injuries/deaths (Carretta *et al.* ~~2021~~2023). One minke whale was disentangled from commercial Dungeness crab pot gear (Oregon) in 2021; the initial and final injury status were non-serious (Carretta *et al*. 2023). The mean annual mortality and serious injury of minke whales from this stock during ~~2015-2019~~ 2017-2021 is ~~0.59~~ 0.17 animals (Table 1).

Vessel Strikes

No vessel strikes of minke whales were reported during the most recent 5-years, ~~2015 to 2019~~2017 to 2021, but most strikes are likely to go undetected compared to larger baleen whales where estimates of vessel strike detection are generally <10% (see blue and fin whale stock assessments).

Other Mortality

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

STATUS OF STOCK

Minke whales are not listed as "endangered" under the Endangered Species Act and are not considered "depleted" under the MMPA. The annual mortality and serious injury due to fisheries (~~0.59~~ 0.17/yr), shootings (0.2/yr) and vessel strikes (0.0/yr) is less than the calculated PBR for this stock (4.1), so they are not considered a "strategic" stock under the MMPA. Estimated fishery mortality is ~~not~~ less than 10% of the PBR; therefore, total fishery mortality is ~~not~~ approaching zero mortality and serious injury rate. Trends in the abundance of this stock are unknown. Harmful algal blooms are a habitat concern for minke whales and at least one death along the U.S. west coast has been attributed to domoic acid toxicity resulting from the consumption of northern anchovy prey (Fire *et al.* 2010). Increasing levels of anthropogenic sound in the world's oceans has been suggested to be a habitat concern for whales, particularly for baleen whales that may communicate using low-frequency sound (Croll *et al*. 2002). Behavioral changes associated with exposure to simulated mid-frequency sonar, including no change in behavior, cessation of feeding, increased swimming speeds, and movement away from simulated sound sources has been documented in tagged blue whales (Goldbogen *et al*. 2013), but it is unknown if minke whales respond in the same manner to such sounds.

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Revised ~~5/5/2021~~ 08/16/2023

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

Rough-toothed dolphins are found throughout the world in tropical and warm-temperate waters (Perrin *et al*. 2009), with genetic analysis of samples from all ocean basins revealing divergence between rough-toothed dolphins in the Atlantic versus those in the Indian and Pacific Oceans, suggesting the occurrence of two subspecies (Albertson *et al*. 2022). They are present around all the main Hawaiian Islands, though are relatively uncommon within the ~~near~~ Maui Nui region (Baird *et al*. 2013), and have been observed close to the islands and atolls at least as far northwest as Pearl and Hermes Reef (Bradford *et al*. 2017). Rough-toothed dolphins are~~were~~ occasionally seen offshore throughout the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands during periodic shipboard surveys~~both 2002, 2010, and 2017 surveys (Barlow 2006, Bradford et al. 2017, Yano *et al*. 2018;~~ (Figure 1). Rough-toothed dolphins have also been documented in American Samoan waters (Oleson 2009).

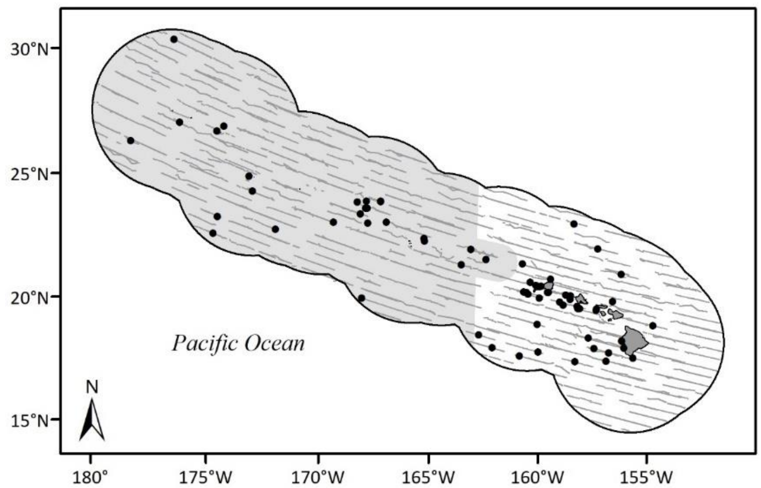

Figure 1. Rough-toothed dolphin sighting locations (circles) and survey effort (gray lines) during the 2002 (~~diamond~~Barlow 2006), 2010 (~~circles~~Bradford *et al*. 2017), and 2017 (~~squares~~Yano *et al.* 2018) shipboard cetacean surveys of the U.S. EEZ ~~waters surrounding~~ around the Hawaiian Islands (~~Barlow 2006, Bradford *et al*. 2017, Yano *et al*. 2018~~outer black line). ~~Outer line represents approximate boundary of survey area and U.S. EEZ. Dark g~~Gray ~~shading indicates area of t~~The ~~original~~ Papahānaumokuākea Marine National Monument in the western portion of the EEZ is shaded gray~~, with the lighter gray shading denoting the full 2016 Expansion area. Dotted line represents the 1000 m isobath~~.

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

For the Marine Mammal Protection Act (MMPA) stock assessment reports, there are two Pacific management stocks: 1) t~~T~~he Hawai‘i Stock (this report), and 2) the American Samoa Stock. The Hawai‘i~~an~~ stock includes animals found both within the Hawaiian Islands EEZ and in adjacent high seas waters; however, because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of this stock is evaluated based on data from the U.S. EEZ ~~waters~~ of the Hawaiian Islands (NMFS ~~2005~~2023a).

POPULATION SIZE

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

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

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

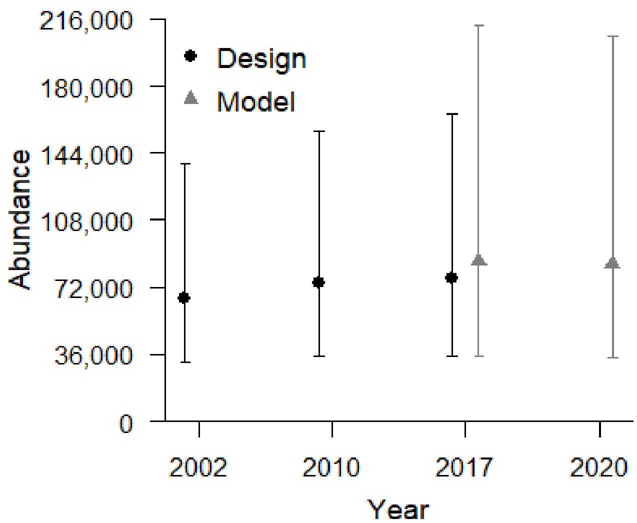

Figure 2. Comparison of design-based (black circles, Bradford *et al.* 2021) and model-based (gray triangles, Becker *et al.* 2021, 2022) estimates of abundance for rough-toothed dolphins for each survey year (2002, 2010, 2017, 2020).

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

~~The updated design-based abundance estimates use sighting data from throughout the central Pacific to estimate the detection function and use Beaufort sea-state-specific trackline detection probabilities for rough-toothed dolphins from Barlow *et al.* (2015). Although previous estimates from the Hawaii EEZ have been published using subsets of this data, Bradford *et al.* (2021), uses a consistent approach for estimating all abundance parameters and the resulting~~

~~estimates are considered the best available for each survey year. Model-based density and abundance estimates were also built for rough-toothed dolphins (Becker *et al*. 2021); however, only static geographic and depth variables were selected within the modeling process, precluding evaluation of inter-annual changes in density relative to other dynamic variables. The best estimate is based on the 2017 survey, or 76,375 (CV=0.41).~~

A population estimate for this species has been made in the eastern tropical Pacific (Wade and Gerrodette 1993), but it is not known whether these animals are part of the same population that occurs around the Hawaiian Islands.

Mark-recapture estimates for the islands of Kauaʻi/Niʻihau and Hawaiʻi were derived from identification photographs obtained between 2003 and 2006, resulting in estimates of 1,665 (CV=0.33) around Kauaʻi/Niʻihau and 198 (CV=0.12) around the island of Hawaiʻi (Baird *et al*. 2008). Such estimates may be representative of smaller island-associated populations at those island areas.

Minimum Population Estimate

The minimum population estimate~~size~~ is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al*. 1995) of the 2020~~2017~~ abundance estimate (from Becker *et al.* 2022) or 56,782~~54,804~~ rough-toothed dolphins within the Hawaiian Islands EEZ.

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Hawaiʻi stock of rough-toothed dolphins is calculated as the minimum population size within the U.S. EEZ of the Hawaiian Islands (56,782~~54,804~~) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.45 (for a stock of unknown status with Hawaiian Islands EEZ mortality and serious injury rate CV > 0.30~~no known Hawaiian Islands EEZ fishery mortality and serious injury~~; Wade and Angliss 1997), resulting in a PBR of 511~~548~~ rough-toothed dolphins per year.

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Fishery Information

Information on fishery-related mortality and serious injury of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaiian fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. Rough-toothed dolphins are known to take bait and catch from several Hawaiian sport and commercial fisheries operating near the main islands (Shallenberger 1981; Schlais 1984; Nitta & Henderson 1993). They have been specifically reported to interact with the day handline fishery for tuna (palu-ahi), the night handline fishery for tuna (ika-shibi), and the troll fishery for billfish and tuna (Schlais 1984; Nitta & Henderson 1993). Baird *et al*. (2008) reported increased vessel avoidance of boats by rough-toothed dolphins off the island of Hawaiʻi relative to those off Kauaʻi or Niʻihau and attributed this to possible shooting of dolphins that are stealing bait or catch from recreational fisherman off the island of Hawaiʻi (Kuljis 1983). Rough-toothed dolphins have been observed in nearshore waters with serious injuries resulting from fishing gear trailing from or wrapped around their bodies, though the source of the gear was not identified (Bradford and Lyman 2018). ~~In 2014 a rough-toothed dolphin was observed off the Kona coast trailing 25-30 ft. of heavy line with two plastic jugs attached to the end of the line (Bradford and Lyman 2018). The jugs were cut from the gear when other attempts (through pressure on the line) did not result in the removal of any other line or hooks, though all other trailing gear remained on the dolphin. This dolphin was considered seriously injured based on the amount of trailing gear. The source of the gear is not known. In 2015 a rough-toothed dolphin was observed with line tightly wrapped around and cutting into its left pectoral flipper, with 3-4 ft. of line trailing behind (Bradford and Lyman 2018). This dolphin was considered seriously injured based on information available at the time of report. This dolphin was subsequently sighted twice free of gear in 2018, indicating it survived the entanglement. As such, the serious injury determination has been revised and the dolphin is considered to be not seriously injured (Bradford and Lyman 2020).~~ Photographs of 52 individuals with greater than 50% of the mouthline photographed showed evidence of injuries consistent with interactions with hook and line fisheries (Welch 2017). No estimates of human-caused mortality or serious injury are currently available for nearshore hook and line fisheries because these fisheries are not observed or monitored for protected species bycatch.

There are currently two distinct longline fisheries based in Hawaiʻi: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas, but are prohibited from operating within the Papahānaumokuākea Marine National Monument (PMNM) and within the Longline Exclusion Zone around the main Hawaiian Islands and the Pacific Remote Islands and Atolls (PRIA) MNM around Johnston Atoll. The PMNM originally included the waters within a 50 nmi radius around the Northwestern Hawaiian Islands. ~~As of~~In August, 2016, the PMNM area was expanded to extend to the 200 nmi EEZ boundary west of 163° W. Between 2017~~2014~~ and 2021~~2018~~, no rough-toothed dolphins were observed hooked or entangled in the SSLL fishery (100% observer coverage), but ~~one~~eight rough-toothed dolphins were ~~was~~ observed taken~~hooked or entangled~~ in the DSLL fishery (~~18~~15-21% observer coverage) (Figure 3, Bradford ~~2018a,~~ 2018~~b~~, 2020, In press, In review,

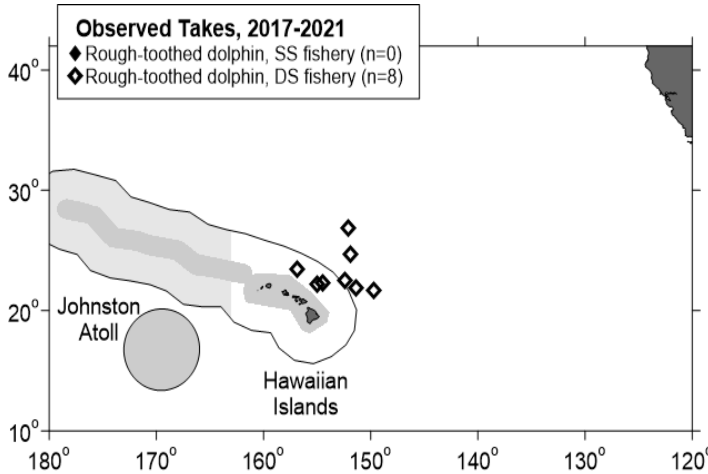

Figure ~~2~~3. Locations of observed rough-toothed dolphin takes within the deep-set fishery (~~filled~~ open diamonds) ~~and unidentified cetacean that maybe rough-toothed dolphins based on the observer's description (crosses)~~ in the Hawaii-based longline fishery, ~~2014-2018~~2017-2021. Solid lines represent the U.S. EEZ. Gray shading notes areas closed to longline fishing~~, with the PMNM Expansion area closed since August 2016~~.

Table 2. Summary of available information on incidental mortality and serious injury (MSI) of rough-toothed dolphins (McCracken & Cooper 2022b ~~2019~~). Mean annual takes are based on 2017-2021~~2014-2018~~ data unless indicated otherwise. Information on all observed takes (T) and ~~combined mortality events and serious injuries~~ (MSI~~)~~ is included. Total takes were prorated to deaths, serious injuries, and non-serious injuries based on the observed proportions of each outcome.

Fishery Name	Year	Data Type	Percent Observer Coverage	~~Observed total interactions (T) and mortality events, and serious injuries (MSI), and total estimated mortality and serious injury (M&SI) of rough-toothed dolphins~~ Outside U.S. EEZ~~s~~: Observed~~.~~ T/MSI	Outside U.S. EEZ~~s~~: Estimated M&SI (CV)	Hawaiian Islands EEZ: Obs~~.~~erved T/MSI	Hawaiian Islands EEZ: Estimated M&SI (CV)
Hawaiʻi-based deep-set longline fishery	~~2014~~	Observer data	~~21%~~	~~0~~	~~0 (-)~~	~~0~~	~~0 (-)~~
	~~2015~~		~~21%~~	~~0~~	~~0 (-)~~	~~0~~	~~0 (-)~~
	~~2016~~		~~20%~~	~~1/1~~	~~5 (0.9)~~	~~0~~	~~0 (-)~~
	2017		20%	0	0 (-)	0	0 (-)
	2018		18%	0	0 (-)	0	0 (-)
	2019		21%	1/0	3 (1.2)	0	0 (-)
	2020		15%	3/2	14 (0.5)	2/2	10 (0.6)
	2021		18%	1/1	5 (0.9)	1/1	6 (0.9)
Mean Estimated Annual Take (CV) 2017-2021					4.4 (0.5)~~1.0 (1.6)~~		~~0 (-)~~3.2 (0.6)
Hawaiʻi-based shallow-set longline fishery	~~2014~~	Observer data	~~100%~~	~~0~~	~~0~~	~~0~~	~~0~~
	~~2015~~		~~100%~~	~~0~~	~~0~~	~~0~~	~~0~~
	~~2016~~		~~100%~~	~~0~~	~~0~~	~~0~~	~~0~~
	2017		100%	0	0	0	0
	2018		100%	0	0	0	0
	2019		100%	0	0	0	0
	2020		100%	0	0	0	0
	2021		100%	0	0	0	0
Mean Annual Takes (100% coverage) 2017-2021					0		0
Minimum total annual takes within U.S. EEZ (2017-2021)							3.2 (0.6)

~~Bradford and Forney 2017,~~ McCracken & Cooper 2022b~~2019~~). In the DSLL fishery, 5 rough-toothed dolphins were taken outside the U.S. EEZ, including 1 rough-toothed dolphin found dead, 2 considered seriously injured, and 1 considered non-seriously injured based on an evaluation of the observer's description of each interaction and following criteria for assessing serious injury in marine mammals (NMFS 2023b) ~~based on the information provided by the observer~~. Inside of the Hawaiian Islands EEZ, 2 were observed dead and 1 determined to be seriously injured.

The total estimated number of dead or seriously injured dolphins is calculated based on observer coverage rate, the location of the observed take (inside or outside of the U.S. EEZ~~s~~), and the ratio of observed dead and seriously injured dolphins versus those judged to be not seriously injured. Observer coverage is measured on a per-trip basis throughout the calendar year as described by McCracken (2019). In years with large fluctuations in observer coverage, such as during the early days of the COVID-19 pandemic when observer coverage dropped to less than 10% during the second quarter of the year, the annual bycatch estimation process may be subset into several periods, as described in McCracken and Cooper (2022a). Average 5-yr estimates of annual mortality and serious injury for rough-toothed dolphins during 2017-2021~~2014-2018~~ are 4.4 (CV=0.5) rough-toothed dolphins outside of the Hawaiian Islands EEZ~~s~~, and 3.2 (CV=0.6) ~~zero~~ rough-toothed dolphins within the Hawaiian Islands EEZ ~~and 1.0 (CV = 1.6) dolphins outside of U.S. EEZs~~ (Table 2, McCracken and Cooper 2022b~~2019~~).

STATUS OF STOCK

The Hawaiʻi stock of rough-toothed dolphins is not considered strategic under the 1994 amendments to the MMPA, The status of rough-toothed dolphins in Hawaiian waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Rough-toothed dolphins are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. ~~Two rough-toothed dolphins have been observed entangled in gear, though no dolphins have been killed or seriously injured in the deep-set longline fishery.~~ There have been no reports of recent mortality or serious injuries related to nearshore fisheries; however, there is no systematic monitoring for interactions with protected species within nearshore fisheries that may take this species, thus total mean annual takes are undetermined. The total number of estimated rough-toothed dolphins killed or seriously injured by longline fisheries ~~rough-toothed dolphin~~ inside (3.2~~zero~~) and outside (4.4~~1.0~~) of the Hawaiian Islands EEZ is less than 10% of PBR (51~~548~~), such that the fishery-related mortality or serious injuries rate for the entire Hawaiʻi stock can be considered to be insignificant and approaching zero. Island-associated populations of rough-toothed dolphins may experience relatively greater rates of fisheries mortality and serious injury. One rough-toothed dolphin stranded in the main Hawaiian Islands tested positive for *Brucella* (Chernov 2010) and another for *Morbillivirus* (Jacob 2012). *Brucella* is a bacterial infection that, if common in the population, may limit recruitment by compromising male and female reproductive systems, and can also cause neurological disorders that may result in death (Van Bressem *et al.* 2009). Although *morbillivus* is known to trigger lethal disease in cetaceans (Van Bressem *et al.* 2009), its impact on the health of the stranded animal is not known as it was found in only a few tested tissues (Jacob *et al.* 2016). The presence of *morbillivirus* in 10 species (Jacob *et al*. 2016) and *Brucella* in 3 species (Chernov 2010) raises concerns about the history and prevalence of these diseases in Hawaii and the potential population impacts, including cumulative impacts of disease with other stressors.

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Revised ~~5/5/2021~~ 08/16/2023

RISSO'S DOLPHIN (*Grampus griseus*): Hawai‘i Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Risso's dolphins are found in tropical to warm-temperate waters worldwide (Perrin *et al*. 2009). Risso’s dolphins represent less than 1% of all odontocete sightings in leeward surveys of the main Hawaiian Islands from 2000 to 2012 (Baird *et al*. 2013); however, they are regularly sighted during periodic ~~six sightings were made during a 2002 survey, 12 during a 2010 survey, and 12 during a 2017~~ shipboard surveys of the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands ~~(Barlow 2006, Bradford et al. 2017, Yano *et al*. 2018;~~ (Figure 1). Most sightings of Risso’s dolphins occur in deep waters offshore. A single satellite tagged animal moved broadly between offshore waters off Kona, Koho‘olawe, and Lāna‘i over a 2-—week period (Baird 2016). Sighting, habitat, and limited movement data do not appear to support finer population structure in Hawaiian waters, though differences in the spectral characteristics of Risso’s dolphin echolocation clicks between Hawai‘i and the U.S. West Coast suggest there may be an indication of population differentiation within the ocean basin (Soldevilla *et al.* 2017).

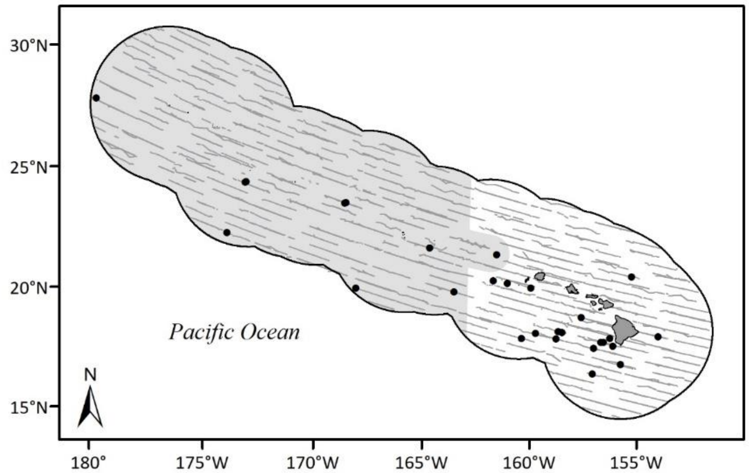

Figure 1. Risso's dolphin sighting locations (circles) and survey effort (gray lines) during the 2002 (~~diamonds~~Barlow 2006), 2010 (~~circle~~Bradford *et al.* 2017), and 2017 (~~square~~Yano *et al.* 2018) shipboard cetacean surveys of the U.S. EEZ ~~waters surrounding~~around the Hawaiian Islands (~~Barlow 2006, Bradford *et al*. 2014, Yano *et al*. 2018~~outer black line). ~~Outer line represents approximate boundary of survey area and U.S. EEZ. Dark and light gGray shading indicates the original and 2016 Expanded area of t~~The Papahānaumokuākea Marine National Monument in the western portion of the EEZ is shaded gray. ~~Dotted line is the 1000 m isobath.~~

For the Marine Mammal Protection Act (MMPA) stock assessment reports, Risso's dolphins within the Pacific U.S. EEZ are divided into two discrete, non-contiguous areas: 1) Hawaiian waters (this report), and 2) waters off California, Oregon and Washington. The Hawai‘~~ian~~ stock includes animals found both within the Hawaiian Islands EEZ and in adjacent high seas waters; however, because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of this stock is evaluated based on data from U.S. EEZ ~~waters~~ of the Hawaiian Islands (NMFS ~~2005~~2023).

POPULATION SIZE

Encounter data from shipboard line-transect surveys of the ~~entire~~ Hawaiian Islands EEZ were~~was~~ recently reevaluated for each survey year, resulting in ~~the~~ updated model-based abundance estimates of Risso’s dolphins in the entirety of the Hawaiian Islands EEZ (Becker *et al*. 2021, 2022; Table 1).

Sighting data from 2002 to 2020~~2017~~ within the Hawaiian Islands EEZ were used to derive habitat-based models of animal density for two periods: 2002-2017 (Becker *et al.* 2021) and 2017-2020 (Becker *et al.* 2022). The most recent set of models include three notable changes from the 2002-2017 models: use of calibrated group size estimates, as in Bradford *et al.* (2021), exclusion of a spatial term on model selection, requiring more explicit reliance on environmental variables, and incorporating new approaches (Miller *et al.* 2022) for more comprehensively estimating uncertainty in model predictions that account for the combined uncertainty around all parameter estimates~~the overall period~~. The modeling framework incorporated Beaufort-specific trackline detection probabilities for Risso’s dolphins from Barlow *et al.* (2015). Models ~~for each period~~ were ~~then~~ used to predict density and abundance for each survey year based on the environmental conditions within that year (see Forney *et al.* 2015, Becker *et al.* 2016).~~ The modeling framework incorporated Beaufort-specific trackline detection probabilities for~~

Table 1. Model-based ~~L~~line-transect abundance estimates for Risso's dolphins in the ~~derived from surveys of the entire~~ Hawaiian Islands EEZ in 2002 and ~~,~~2010~~, and 2017~~ (Becker *et al.* 2021) and 2017 and 2020 (Becker *et al.* 2022), derived from NMFS surveys in the central Pacific since 2000. The Becker *et al.* (2022) analysis incorporates a more comprehensive model-based approach to estimating model uncertainty, such that the CVs and 95% confidence limits for 2002/2010 and 2017/2020 are not directly comparable.

Year	Model-based Abundance	CV	95% Confidence Limits
2020	6,979	0.29	3,649-13,348
2017	7,437 ~~7,385~~	0.30 ~~0.22~~	4,027-13,736 ~~4,817-11,322~~
2010	6,174	0.20	4,159-9,165
2002	6,916	0.21	4,623-10,346

~~Risso's dolphins from Barlow *et al.* (2015).~~ Bradford *et al.* (2021) produced design-based abundance estimates for Risso's dolphins in~~for~~ 2002, 2010, and 2017 ~~each survey years~~ that can be used as a point of comparison to the model-based estimates for those years. While on average, the estimates are broadly similar between the two approaches, the annual design-based estimates show much greater variability between years than do the model-based estimates (Figure 2). The model-based approach reduces variability through explicit examination of habitat relationships across the full dataset, while the design-based approach evaluates encounter data for each year separately and thus is more susceptible to the effects of encounter rate variation. Model based-estimates are based on the implicit assumption that changes in abundance are attributed to environmental variability alone. ~~There are insufficient data to e~~Explicitly incorporating~~e~~ a trend term into the model is not possible due to the insufficient sample size to test for temporal effects. Despite not fully accounting for inter-annual variation in total abundance, the model-based estimates are considered the best available estimate for each survey year. Becker *et al.* (2022) and Bradford *et al.* (2022) evaluated seasonal changes in the abundance of Risso's dolphins within the main Hawaiian Islands using summer-fall data from 2017 and winter survey data from 2020. Both analyses showed slightly higher densities of Risso's dolphins in the MHI in winter, with the spatially-explicit model showing marked differences in winter and non-winter distribution driven by the relationship with mixed layer depth for this species. Previously published abundance~~design-based~~ estimates for the Hawaiian Islands EEZ ~~from 2002 and 2010 surveys~~ (e.g. Barlow 2006, Becker *et al.* 2012, Forney *et al.* 2015, Bradford *et al.* 2017) used a subset of the dataset used by Becker *et al.* (2021, 2022) and Bradford *et al.* (2021) to derive line-transect parameters, such that these estimates have been superseded by the estimates presented here. The best estimate of abundance is based on the 2020~~17~~ survey, or 6,979 ~~7,385~~ (CV=0.29~~2~~) Risso's dolphins.

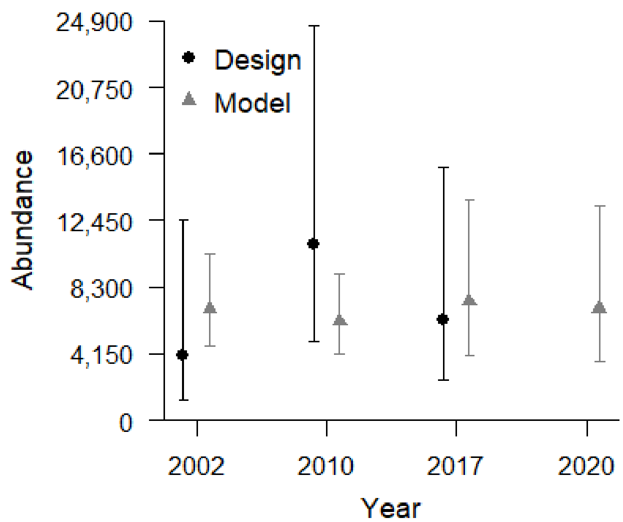

Figure 2. Comparison of design-based (black circles, Bradford *et al.* 2021) and model-based (gray triangles, Becker *et al.* 2021, 2022) estimates of abundance for Risso's dolphins for each survey year (2002, 2010, 2017, 2020).

Population estimates have been made off Japan (Miyashita 1993), in the eastern tropical Pacific (Wade and Gerrodette 1993), and off the U.S. West Coast (Barlow 2016), but it is not known whether these animals are part of the same population that occurs around the Hawaiian Islands and in the central North Pacific.

Minimum Population Estimate

The minimum population estimate~~size~~ is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2020~~2017~~ abundance estimate (from Becker *et al.* 2022), or 5,283~~6,150~~ Risso's dolphins within the Hawaiian Islands EEZ.

Current Population Trend

The model-based abundance estimates for Risso's dolphins provided by Becker *et al.* (2021, 2022) do not explicitly allow for examination of population trend other than that driven by environmental factors. Model-based examination of Risso's dolphin trends including sighting data beyond the Hawaiian Islands EEZ will be required to more fully examine trend for this stock.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Hawaii stock of Risso's dolphins is calculated as the minimum population size within the U.S. EEZ of the Hawaiian Islands (5,283~~6,150~~) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.5 (for a stock of unknown status with no known fishery mortality and serious injury within the Hawaiian Islands EEZ; Wade & Angliss 1997), resulting in a PBR of 53~~61~~ Risso's dolphins per year.

HUMAN CAUSED MORTALITY AND SERIOUS INJURY

Fishery Information

Information on fishery-related mortality and serious injury of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaiian fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. No interactions between nearshore fisheries and Risso's dolphins have been reported in Hawaiian waters. No estimates of human-caused mortality or serious injury are currently available for nearshore hook and line fisheries because these fisheries are not observed or monitored for protected species bycatch.

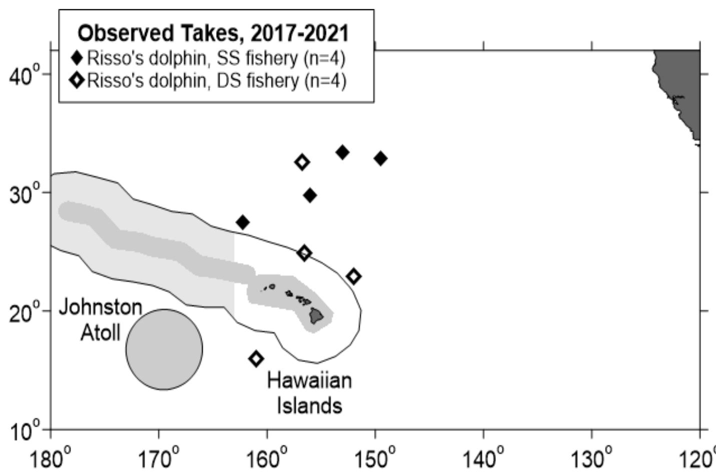

Figure 3. Locations of observed Risso's dolphin takes within the shallow-set fishery (filled diamonds) and deep-set fishery (open diamonds) ~~unidentified cetacean that maybe rough-toothed dolphins based on the observer's description (crosses)~~ in the Hawai'i-based longline fishery,~~2014-2018~~2017-2021. Solid lines represent the U.S. EEZ~~s~~. Gray shading notes areas closed to longline fishing.

There are currently two distinct longline fisheries based in Hawai'i: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas, but are prohibited from operating within the Papahānaumokuākea Marine National Monument (PMNM) and within the Longline Exclusion Zone around the main Hawaiian Islands and the Pacific Remote Islands and Atolls (PRIA) MNM around Johnston Atoll. The PMNM originally included the waters within a 50 nmi radius around the Northwestern Hawaiian Islands. ~~As of~~In August, 2016, the PMNM area was expanded to extend to the 200 nmi EEZ boundary west of 163° W. Between 2017 and 2021~~2014 and 2018~~, 4~~15~~ Risso's dolphins were observed hooked or entangled~~killed or seriously injured~~ in the SSLL fishery (100% observer coverage), ~~and the injury status of one could not be determined based on the observer's description,~~ and 4~~43~~ Risso's dolphins were observed taken~~killed or seriously injured~~ in the DSLL fishery (~~18~~15-21% observer coverage) (Figure 3, Bradford ~~2018a,~~ 2018~~b~~, 2020, 2021, In press, In review, ~~Bradford and Forney 2017,~~ McCracken & Cooper 2022b~~2019~~). One Risso's dolphin in the DSLL fishery was~~and four in the SSLL fishery were~~ killed, 4~~10~~ in the SSLL fishery and 2 in the DSLL fishery were considered to have been seriously injured, all outside the U.S. EEZ.~~, and the remaining interactions in the SSLL fishery could not be determined based on an evaluation of the observer's description of the interaction. When otherwise~~

~~undetermined, the injury status of takes is prorated to serious versus non-serious using the historic rate of serious injury within the observed takes.~~

The total estimated number of dead or seriously injured dolphins is calculated based on observer coverage rate, the location of the observed take (inside or outside of the EEZ), and the ratio of observed dead and seriously injured whales versus those judged to be not seriously injured. Observer coverage is measured on a per-trip basis throughout the calendar year as described by McCracken (2019). In years with large fluctuations in observer coverage, such as during the early days of the COVID-19 pandemic when observer coverage dropped to less than 10% during the second quarter of the year, the annual bycatch estimation process may be subset into several periods, as described in McCracken and Cooper (2022a). Average 5-yr estimates of annual mortality and serious injury for 2017-2021~~2014-2018~~ are 5.0 (CV=0.4)~~ 5.7 (CV=0.7)~~ Risso's dolphins outside of U.S. EEZ~~s~~, and 0 within the Hawaiian Islands EEZ (Table 2, McCracken and Cooper 2022b~~ 2019~~). ~~One additional unidentified cetacean, possibly a Risso's dolphin based on the observer's description, and three other unidentified delphinids were taken in the DSLL fishery, some of which may have been Risso's dolphins.~~

Table 2. Summary of available information on incidental mortality and serious injury (MSI) of Risso's dolphin (Hawaii stock) in commercial longline fisheries, within and outside of U.S. EEZ~~s~~ (McCracken and Cooper, 2022b~~2019~~). Mean annual takes are based on 2017-2021~~2014-2018~~ data unless indicated otherwise. Information on all observed takes (T) and ~~combined mortality events & serious injuries (~~MSI~~)~~ is included. Total takes were prorated to deaths, serious injuries, and non-serious injuries based on the observed proportions of each outcome.

Fishery Name	Year	Data Type	Percent Observer Coverage	~~Observed total interactions (T) and mortality events, and serious injuries (MSI), and total estimated mortality and serious injury (M&SI) of Risso's dolphins~~			
				Outside U.S. EEZ~~s~~		Hawaiian Islands EEZ	
				Observed~~.~~ T/MSI	Estimated M&SI (CV)	Observed~~.~~ T/MSI	Estimated M&SI (CV)
Hawaiʻi-based deep-set longline fishery	~~2014~~	Observer data	~~21%~~	~~0~~	~~0 (-)~~	~~0~~	~~0 (-)~~
	~~2015~~		~~21%~~	~~2/2~~	~~10 (0.6)~~	~~0~~	~~0 (-)~~
	~~2016~~		~~20%~~	~~0~~	~~0 (-)~~	~~0~~	~~0 (-)~~
	2017		20%	1/1	5 (0.9)	0	0 (-)
	2018		18%	0	0 (-)	0	0 (-)
	2019		21%	1/1	7 (0.9)	0	0 (-)
	2020		15%	2/1	16 (0.5)	0	0 (-)
	2021		18%	0/0	0 (-)	0	0 (-)
Mean Estimated Annual Take (CV) 2017-2021					**5.0 (0.4)~~2.9 (0.7)~~**		**0 (-)**
Hawaiʻi-based shallow-set longline fishery	~~2014~~	Observer data	~~100%~~	~~6/6‡~~	~~6~~	~~0~~	~~0~~
	~~2015~~		~~100%~~	~~3/3~~	~~3~~	~~0~~	~~0~~
	~~2016~~		~~100%~~	~~2/2~~	~~2~~	~~0~~	~~0~~
	2017		100%	2/2	2	0	0
	2018		100%	2/2	2	0	0
	2019		100%	0/0	0	0	0
	2020		100%	0/0	0	0	0
	2021		100%	0/0	0	0	0
Mean Annual Takes (100% coverage) 2017-2021					**0.8~~2.8~~**	0	**0**
Minimum total annual takes within U.S. EEZ (2017-2021)							**0 (-)**

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

STATUS OF STOCK

The Hawaiʻi stock of Risso's dolphins is not considered strategic under the 1994 amendments to the MMPA. The status of Risso's dolphins in Hawaiian waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Risso's dolphins are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. Given the absence of recent recorded fishery-related mortality or serious injuries within the ~~U.S.~~Hawaiian Islands EEZ, the total fishery mortality and serious injury can be considered to be insignificant and approaching zero. One Risso's dolphin stranded on the MHI tested positive for *Morbillivirus* (Jacob *et al*. 2016). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters, all identified as a unique strain of *morbillivirus*~~,~~ (Jacob *et al*. 2016), raises concerns about the history and prevalence of this disease in Hawai'i and the potential population impacts, including cumulative impacts of disease with other stressors.

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Revised ~~5/5/2021~~ 8/16/2023

COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus truncatus*): Hawaiian Islands Stock Complex – Kaua‘i/Ni‘ihau, O‘ahu, Maui Nui ~~4-Islands~~, Hawai‘i Island, and Hawai‘i Pelagic Stocks

STOCK DEFINITION AND GEOGRAPHIC RANGE

Common bottlenose dolphins are widely distributed throughout the world in tropical and warm-temperate waters (Perrin *et al.* 2009). Bottlenose dolphins are common throughout the Hawaiian Islands, both in nearshore waters as well as at great distances from shore (*e.g.* Baird *et al.* 2013, Bradford *et al.* 2021; Figure 1). ~~from the island of Hawaii to Kure Atoll (Shallenberger 1981, Baird *et al.* 2013). Summer/fall s~~Shipboard surveys of the ~~waters within the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands resulted in 18 sightings in 2002, 20 sightings in 2010, and 4 sightings in 2017 (Barlow 2006, Bradford *et al.* 2017, Yano *et al.* 2018; Figure 1). In the Hawaiian Islands, bottlenose dolphins are found in shallow inshore waters and deep water (Baird *et al.* 2009~~).

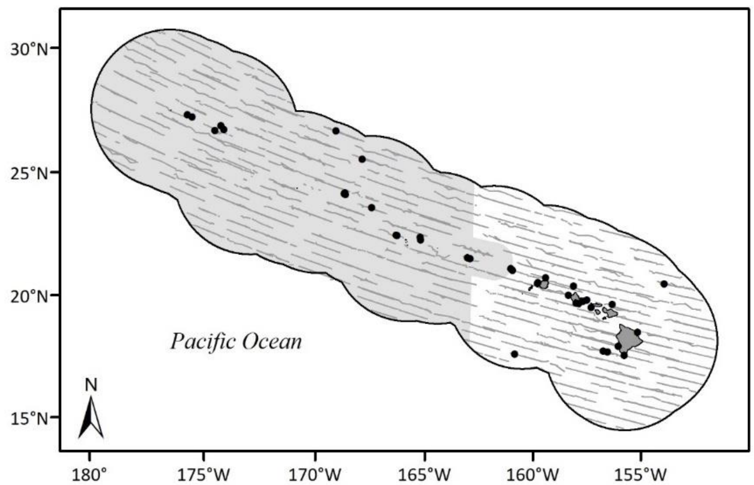

Figure 1. Bottlenose dolphin sighting locations (circles) and survey effort (gray lines) during the 2002 (Barlow 2006) ~~(diamond)~~, 2010 (Bradford *et al.* 2017)~~(circle)~~, and 2017 (Yano *et al.* 2018 ~~(square)~~ shipboard cetacean surveys of the U.S. EEZ ~~waters surrounding~~around the Hawaiian Islands ~~(Barlow 2006, Bradford *et al.* 2017, Yano *et al.* 2018~~outer black line). ~~Dark gray shading indicates~~ The ~~original~~ Papahānaumokuākea Marine National Monument in the western portion of the EEZ is shaded gray~~, with the lighter gray shading denoting the full 2016 Expansion area. Dotted line represents the 1000 m isobaths~~. Insular stock boundaries are shown in Figure 2.

Separate offshore and coastal forms of bottlenose dolphins have been identified along continental coasts (Ross and Cockcroft 1990; Van Waerebeek *et al.* 1990), and there is evidence that similar onshore-offshore forms may exist in Hawaiian waters (Baird 2016). In their analysis of sightings of bottlenose dolphins in the eastern tropical Pacific (ETP), Scott and Chivers (1990) noted a large hiatus between the westernmost sightings and the Hawaiian Islands. These data suggest that bottlenose dolphins in Hawaiian waters belong to a separate stock from those in the ETP. Furthermore, recent photo-identification and genetic studies of bottlenose dolphins sampled near each of the main Hawaiian Islands~~Oahu, Maui, Lanai, Kauai, Niihau, and Hawaii~~ suggest limited movement of bottlenose dolphins between islands and offshore waters (Baird *et al.* 2009; Martien *et al.* 2012, Harnish 2021). These data support~~suggest~~ the existence of demographically independent~~distinct~~ resident populations at each of ~~the~~ four main Hawaiian Island groupings – Kaua‘i and Ni‘ihau, ~~Oahu~~O‘ahu, the Maui Nui ~~‘4-Islands’~~ region (Moloka‘i, Lāna‘i, Maui, Kaho‘olawe), and Hawai‘i Island. Genetic data support inclusion of bottlenose dolphins in

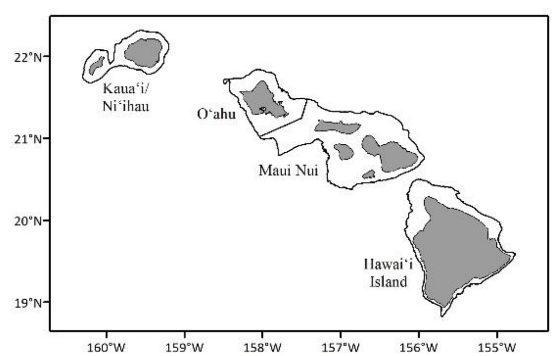

Figure 2. Main Hawaiian Islands insular bottlenose dolphin stock boundaries (gray lines). ~~Areas beyond the 1000 m isobath represent the pelagic stock range.~~

deeper waters surrounding the main Hawaiian Islands as part of the broadly distributed pelagic population (Martien *et al.* 2012). Over 99% of the bottlenose dolphins linked through photo-identification to one of the insular population around the main Hawaiian Islands (Baird *et al.* 2009) have been documented in waters of 1000 m or less (Martien and Baird 2009). Based on these data, Martien and Baird (2009) suggested that the boundaries between the insular stocks and the Hawai‘i Pelagic stock be placed along the 1000 m isobath. Since that isobath does not separate Oahu from the ~~4 Islands~~Maui Nui Region, the boundary between those stocks runs approximately equidistant between the 500 m isobaths around O‘ahu and the ~~4 Islands~~Maui Nui Region, through the middle of Kaiwi Channel. These boundaries (Figure 2) are applied in this report to recognize separate insular and pelagic bottlenose dolphin stocks for management (NMFS ~~2005~~2023a). These boundaries may be revised in the future as additional information becomes available. To date, no data are available regarding population structure of bottlenose dolphins in the Northwestern Hawaiian Islands (NWHI), though sightings during a shipboard survey in~~the~~ 2010~~ survey~~ indicate they are commonly found close to the islands and atolls there (Bradford *et al.* 2017). Given the evidence for island resident populations in the main Hawaiian Islands, the larger distances between islands in the NWHI, and the finding of population structure within the NWHI in other dolphin species (e.g., Andrews *et al.* 2010), it is likely that additional demographically independent populations of bottlenose dolphins exist in the NWHI. However, until data become available upon which to base stock designations in this area, bottlenose dolphins in the NWHI will remain part of the Hawai‘i Pelagic Stock.

For the Marine Mammal Protection Act (MMPA) Pacific stock assessment reports, bottlenose dolphins within the Pacific U.S. Exclusive Economic Zone (EEZ) are divided into seven stocks: 1) California, Oregon and Washington offshore stock, 2) California coastal stock, and five Pacific Islands Region management stocks (this report): 3) Kaua‘i/Niihau, 4) O‘ahu, 5) ~~4 Islands~~Maui Nui (Moloka‘i, Lāna‘i, Maui, Kaho‘olawe), 6) Hawai‘i Island and 7) the Hawai‘i~~an~~ Pelagic Stock, including animals found both within the U.S. EEZ around the Hawaiian Islands ~~EEZ~~ and in adjacent high seas waters. Because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of the Hawai‘i Pelagic stock is evaluated based on data from U.S. EEZ ~~waters~~ of the Hawaiian Islands (NMFS ~~2005~~2023a). Estimates of abundance, potential biological removals, and status determinations for the five Hawaiian stocks are presented separately below.

HUMAN CAUSED MORTALITY AND SERIOUS INJURY

Fishery Information

Information on fishery-related mortality of cetaceans in Hawaiian waters is limited, but the gear types used in Hawai‘i fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. There are at least two reports of entangled bottlenose dolphins dying in gillnets off Maui (Nitta and Henderson 1993, Maldini 2003, Bradford and Lyman 2013). Although gillnet fisheries are not observed or monitored through any State or Federal program, State regulations now ban gillnetting around Maui and much of O‘ahu and require gillnet fishermen to monitor their nets for bycatch every 30 minutes in those areas where gillnetting is permitted. In 2018, a bottlenose dolphin calf was observed with a gunshot wound through its melon, possibly as a result of a fisheries interaction (Harnish *et al.* 2019) ~~(Bradford and Lyman 2020)~~. Although the wound was initially judged to be serious, ~~ten~~ sightings of this animal since the injury was initially observed have indicated the wound is healing and the animal has survived (Harnish *et al.* 2019), such that the injury was ultimately determined to be non-serious (Bradford and Lyman 2020) under criteria for assessing serious injury in marine mammals (NMFS 2023b). In 2019, this same individual was observed hooked in the mouth and entangled around its pectoral fin by the trailing line, also initially judged to be a serious injury (Bradford and Lyman 2022). However, ~~B~~based on the ~~most recent~~ observations in 2021

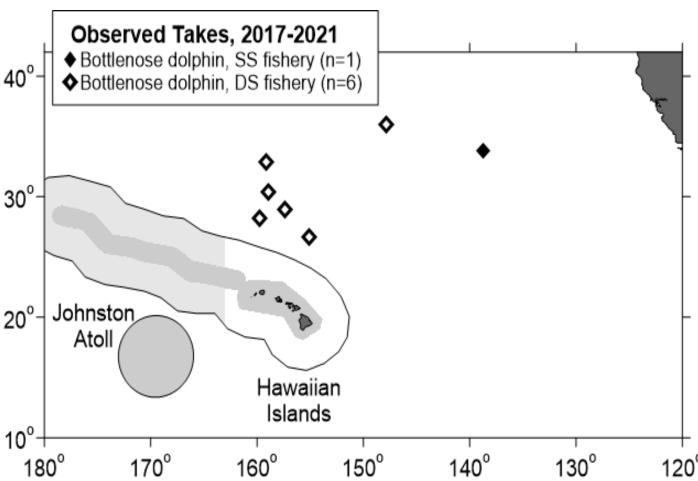

Figure 3. Locations of observed Pelagic Stock bottlenose dolphin takes within the shallow-set fishery (filled diamonds) and deep-set fishery (open diamonds)~~, and unidentified cetaceans considered to possibly or likely be bottlenose dolphins based on the observer's description (crosses)~~ in the Hawaii-based longline fishery, ~~2014-2018~~2017-2021. Solid lines represent the U. S. EEZ. Gray shading notes areas closed to ~~commercial fishing, with the PMNM Expansion area closed since August 2016~~longline fishing.

of the animal in good body condition, the injury is currently considered to be ~~not~~non-serious (Bradford and Lyman 2022) ~~under the most recently developed criteria for assessing serious injury in marine mammals (NMFS 2012~~). In 2020, an adult bottlenose dolphin was found dead as a result of an ingested circle hook piercing its esophagus, with the hook and attached monofilament line attributed to a nearshore fishery (Bradford and Lyman 2023). This recent mortality indicates that nearshore fisheries still pose a risk to bottlenose dolphins around the Hawaiian Islands. However, ~~N~~no estimates of human-caused mortality or serious injury are currently available for nearshore hook and line or gillnet fisheries because these fisheries are not observed or monitored for protected species bycatch.

Bottlenose dolphins are one of the species commonly reported to steal bait and catch from several Hawai‘i sport and commercial fisheries (Nitta and Henderson 1993, Schlais 1984). Observations of bottlenose dolphins stealing bait or catch have been made in the day handline fishery for tuna (palu-ahi), the night handline fishery for tuna (ika-shibi), the handline fishery for mackerel scad, the troll fishery for billfish and tuna, and the inshore set gillnet fishery (Nitta and Henderson 1993). Nitta and Henderson (1993) indicated that bottlenose dolphins remove bait and catch from handlines used to catch bottomfish off the island of Hawai‘i and Kaula Rock and formerly on several banks of the Northwestern Hawaiian Islands. Bottlenose dolphins were thought to interact with the bottomfish fishery in the NWHI (Kobayashi and Kawamoto 1995), though this fishery is no longer permitted for the NWHI. Fishermen around the main Hawaiian Islands claim interactions with dolphins that steal bait and catch are increasing, including anecdotal reports of bottlenose dolphins getting “snagged” (Rizzuto 2007). An assessment of the incidence of potential fishing gear-associated scarring on bottlenose dolphins near Maui Nui revealed 27% of non-calf well-marked individuals photographed between 1996 and 2020 had one or more scars that may be attributed to fishing gear (Machernis *et al.* 2021). ~~Bottlenose dolphins were thought to interact with the bottomfish fishery in the NWHI (Kobayashi & Kawamoto 1995), though Interaction rates between dolphins and the NWHI bottomfish fishery were estimated based on studies conducted in 1990-1993, indicating that an average of 2.67 dolphin interactions, defined as incidence of dolphins removing bait or catch from hooks, occurred for every 1000 fish brought on board (Kobayashi and Kawamoto 1995). These interactions generally involved bottlenose dolphins, and it is not known whether these interactions result~~ed ~~in serious injury or mortality of dolphins. This fishery was observed from 2003 through 2005 at 18-25% coverage, during which time no incidental takes of cetaceans were reported. The bottomfish~~ this ~~fishery is no longer permitted for the NWHINorthwestern Hawaiian Islands.~~

There are currently two distinct longline fisheries based in Hawai‘i: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas, but are prohibited from operating within the Papahānaumokuākea Marine National Monument (PMNM) and within the Longline Exclusion Zone around the main Hawaiian Islands and the Pacific Remote Islands and Atolls (PRIA) MNM around Johnston Atoll. The PMNM originally included the waters within a 50 nmi radius around the NWHI. ~~As of~~In August, 2016, the PMNM area was expanded to extend to the 200 nmi EEZ boundary west of 163˚~~o~~ W. Between 2017 and 2021~~2014 and 2018~~, one~~eight~~ bottlenose dolphin~~s~~ was~~were~~ observed hooked or entangled in the SSLL fishery (100% observer coverage), and six~~three~~ bottlenose dolphins were observed taken in the DSLL fishery (~~18-22~~15-21% observer coverage)~~)~~ within the Hawaiian Islands EEZ or adjacent high-seas waters ~~(~~ (Bradford 2018, 2020, 2021, 2023, in review)~~(Bradford 2018a, 2018b, 2020, Bradford and Forney 2017, McCracken 2019)~~. Based on the observed take locations (Figure 3~~2~~), these takes are all considered to have been from the Pelagic Stock of bottlenose dolphins. All 7~~11~~ dolphins were considered to have been seriously injured ~~(Bradford 2018a, 2018b, 2020, in press, in review)Bradford and Forney 2017)~~, based on an evaluation of the observer’s description of the interaction and following the most recently developed criteria for assessing serious injury in marine mammals (NMFS ~~2012~~ 2023b).

The total estimated number of dead or seriously injured dolphins is calculated based on observer coverage rate, the location of the observed take (inside or outside of the U.S. EEZ~~s~~), and the ratio of observed dead and seriously injured dolphins versus those judged to be not seriously injured. Observer coverage is measured on a per-trip basis throughout the calendar year as described by McCracken (2019). In years with large fluctuations in observer coverage, such as during the early days of the COVID-19 pandemic when observer coverage dropped to less than 10% during the second quarter of the year, the annual bycatch estimation process may be subset into several periods, as described in McCracken and Cooper (2022a). Average 5-yr estimates of annual mortality and serious injury for the Pelagic Stock during 2017-2021~~2014-2018~~ are 6.6~~3.0~~ (CV=0.4~~0.6~~) bottlenose dolphins outside of the Hawaiian Islands~~U.S.~~ EEZ~~s~~, and 0 within the Hawaiian Islands EEZ (Table 1, McCracken and Cooper 2022b~~2019~~). One~~Two~~ unidentified cetacean~~s~~, ~~considered~~ likely to be a bottlenose dolphin~~s~~ based on the observer’s description, was~~were~~ taken in the DSLL fishery in 2017 (Bradford 2018),~~and three unidentified cetaceans was taken in the DSLL fishery, some of which may have been bottlenose dolphins.~~

Table 1. Summary of available information on incidental mortality and serious injury (MSI) of bottlenose dolphins

(Hawaiʻi Pelagic stock) in commercial longline fisheries, within and outside of the U.S. EEZ~~s~~ (McCracken and Cooper 2022b ~~2019~~). Mean annual takes are based on 2017-2021~~2014-2018~~ data unless otherwise indicated. Information on all observed takes (T) and ~~combined mortality events & serious injuries (~~MSI~~)~~ is included along with MSI estimates. Total takes were prorated to deaths, serious injuries, and non-serious injuries based on the observed proportions of each outcome.

Fishery Name	Year	Data Type	Percent Observer Coverage	~~Observed total interactions (T) and mortality events, and serious injuries (MSI), and total estimated mortality and serious injury (M&SI) of Hawaii Pelagic stock bottlenose dolphins~~			
				Outside U.S. EEZ~~s~~		Hawaiian Islands EEZ	
				Observed~~.~~ T/MSI	Estimated M&SI (CV)	Observed~~.~~ T/MSI	Estimated M&SI (CV)
Hawaiʻi-based deep-set longline fishery	~~2014~~	Observer data	~~21%~~	~~0~~	~~0 (-)~~	~~0~~	~~0 (-)~~
	~~2015~~		~~21%~~	~~0~~	~~0 (-)~~	~~0~~	~~0 (-)~~
	~~2016~~		~~20%~~	~~1/1~~	~~5 (0.9)~~	~~0~~	~~0 (-)~~
	2017		20%	1/1	6~~7~~ (0.9)	0	0 (-)
	2018		18%	1/1	3 (0.9)	0	0 (-)
	2019		21%	0	0 (-)	0	0 (-)
	2020		15%	1/1	10 (0.6)	0	0 (-)
	2021		18%	3/3	9 (0.6)	0	0 (-)
Mean Estimated Annual Take (CV) 2017-2021					**5.6 (0.4)~~3.0 (0.6)~~**	0	**0 (-)**
Hawaiʻi-based shallow-set longline fishery	~~2014~~	Observer data	~~100%~~	~~4/4~~	~~4~~	~~0~~	~~0~~
	~~2015~~		~~100%~~	~~2/2~~	~~2~~	~~0~~	~~0~~
	~~2016~~		~~100%~~	~~1/1~~	~~1~~	~~0~~	~~0~~
	2017		100%	0	0	0	0
	2018		100%	1/1	1	0	0
	2019		100%	0	0	0	0
	2020		100%	0	0	0	0
	2021		100%	0	0	0	0
Mean Annual Takes (100% coverage) 2017-2021					**1~~2~~**		**0**
Minimum total annual takes within U.S. EEZ (2017-2021)							**0 (-)**

KAUAʻI / NIʻIHAU STOCK

POPULATION SIZE

Photographic data from multiple contributors spanning 2000 to 2018 were used to assess the annual abundance of each main Hawaiian Islands insular population of bottlenose dolphins using a POPAN model stratified within each stock area based on spatial gaps in sightings and significant bathymetric or geographic features ~~or spatial variability in encounters through time~~ (Van Cise *et al.* 2021). Annual abundance estimates for the Kauaʻi/Niihau stock of bottlenose dolphins were produced for 2003 through 2007 and 2011 through 2018. The 2018 abundance estimate for the Kauaʻi/Niʻihau stock was 112 (CV=0.24) bottlenose dolphins.

~~A photo-identification study conducted from 2003 to 2005 identified 102 individual bottlenose dolphins around Kauai and Niihau (Baird *et al.* 2009). A Lincoln-Peterson mark-recapture analysis of the photo-identification data resulted in an abundance estimate of 147 (CV=0.11), or 184 animals when corrected for the proportion of marked individuals (Baird *et al.* 2009). The CV of this estimate is likely negatively-biased, as it does not account for variation in the proportion of marked animals within groups. There is no current abundance estimate for this stock.~~

Minimum Population Estimate

The minimum population estimate for the Kauaʻi/Niʻihau stock of bottlenose dolphins is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2018 abundance estimate (from Van Cise *et al.* 2021), or 92 bottlenose dolphins.~~ler whales.the number of distinctive individuals identified during 2012 to 2015 photo-identification studies, or 97 dolphins (Baird *et al.* 2017). The data used in the 2003-2005 mark-recapture estimate (Baird *et al.* 2009) are considered outdated, and therefore are not suitable for deriving a minimum abundance estimate.~~

Current Population Trend

Annual abundance estimates derived in Van Cise *et al.* (2021) ~~may~~ suggest that the Kauaʻi/Niʻihau stock of bottlenose dolphins ~~has~~may have declined over the nearly 20-year period of the study, with a high of 193 (CV=0.25) dolphins in 2003 to the low of 112 (CV=0.24) in 2018, representing an overall average annual decline of 2.6% (95% CI -6.9 to -1.7). However, the annual estimates did not differ significantly throughout the study period and varied only by a few individuals between 2011 and 2018, such that the trends are not considered reliable (Van Cise *et al.* 2021). Further, while survey effort was most consistent for the Kauaʻi/Niʻihau stock, sampling variability was not fully accounted for in the estimates of abundance and trend. ~~Only one abundance estimate is available for this stock, such that there is insufficient information to assess population trends~~.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (92~~97~~) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a stock of unknown status with no reported fishery mortality or serious injury within the Kauaʻi/Niʻihau stock range; Wade and Angliss 1997), resulting in a PBR of 0.9~~1.0~~ bottlenose dolphins per year.

STATUS OF STOCK

The Kauaʻi/Niʻihau Stock of bottlenose dolphins is not considered strategic under the 1994 amendments to the MMPA. The status of bottlenose dolphins in the Kauaʻi/Niʻihau stock relative to OSP is unknown. Although recent analyses suggest this stock may be declining (Van Cise *et al.* 2021), sampling limitations increase uncertainty around this conclusion.~~, and there are insufficient data to evaluate abundance trends.~~ Bottlenose dolphins are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. There have been no reports of recent mortality or serious injuries; however, there is no systematic monitoring for interactions with protected species within nearshore fisheries that may take this species, thus mean annual takes are undetermined. Insufficient information is available to determine whether the total fishery mortality and serious injury for Kauaʻi/Niʻihau bottlenose dolphins is insignificant and approaching zero mortality and serious injury rate. One stranded bottlenose dolphin from the Kauaʻi/Niʻihau stock tested positive for *Morbillivirus* (Jacob *et al.* 2016). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters (Jacob *et al.* 2016), raises concerns about the history and prevalence of this disease in Hawaiʻi and the potential population impacts, including the cumulative impacts of disease with other stressors.

OʻAHU STOCK

POPULATION SIZE

Photographic data from multiple contributors spanning 2000 to 2018 was used to assess the annual abundance of each main Hawaiian Islands insular population of bottlenose dolphins using a POPAN model stratified within each stock area based on spatial gaps in sightings and significant bathymetric or geographic features (Van Cise *et al.* 2021). Annual abundance estimates for the Oʻahu stock of bottlenose dolphins were produced for 2002 through 2018, except for 2005. The 2018 abundance estimate for the Oʻahu stock was 112 (CV=0.17) bottlenose dolphins.

~~A photo-identification study conducted in 2002, 2003 and 2006 identified 67 individual bottlenose dolphins around Oahu (Baird *et al.* 2009). A Lincoln-Peterson mark-recapture analysis of the photo-identification data resulted in an abundance estimate of 594 (CV=0.54), or 743 animals when corrected for the proportion of marked individuals (Baird *et al.* 2009). The estimate does not include individuals from the Northeastern (windward) side of the island. There is no current abundance estimate for this stock~~.

Minimum Population Estimate

The minimum population estimate for the Oʻahu stock of bottlenose dolphins is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2018 abundance estimate (from Van Cise *et al.* 2021), or 97 bottlenose dolphins.

~~There is no current minimum population estimate for the Oahu stock of bottlenose dolphins. The data used in the 2002-2006 mark-recapture estimate (Baird *et al.* 2009) are considered outdated, and therefore are not suitable for deriving a minimum abundance estimate, and the number of distinctive individuals identified during 2009 to 2012~~

~~photo-identification studies (Baird *et al.* 2017) is derived from insufficient survey effort to be considered a reasonable estimate of minimum population size.~~

Current Population Trend

Annual abundance estimates derived in Van Cise *et al.* (2021) ~~may~~ suggest that the O'ahu stock of bottlenose dolphins ~~has~~may have declined over the nearly 20-year period of the study, with a high of 193 (CV=0.31) dolphins in 2002 to the low of 112 (CV=0.17) in 2018, representing an overall average annual decline of 3% (95% CI -10.3 to +2.7). However, the annual estimates did not differ significantly throughout the study period and varied by only a few individuals over the last half of the study period, such that the trends are not considered reliable (Van Cise *et al.* 2021). Similar to other stocks, sampling variability was not fully accounted for in the estimates of abundance and trend, but particularly for the O'ahu stock, it is possible that the apparent decline is an artifact of increased citizen science contributions in one subarea and contracted survey effort over the study period.~~Only one abundance estimate is available for this stock, such that there is insufficient information to assess population trends.~~

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (97) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a stock of unknown status with no reported fishery mortality in the stock range (Wade and Angliss 1997), resulting in a PBR of 1.0 bottlenose dolphins per year. ~~Because there is no minimum population size estimate for this stock, the PBR is undetermined.~~

STATUS OF STOCK

The O'ahu stock of bottlenose dolphins is not considered strategic under the 1994 amendments to the MMPA. The status of bottlenose dolphins in O'ahu waters relative to OSP is unknown. Although recent analyses suggest this stock may be declining (Van Cise *et al.* 2021), sampling limitations increase uncertainty around this conclusion.~~, and there are insufficient data to evaluate abundance trends.~~ Bottlenose dolphins are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. There have been no reports of recent mortality or serious injuries, though there is evidence of a bottlenose dolphin that was shot in the head off O'ahu (Harnish *et al.* 2019) that later became hooked and entangled in fishing gear (Bradford and Lyman 2022). ~~; however, t~~There is no systematic monitoring for interactions with protected species within nearshore fisheries that may take this species, thus mean annual takes are undetermined. Insufficient information is available to determine whether the total fishery mortality and serious injury for O'ahu bottlenose dolphins is insignificant and approaching zero mortality and serious injury rate. *Morbilivirus* has been detected within other insular stocks of bottlenose dolphins in Hawai'i (Jacob *et al.* 2016). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters raises concerns about the history and prevalence of this disease in Hawai'i and the potential population impacts, including the cumulative impacts of disease with other stressors.

~~4-ISLANDS~~MAUI NUI STOCK

POPULATION SIZE

Photographic data from multiple contributors spanning 2000 to 2018 was used to assess the annual abundance of each main Hawaiian Islands insular population of bottlenose dolphins using a POPAN model stratified within each stock area based on spatial gaps in sightings and significant bathymetric or geographic features (Van Cise *et al.* 2021). Annual abundance estimates for the ~~4-Islands~~Maui Nui stock of bottlenose dolphins were produced for all years except 2008. The 2018 abundance estimate for the ~~4-Island~~Maui Nui stock was 64 (CV=0.15) bottlenose dolphins.

~~A photo-identification study conducted from 2000-2006 identified 98 individual bottlenose dolphins around Maui and Lanai (Baird *et al.* 2009). A Lincoln-Peterson mark-recapture analysis of the photo-identification data resulted in an abundance estimate of 153 (CV=0.24), or 191 animals when corrected for the proportion of marked individuals (Baird *et al.* 2009). This abundance estimate likely underestimates the total number of bottlenose dolphins in the 4-islands region because it does not include individuals from the Northeastern (windward) sides of Maui and Molokai. The CV of this estimate is likely negatively-biased, as it does not account for variation in the proportion of marked animals within groups. There is no current abundance estimate for this stock.~~

Minimum Population Estimate

The minimum population estimate for the ~~4-Islands~~Maui Nui stock of bottlenose dolphins is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2018 abundance estimate (from Van Cise *et al.* 2021), or 56 bottlenose dolphins.

~~There is no current minimum population estimate for the 4-Islands stock of bottlenose dolphins. The data used in the 2000-2006 mark-recapture estimate (Baird *et al.* 2009) are considered outdated, and therefore are not suitable for deriving a minimum abundance estimate, and the number of distinctive individuals identified during 2009 to 2012 photo-identification studies (Baird *et al.* 2017) is derived from insufficient survey effort to be considered a reasonable estimate of minimum population size.~~

Current Population Trend

Annual abundance estimates derived in Van Cise *et al.* (2021) suggest that the ~~4-Islands~~Maui Nui stock of bottlenose dolphins has declined over the nearly 20-year period of the study, with a high of 288 (CV=0.17) dolphins in 2000 to the low of 64 (CV=0.15) in 2018, representing an overall average annual decline of 8.6% (95% CI -13 to -6). While the analysis suggests a statistically significant decline in this stock (Van Cise *et al*. 2021), similar to other stocks, sampling variability was not fully accounted for in the estimates of abundance and trend. Particularly for the ~~4-Islands~~Maui Nui stock, it is possible that the apparent decline is an artifact of increased citizen science contributions in one subarea and contracted survey effort over the study period.~~Only one abundance estimate is available for this stock, such that there is insufficient information to assess population trends.~~

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (56) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a stock of unknown status with no reported fishery mortality in the ~~4-Islands~~Maui Nui stock area (Wade and Angliss 1997), resulting in a PBR of 0.6 bottlenose dolphins per year. ~~Because there is no minimum population size estimate for this stock, the PBR is undetermined.~~

STATUS OF STOCK

The ~~4-Islands~~Maui Nui Region Stock of bottlenose dolphins is not considered strategic under the 1994 amendments to the MMPA. The status of bottlenose dolphins in ~~4-Islands~~Maui Nui waters relative to OSP is unknown. Although recent analyses suggest this stock may be declining (Van Cise *et al.* 2021), sampling limitations increase uncertainty around this conclusion.~~, and there are insufficient data to evaluate trends in abundance.~~ Bottlenose dolphins are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor as "depleted" under the MMPA. There have been no reports of recent mortality or serious injuries of this stock; however, there is no systematic monitoring for interactions with protected species within nearshore fisheries that may take this species, thus mean annual takes are undetermined. Insufficient information is available to determine whether the total fishery mortality and serious injury for ~~4-Islands~~Maui Nui bottlenose dolphins is insignificant and approaching zero mortality and serious injury rate. *Morbilivirus* has been detected within other insular stocks of bottlenose dolphins in ~~Hawaii~~Hawai'i (Jacob *et al.* 2016). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters raises concerns about the history and prevalence of this disease in ~~Hawaii~~Hawai'i and the potential population impacts, including the cumulative impacts of disease with other stressors.

HAWAI'I ISLAND STOCK

POPULATION SIZE

Photographic data from multiple contributors spanning 2000 to 2018 was used to assess the annual abundance of each main Hawaiian Islands insular population of bottlenose dolphins using a POPAN model stratified within each stock area based on spatial gaps in sightings and significant bathymetric or geographic features (Van Cise *et al.* 2021). Annual abundance estimates for the Hawai'i Island stock of bottlenose dolphins were produced for all years from 2002 to 2018. The 2018 abundance estimate for the Hawai'i Island stock was 136 (CV=0.43) bottlenose dolphins.

~~A photo-identification study conducted from 2000-2006 identified 69 individual bottlenose dolphins around the island of Hawaii (Baird *et al.* 2009). A Lincoln-Peterson mark-recapture analysis of the photo-identification data~~

~~resulted in an abundance estimate of 102 (CV=0.13), or 128 animals when corrected for the proportion of marked individuals (Baird *et al.* 2009). This abundance estimate likely underestimates the total number of bottlenose dolphins around the island of Hawaii because it does not include individuals from the Northeastern (windward) side of the island. The CV of this estimate is likely negatively-biased, as it does not account for variation in the proportion of marked animals within groups. There is no current abundance estimate for this stock.~~

Minimum Population Estimate

The minimum population estimate for the Hawaiʻi Island stock of bottlenose dolphins is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2018 abundance estimate (from Van Cise *et al.* 2021), or 96 bottlenose dolphins.

~~The minimum population estimate for the Hawaii Island bottlenose dolphins is the number of distinctive individuals identified during 2010 to 2013 photo-identification studies, or 91 dolphins (Baird *et al.* 2017). The data used in the 2000-2006 mark-recapture estimates (Baird *et al.* 2009) are considered outdated, and therefore are not suitable for deriving a minimum abundance estimate.~~

Current Population Trend

Annual abundance estimates derived in Van Cise et al. (2021) suggest that the Hawaiʻi Island stock of bottlenose dolphins has increased over the nearly 20-year period of the study, with a low of 10 (CV=0.17) in 2000 to the high of 136 (CV=0.43) in 2018, representing an overall average annual increase of 10.5% (95% CI 0.94 to 15.31). This estimated annual growth rate is greater than the species maximum expected growth rate of 4% and was driven largely by influxes of new individuals during the study period. Similar to other stocks, sampling variability was not fully accounted for in the estimates of abundance and trend, but particularly for the Hawaiʻi Island stock, the abundance estimates likely underestimate true stock size because sampling for this stock was entirely on the leeward side of Hawaiʻi Island (Van Cias *et al.* 2021). Thus, the increasing trend may be an artifact of variability in sampling and individual habitat use.~~Only one abundance estimate is available for this stock, such that there is insufficient information to assess population trends.~~

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (96~~91~~) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a stock of unknown status with no reported fishery mortality in the Hawaiʻi Islands stock area (Wade and Angliss 1997), resulting in a PBR of 1.0~~0.9~~ bottlenose dolphins per year.

STATUS OF STOCK

The Hawaiʻi Island stock of bottlenose dolphins is not considered strategic under the 1994 amendments to the MMPA. The status of bottlenose dolphins in waters around ~~Hawaii~~Hawaiʻi Island relative to OSP is unknown. Although recent analyses suggest this stock may be increasing (Van Cise *et al.* 2021), sampling limitations increase uncertainty around this conclusion. ~~, and there are insufficient data to evaluate trends in abundance.~~ ~~Hawaii Island bottlenose dolphins are regularly seen near aquaculture pens off the Kona coast, and aquaculture workers have been observed feeding bottlenose dolphins. Bottlenose dolphins in this region are also known to interact with divers.~~ Bottlenose dolphins are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. ~~In the past 5 years, one animal was partially disentangled by a diver, but with hook and line remaining in its mouth was considered a serious injury.~~ In the past 5 years, one bottlenose dolphin was found dead on Hawaiʻi Island as a result of an ingested circle hook piercing its esophagus (Bradford and Lyman 2023). There is no systematic monitoring of takes in nearshore fisheries that may take this species, thus the single observed mortality ~~serious injury~~ may be an underestimate of the total fishery mortality for this stock. Total fishery mortality and serious injury for Hawaiʻi Island bottlenose dolphins is not approaching zero mortality and serious injury rate. ~~Hawaii~~Hawai'i Island bottlenose dolphins are regularly seen near aquaculture pens off the Kona coast, and aquaculture workers have been observed feeding bottlenose dolphins. Bottlenose dolphins in this region are also known to interact with divers. Since 2007, about one quarter (36) of Hawaiʻi Islands bottlenose dolphins have been observed associated with a pelagic mariculture operation for kanpachi off the Kona coast of Hawaiʻi Island, with 22 of those individuals seen at the farm on more than one occasion (Harnish *et al.* In press). Farm-associated dolphins are weakly linked to the rest of the Hawaiʻi Island population, and are seen in smaller groups near the farm than those groups seen away from the farm, factors that have been linked to lower survival in other populations (Stanton and Mann, 2012). *Morbillivirus* has been detected within other insular stocks of bottlenose dolphins in Hawai‘i (Jacob *et al.* 2016). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters raises concerns about the history and prevalence of this disease in Hawai‘i and the potential population impacts, including the cumulative impacts of disease with other stressors.

HAWAI‘I PELAGIC STOCK

POPULATION SIZE

Encounter data from shipboard line-transect surveys of the ~~entire~~ Hawaiian Islands EEZ were recently reevaluated for each survey year, resulting in the following~~updated~~ abundance estimates of bottlenose dolphins in the entirety of the Hawaiian Islands EEZ (Becker *et al.* 2022, Bradford *et al.* 2021; Table 2).

Table 2. Line-transect abundance estimates for bottlenose dolphins in the ~~derived from surveys of the entire~~ Hawaiian Islands EEZ in 2002, 2010, ~~and~~ 2017, and 2020, derived from NMFS surveys in the central Pacific since 1986~~1997~~ (Becker *et al.* 2022, Bradford *et al.* 2021).

Year	Design-based Abundance	CV	95% Confidence Limits	Model-based Abundance	CV	95% Confidence Limits
2020	-	-	-	24,669	0.57	8,774-69,361
2017	- ~~NA~~	-	-	25,857	0.56	9,356-71,464
2010	25,188	0.58	8,791-72,168	-	-	-
2002	9,678	0.49	3,924-23,868	-	-	-

Sighting data from 2002 to 2020 within the Hawaiian Islands EEZ were used to derive stock-specific habitat-based models of animal density for the 2017 to 2020 period. The models were then used to predict density and abundance for each survey year based on the environmental conditions within that year (see Forney *et al.* 2015, Becker *et al.* 2016). The modeling framework incorporated Beaufort-specific trackline detection probabilities for bottlenose dolphins from Barlow *et al.* (2015). Although model-based estimates were previously derived for years 2002, 2010, and 2017 (Becker *et al.* 2021), those estimates were not specific to the Hawai‘i Pelagic stock and as such may have reflected both the habitat associations and abundance of the insular stocks within the main Hawaiian Islands. Stock-specific model-based estimates were derived only for the most recent years (2017-2020), such that direct comparison of model and design-based estimates for the full survey time series is not possible at this time. Bradford *et al.* (2021) produced design-based abundance estimates for bottlenose dolphins for each full EEZ survey year with bottlenose dolphin encounters, with the 2010 design-based and 2017 and 2020 model-based estimates largely similar in the mean estimate and confidence limits (Figure 4). Current model based-estimates are based on the implicit assumption that annual changes in abundance are attributed to environmental variability alone. ~~There are insufficient data to e~~Explicitly incorporatinge a trend term into the model is not possible due to the insufficient sample size to test for temporal effects. Despite not fully accounting for inter-annual variation in total abundance, the model-based estimates are considered the best available estimate for the most recent survey year. Previously published design-based estimates for the Hawaiian Islands EEZ from 2002 and 2010 surveys (e.g. Barlow 2006, Bradford *et al.* 2017) used a subset of the dataset used by Becker *et al.* (~~2021,~~ 2022) and Bradford *et al.* (2021)

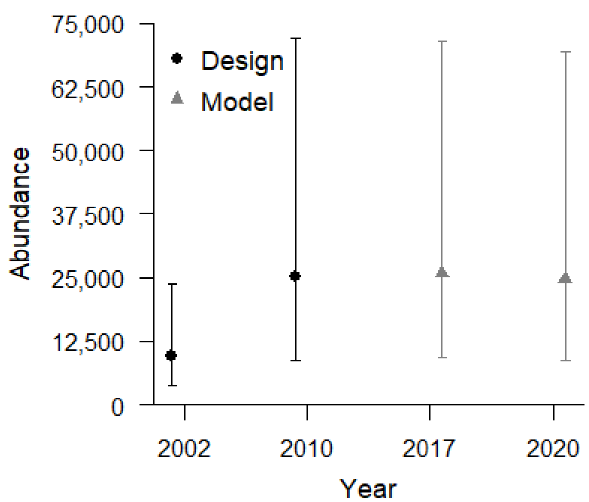

Figure 4. Comparison of design-based (black circles, Bradford *et al.* 2021) and model-based (gray triangles, Becker *et al.* 2022) estimates of abundance for Hawai‘i pelagic bottlenose dolphins for each survey year (2002, 2010, 2017, 2020).

to derive line-transect parameters, such that these estimates have been superseded by the estimates presented here. The best estimate of abundance is based on the 2020 survey, or 24,669 (CV=0.57) bottlenose dolphins.
~~The updated design-based abundance estimates use sighting data from throughout the central Pacific to estimate the detection function and use Beaufort sea-state-specific trackline detection probabilities for bottlenose dolphins from Barlow *et al.* (2015). Although previous estimates from the Hawaii EEZ have been published using subsets of this data, Bradford *et al.* (2021) uses a consistent approach for estimating all abundance parameters and as such are considered the best available estimates for each survey year. There were no sightings of bottlenose dolphins during systematic survey effort in 2017 and therefore design-based estimates are not available for that survey year. Model-based abundance estimates are available for all survey years (Becker et al. 2021), but are derived from sightings representing all bottlenose dolphins stocks within the Hawaiian islands, as removal of sightings of island-associated stock individuals would leave insufficient sample size to derive a robust model. Model covariates may not accurately reflect the habitat associations of pelagic bottlenose dolphins given the large number of insular sightings used in model development. Because the model is not stock-specific and pelagic stock abundance cannot be reliably extracted from model outputs, the design-based estimates are considered the best available for the pelagic stock.~~

Minimum Population Estimate

The minimum population estimate for the Hawai‘i P~~p~~elagic stock of bottlenose dolphins is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2020 abundance estimate (from Becker *et al.* 2022), or 15,783 bottlenose dolphins.~~There is no current minimum population estimate for the Hawaii pelagic stock of bottlenose dolphins. The 2010 estimate is considered outdated, and therefore are not suitable for deriving a minimum abundance estimate.~~

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size within the U.S EEZ of the Hawaiian Islands (15,783) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.5 (for a stock of unknown status with a Hawaiian Islands EEZ fishery mortality and serious injury rate CV of 0; Wade and Angliss 1997), resulting in a PBR of 158 bottlenose dolphins per year. ~~Because there is no minimum population size estimate for this stock, the PBR is undetermined.~~

STATUS OF STOCK

The Hawai‘i Pelagic Stock of bottlenose dolphins is not considered strategic under the 1994 amendments to the MMPA. The status of bottlenose dolphins in Hawaiian waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. It is not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. ~~Although the PBR for this stock is undetermined, t~~The estimated rate of fisheries related mortality or serious injury within the Hawaiian Islands EEZ is zero, such that ~~. T~~the total fishery mortality and serious injury for Hawai‘i Pelagic bottlenose dolphins is insignificant and approaching zero mortality and serious injury rate. *Morbilivirus* has been detected within ~~other~~ insular stocks of bottlenose dolphins in Hawai‘i (Jacob *et al.* 2016). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters raises concerns about the history and prevalence of this disease in Hawai‘i and the potential population impacts, including the cumulative impacts of disease with other stressors.

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Revised ~~5/5/2021~~ 08/16/2023

PANTROPICAL SPOTTED DOLPHIN (*Stenella attenuata attenuata*): Hawaiian Islands Stock Complex – Oʻahu, Maui Nui~~4 Islands~~, Hawaiʻi Island, and Hawaiʻi Pelagic Stocks

STOCK DEFINITION AND GEOGRAPHIC RANGE

Pantropical spotted dolphins are primarily found in tropical and subtropical waters worldwide (Perrin *et al.* 2009). Much of what is known about the species in the North Pacific has been learned from specimens obtained in the large directed fishery in Japan and in the eastern tropical Pacific (ETP) tuna purse-seine fishery (Perrin *et al*. 2009). Spotted dolphins are common and abundant throughout the Hawaiian ~~archipelago~~Islands, including nearshore where they are the second most frequently sighted species during nearshore surveys (Baird *et al*. 2013)~~,~~ and offshore where they are frequently observed during periodic ~~. Summer/fall~~ shipboard surveys of the waters within the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands ~~resulted in 14 sightings in 2002, 49 sightings in 2010, and 25 sightings in 2017 (Barlow 2006, Bradford *et al*. 2017, Yano *et al*. 2018;~~ (Figure 1). Morphological differences and distribution patterns indicate that the spotted dolphins around the Hawaiian Islands belong to a stock that is distinct from those in the ETP (Perrin 1975; Dizon *et al.* 1994; Perrin *et al*. 1994b).

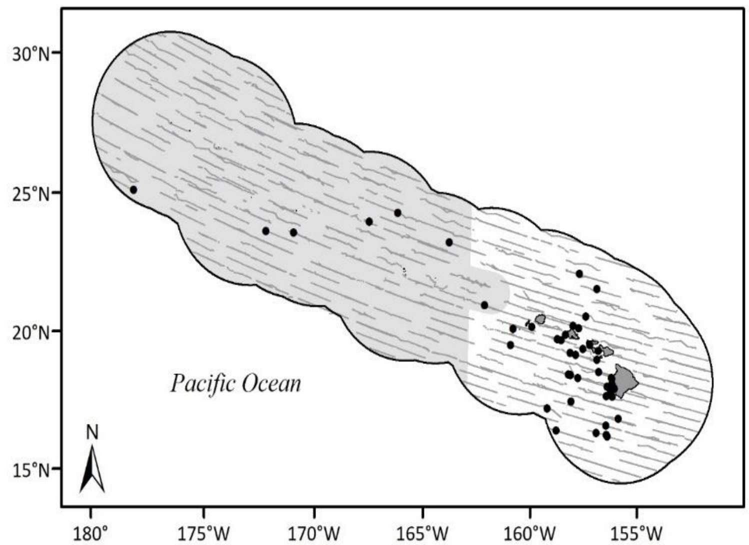

Figure 1. Pantropical spotted dolphin sighting locations (circles) and survey effort (~~light~~ gray lines) during the 2002 (~~diamonds~~Barlow 2006), 2010 (~~circle~~Bradford *et al.* 2017), and 2017 (~~square~~Yano *et al.* 2018) shipboard surveys of the U.S. EEZ ~~waters surrounding~~around the Hawaiian Islands (~~Barlow 2006, Bradford *et al*. 2017, Yano et al 2018~~outer black line). ~~Outer line represents approximate boundary of survey area and U.S. EEZ.~~ The Papahānaumokuākea Marine National Monument in the western portion of the EEZ is shaded gray~~, with the lighter gray shading denoting the full 2016 Expansion area~~. ~~Dotted line represents the 1000 m isobath.~~ Insular stock boundaries are shown in Figure 2.

Pantropical spotted dolphins have been observed in all months of the year around the main Hawaiian Islands, and in areas ranging from shallow nearshore waters to depths of 5,000 m, although they peak in sighting rates in depths from 1,500 to 3,500 m (Baird *et al.* 2013). Although they represent from 22.9 to 26.5% of the odontocete sightings from Oʻahu, the Maui Nui~~4 Islands~~ (Molokaʻi, Lānaʻi, Maui, Kahoʻolawe), and Hawaiʻi Island, they are largely absent from the nearshore waters around Kauaʻi and Niʻihau, representing only 3.9% of sightings in that area (Baird *et al.* 2013). Genetic analyses of 176 unique samples of pantropical spotted dolphins collected during near-shore surveys off each of the main Hawaiian Islands from 2002 to 2003, and near Hawaiʻi Island from 2005 ~~through~~ to 2008, suggest three island-associated stocks are evident (Courbis *et al*. 2014). The results of the Courbis *et al*. (2014) study indicate that pantropical spotted dolphins in Hawaiʻi's nearshore waters have low haplotypic diversity with haplotypes unique to each of the island areas. Courbis *et al*. (2014) conducted extensive tests on the relatedness of individuals among islands using the microsatellite dataset and found significant differences in haplotype frequencies between islands, suggesting genetic differentiation in spotted dolphins among islands. This suggestion is supported by the results of assignments tests, which indicate support for 3 island-associated populations: Hawaiʻi Island, the Mui Nui~~4 Islands~~ region, and Oʻahu. Samples from Kauaʻi and Niʻihau did not cluster together, but instead were spread among the Hawaiʻi and Oʻahu clusters. Analysis of migration rate further support the separation of pantropical spotted dolphins into three island-associated stocks, with migration between regions on the order of a few individuals per generation. Based on an overview of all available information on pantropical spotted dolphins in Hawaiian waters, and NMFS guidelines for assessing marine mammal stocks (NMFS ~~2005~~2023), Oleson *et al.* (2013) proposed designation of three new island associated stocks in Hawaiian waters, as well as recognition of a fourth broadly distributed spotted dolphin stock given the frequency of sightings in pelagic waters. Stock boundaries for main Hawaiian Islands spotted dolphin stocks are based on the furthest distance from shore of an insular sighting. Around Oʻahu and Maui Nui the stock extends to 20km form shore and around Hawaiʻi Island to 65km. Fishery interactions with pantropical spotted dolphins and sightings near Palmyra and Johnston Atolls (NMFS PIRO unpublished data) demonstrate that this species also occurs in the U.S. EEZ ~~waters there~~of those locations, but it is not known whether these animals are part of the Hawaiʻi Pelagic~~Hawaiian~~ population or are a separate stock or stocks of pantropical spotted dolphins.

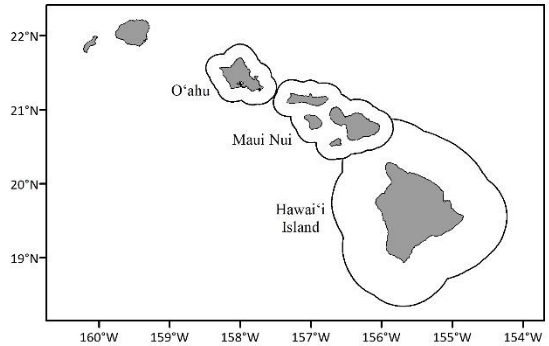

Figure 2. Main Hawaiian Islands insular ~~spotted dolphin~~ stock boundaries (gray lines). Areas beyond the insular boundaries represent the pelagic stock range ~~based on distance of furthest encounter~~. ~~The Oahu and 4 Islands stocks extend 20 km from shore, while the Hawaii Island stock extends to 65 km from shore based on distance of furthest encounter~~.

For the Marine Mammal Protection Act (MMPA) stock assessment reports, there are four Pacific management stocks within the Hawaiian Islands EEZ (Oleson *et al.* 2013): 1) the Oʻahu stock, which includes spotted dolphins within 20 km of Oʻahu, 2) the Maui Nui~~4 Island~~ stock, which includes spotted dolphins within 20 km of Maui, Molokaʻi, Lānaʻi, and Kahoʻolawe, collectively, 3) the Hawaiʻi Island stock, which includes spotted dolphins found within 65 km from Hawaiʻi Island, and 4) the Hawaiʻi Pelagic stock, which includes spotted dolphins inhabiting the waters throughout the Hawaiian Islands EEZ, outside of the insular stock areas, but including adjacent high seas waters. Because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of the Hawaii pelagic stock is evaluated based on data from the U.S. EEZ ~~waters of~~ around the Hawaiian Islands (NMFS ~~2005~~2023). Spotted dolphins involved in eastern tropical Pacific tuna purse-seine fisheries are managed separately under the MMPA.

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Fishery Information

Information on fishery-related mortality of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaiian fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. Entanglement in gillnets and hooking or entanglement in various hook and line fisheries have been reported for small cetaceans in Hawaiʻi (Nitta and Henderson 1993). No estimates of human-caused mortality or serious injury are currently available for nearshore hook and line or gillnet fisheries because these fisheries are not observed or monitored for protected species bycatch. Commercial and recreational troll fisherman have been observed “fishing” dolphins off the islands of Hawaiʻi, Lānaʻi, and Oʻahu, including spotted dolphins, in order to catch tuna associated with the animals (Courbis *et al*. 2009, Rizzuto 2007, Shallenberger 1981). Anecdotal reports from fisherman indicate that spotted dolphins are sometimes hooked (Rizzuto 1997) ~~and photographs of dolphins suggest animals may be injured by both lines and propeller strikes (Baird unpublished data)~~. An assessment of the incidence of potential fishing gear-associated scarring on pantropical spotted dolphins near Maui Nui revealed 13% of non-calf well-marked individuals photographed between 1996 and 2020 had one or more scars that may be attributed to fishing gear (Machernis *et al.* 2021). A study of the incidence of fishing vessels associated with spotted dolphins revealed that hundreds of boats appear to be engaging in this fishing method, including a high incidence of trolling through the group of dolphins ~~and/~~or maneuvering through the group to drop hook and line gear ahead of the dolphin group (Baird and Webster 2020). ~~In 2014, a spotted dolphin (Hawaii Island stock) was observed hooked above the jaw and trailing 8-10 feet of fishing line (Bradford and Lyman 2018).~~ In 2017, a spotted dolphin (Maui Nui~~4-Islands~~ stock) was ~~spotted~~ seen near Lāna‘i with a band of debris around its rostrum preventing it from opening its mouth, which was determined to be a serious injury (Bradford and Lyman 2019). Serious injuries from nearshore fishing gear have previously been observed in other insular stocks (Bradford and Lyman 2018). ~~Based on the information provided, both of these injuries are considered serious injuries. The responsible fishery is not known for either case.~~

There are currently two distinct longline fisheries based in Hawai‘i: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas~~.~~, but are prohibited from operating within the Papahānaumokuākea Marine National Monument (PMNM) and within the Longline Exclusion Zone around the main Hawaiian Islands and the Pacific Remote Islands and Atolls (PRIA) MNM around Johnston Atoll. The PMNM originally included the waters within a 50 nmi radius around the Northwestern Hawaiian Islands. In August, 2016, the PMNM area was expanded to extend to the 200 nmi EEZ boundary west of 163° W. Between 2017~~2014~~ and 2021~~2018~~, no pantropical spotted dolphins were observed hooked or entangled in the SSLL fishery (100% observer coverage) or in the DSLL fishery (~~18~~15-21% observer coverage) (McCracken and Cooper 2022~~Bradford 2018a, 2018b, 2020, Bradford and Forney 2017~~). ~~Three additional unidentified delphinids were taken in the DSLL fishery, some of which may have been spotted dolphins.~~

O‘AHU STOCK

POPULATION SIZE

The population size of the O‘ahu stock of pantropical spotted dolphins has not been estimated. Model-based estimates using line-transect datasets have been explored for this stock (Becker *et al.* 2022), though the small sample size and an uneven distribution of survey effort resulted in unreliable estimates.

Minimum Population Estimate

There is no information on which to base a minimum population estimate of the O‘ahu stock of pantropical spotted dolphins.

Current Population Trend

No data are available on current population trend.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the O‘ahu stock of pantropical spotted dolphins is calculated as the minimum population estimate times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a species of unknown status with no estimated fishery mortality or serious injury within the O‘ahu stock area; Wade and Angliss 1997). Because there is no minimum population estimate available, the PBR for O‘ahu stock of pantropical spotted dolphins is undetermined.

STATUS OF STOCK

The O‘ahu stock of pantropical spotted dolphins is not considered a strategic stock under the MMPA. The status of O‘ahu spotted dolphins relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance for this stock. Spotted dolphins are not listed as “threatened” or “endangered” under the Endangered Species Act (1973), nor designated as “depleted” under the MMPA. ~~There is no information with which to determine whether the total fishery mortality and serious injury for this stock is insignificant and approaching zero mortality and serious injury rate.~~ There have been no reports of recent mortality or serious injuries, though fishermen may target groups of spotted dolphins around O‘ahu in order to catch associated tuna, increasing the likelihood of dolphins being hooked or entangled (Baird and Webster 2020). There is no systematic monitoring for interactions with protected species within nearshore fisheries that may take this species, thus mean annual takes are undetermined. Insufficient information is available to determine whether the total fishery mortality and serious injury for this stock is insignificant and approaching zero mortality and serious injury rate. *Morbillivirus* has been detected within other insular stocks of pantropical spotted dolphins in Hawai‘i (Jacob *et al.* 2016). The presence of *morbillivirus* in 10 species of cetacean in

Hawaiian waters raises concerns about the history and prevalence of this disease in Hawaiʻi and the potential population impacts, including the cumulative impacts of disease with other stressors.

~~4-ISLANDS~~MAUI NUI STOCK

POPULATION SIZE

The population size of Maui Nui~~4-Islands~~ stock of pantropical spotted dolphins has not been estimated. Model-based estimates using line-transect datasets have been explored for this stock (Becker *et al.* 2022), though the small sample size and an uneven distribution of survey effort resulted in unreliable estimates.

Minimum Population Estimate

There is no information on which to base a minimum population estimate of the Maui Nui~~4-Islands~~ stock of pantropical spotted dolphins.

Current Population Trend

No data are available on current population trend.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Mui Nui~~4-Islands~~ stock of pantropical spotted dolphins is calculated as the minimum population estimate times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a species of unknown status with no estimated fishery mortality or serious injury within the Mui Nui~~4-Islands~~ stock area; Wade and Angliss 1997). Because there is no minimum population estimate available for this stock, the PBR for 4-Islands stock of pantropical spotted dolphins is undetermined.

STATUS OF STOCK

The Maui Nui~~4-Islands~~ stock of pantropical spotted dolphins is not considered a strategic stock under the MMPA. The status of Maui Nui~~4-Islands~~ spotted dolphins relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance for this stock. Spotted dolphins are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. Aside from the spotted dolphin entangled in marine debris in 2017 (Bradford and Lyman 2018), there are no other reports of recent mortality or serious injuries, though fishermen may target groups of spotted dolphins around the Maui Nui region to catch associated tuna, increasing the likelihood of dolphins being hooked or entangled (Baird and Webster 2020), with injuries potentially associated with fishing line observed in a portion of well-marked animals (Machernis *et al.* 2021). There is no systematic monitoring for interactions with protected species within nearshore fisheries that may take this species, thus mean annual takes are undetermined. Insufficient information is available to determine whether the total fishery mortality and serious injury for this stock is insignificant and approaching zero mortality and serious injury rate. *Morbillivirus* has been detected within other insular stocks of pantropical spotted dolphins in Hawaiʻi (Jacob *et al.* 2016). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters raises concerns about the history and prevalence of this disease in Hawaiʻi and the potential population impacts, including the cumulative impacts of disease with other stressors.

HAWAIʻI ISLAND STOCK

POPULATION SIZE

The population size of the Hawaiʻi Island stock of pantropical spotted dolphins has not been estimated. Design and model-based estimates using line-transect datasets have been explored for this stock (Becker *et al.* 2022, Bradford *et al.* 2022), though the small sample size and an uneven distribution of survey effort resulted in unreliable estimates.

Minimum Population Estimate

There is no information on which to base a minimum population estimate of the Hawaiʻi Island stock of pantropical spotted dolphins.

Current Population Trend

No data are available on current population trend.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

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

STATUS OF STOCK

The Hawaiʻi Island stock of pantropical spotted dolphins is not considered a strategic stock under the MMPA. The status of Hawaiʻi Island spotted dolphins relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance for this stock. Spotted dolphins are not listed as “threatened” or “endangered” under the Endangered Species Act (1973), nor designated as “depleted” under the MMPA. There have been no reports of recent mortality or serious injuries, though fishermen target groups of spotted dolphins around Hawaiʻi Island to catch associated tuna, increasing the likelihood of dolphins being hooked or entangled (Baird and Webster 2020). There is no systematic monitoring for interactions with protected species within nearshore fisheries that may take this species, thus mean annual takes are undetermined. Insufficient information is available to determine whether the total fishery mortality and serious injury for this stock is insignificant and approaching zero mortality and serious injury rate. One spotted dolphin found stranded on Hawaiʻi Island has tested positive for *Morbillivirus* (Jacob *et al.* 2016). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters (Jacob 2012) raises concerns about the history and prevalence of this disease in Hawaii and the potential population impacts, including the cumulative impacts of disease with other stressors.

HAWAIʻI PELAGIC STOCK

POPULATION SIZE

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

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

Year	Design-based Abundance	CV	95% Confidence Limits	Model-based Abundance	CV	95% Confidence Limits
2020				67,313	0.27	40,096-113,005
2017	39,798	0.51	15,432-102,637	73,667	0.28	42,769-126,886
2010	49,488	0.39	23,551-103,992			
2002	16,931	0.65	5,289-54,202			

Sighting data from 2002 to 2020 within the Hawaiian Islands EEZ were used to derive stock-specific habitat-based models of animal density for the 2017 to 2020 period. The models were then used to predict density and abundance for each survey year based on the environmental conditions within that year (see Forney *et al.* 2015, Becker *et al.* 2016). The modeling framework incorporated Beaufort-specific trackline detection probabilities for spotted dolphins from Barlow *et al.* (2015). Although model-based estimates were previously derived for years 2002, 2010, and 2017 (Becker *et al.* 2021), those estimates were not pelagic stock specific and as such may have reflected both the habitat associations and abundance of the insular stocks within the main Hawaiian Islands. Stock-specific model-based estimates were derived only for the most recent years (2017-2020), such that direct comparison of model and design-based estimates for the full survey time series is not possible at this time. Bradford *et al.* (2021) produced design-based abundance estimates for spotted dolphins for each full EEZ survey year. Design-based estimates for spotted dolphins are generally lower than model-based estimates, though the confidence limits broadly overlap with those produced from the model-based approach (Figure 3). Current model based-estimates are based on the implicit assumption that annual changes in abundance are attributed to environmental variability alone. Explicitly incorporating a trend term into the model is not possible due to the insufficient sample size to test for temporal effects. Despite not fully accounting for inter-annual variation in total abundance, the model-based estimates are considered the best available estimate for the most recent survey year. Becker *et al.* (2022) and Bradford *et al.* (2022) evaluated seasonal changes in the abundance of spotted dolphins within the main Hawaiian Islands using summer-fall data from 2017 and winter survey data from 2020 and found no significant difference, with no reliance on season within the model-based approach and largely similar design-based estimates for summer-fall 2017 versus winter 2020. Previously published design-based estimates for the Hawaiian Islands EEZ from 2002 and 2010 surveys (e.g. Barlow 2006, Bradford *et al.* 2017) used a subset of the dataset used by Becker *et al.* (2021, 2022) and Bradford *et al.* (2021, 2022) to derive line-transect parameters, such that these estimates have been superseded by the estimates presented here. The best estimate of abundance is based on the 2020 survey, or 67,313 (CV=0.27) pantropical spotted dolphins.

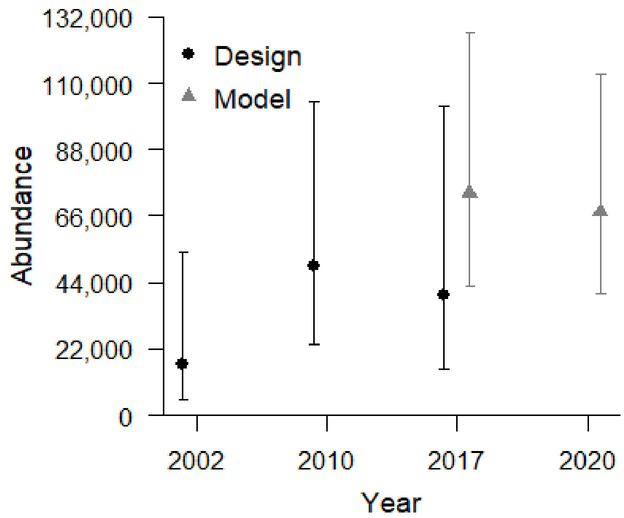

Figure 3. Comparison of design-based (black circles, Bradford *et al.* 2021) and model-based (gray triangles, Becker *et al.* 2021, 2022) estimates of abundance for hawai‘i pelagic pantropical spotted dolphins for each survey year (2002, 2010, 2017, 2020).

Population estimates are available for Japanese waters (Miyashita 1993), but it is not known whether any of these animals are part of the same population that occurs around the Hawaiian Islands.

Minimum Population Estimate

The minimum population estimate for the Hawai‘i pelagic stock of pantropical spotted dolphins~~size~~ is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2020~~2017~~ abundance estimate for the pelagic stock area (from Becker *et al.* 2022), or 53,839~~26,548~~ pantropical spotted dolphins.

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Hawai‘i pelagic stock of pantropical spotted dolphins ~~stock~~ is calculated as the minimum population estimate within the U.S. EEZ of the Hawaiian Islands (53,839~~26,548~~) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a species of unknown status with no known fishery mortality within the U.S. EEZ of the Hawaiian Islands; Wade and Angliss 1997), resulting in a PBR of 538~~265~~ Hawai‘i pelagic pantropical spotted dolphins per year.

STATUS OF STOCK

The Hawai'i pelagic stock of spotted dolphins is not considered strategic under the 1994 amendments to the MMPA. The status of Hawai'i pelagic pantropical spotted dolphins relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Pantropical spotted dolphins are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. Given the absence of recent recorded fishery-related mortality or serious injuries within U.S. EEZ, the total fishery mortality and serious injury can be considered to be insignificant and approaching zero. *Morbillivirus* has been detected within other insular stocks of bottlenose dolphins in Hawai'i (Jacob *et al.* 2016). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters raises concerns about the history and prevalence of this disease in Hawai'i and the potential population impacts, including the cumulative impacts of disease with other stressors.

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Revised ~~5/5/2021~~ 08/16/2023

STRIPED DOLPHIN (*Stenella coeruleoalba*): Hawaiʻi Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Striped dolphins are found in tropical to warm-temperate waters throughout the world (Perrin *et al.* 2009). Sightings have historically been infrequent in shallow waters (Shallenberger 1981, Mobley *et al.* 2000), though they are common, even nearshore, in waters greater than 3500m (Baird 2016). Striped dolphins are often seen offshore throughout the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands during periodic shipboard surveys ~~Summer/fall shipboard surveys of the waters within the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands, resulted in 15 sightings of striped dolphins in 2002. 29 in 2010, and 27 in 2017~~ (Figure 1~~; Barlow 2006, Bradford *et al.* 2017, Yano *et al.* 2018~~).

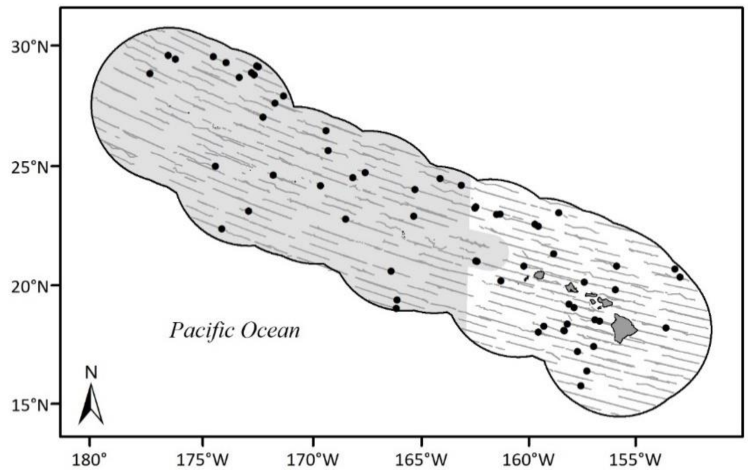

Figure 1. Striped dolphin sighting locations (circles) and survey effort (gray lines) during the 2002 (Barlow 2006~~diamonds~~), 2010 (Bradford *et al.* 2017~~circles~~), and 2017 (Yano *et al.* 2018~~squares~~) shipboard surveys of the U.S. EEZ ~~waters surrounding~~ around the Hawaiian Islands (~~Barlow 2006, Bradford *et al.* 2017, Yano *et al.* 2018~~outer black line). ~~Outer line represents approximate boundary of survey area and U.S. EEZ. Dark gray shading indicates the original~~ The Papahānaumokuākea Marine National Monument in the western portion of the EEZ is shaded gray~~, with the lighter gray shading denoting the full 2016 Expansion area. Dotted line represents the 1000 m isobath~~.

Striped dolphins have been intensively exploited in the western North Pacific, where three migratory stocks are provisionally recognized (Kishiro and Kasuya 1993). In the eastern tropical Pacific, all striped dolphins are provisionally considered to belong to a single stock (Dizon *et al.* 1994). There is insufficient data to examine finer stock structure within Hawaiian waters, though data available to date do not suggest island-associated populations for this species (Baird 2016).

For the Marine Mammal Protection Act (MMPA) stock assessment reports, striped dolphins within the Pacific U.S. EEZ are divided into two discrete, non-contiguous areas: 1) waters off California, Oregon and Washington, and 2) waters around Hawaiʻi (this report), including animals found both within the Hawaiian Islands EEZ and in adjacent high seas waters. Because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of the Hawaiʻi stock is evaluated based on data from the U.S. EEZ ~~waters of~~ around the Hawaiian Islands (NMFS ~~2005~~2023). Striped dolphins involved in eastern tropical Pacific tuna purse-seine fisheries are managed separately under the MMPA.

POPULATION SIZE

Encounter data from shipboard line-transect surveys of the ~~entire~~ Hawaiian Islands EEZ were~~was~~ recently reevaluated for each survey year, resulting in updated model-based abundance estimates of striped dolphins in the entirety of the Hawaiian Islands EEZ (Becker *et al.* 2021, 2022; Table 1).

Sighting data from 2002 to 2020~~2017~~ within the Hawaiian Islands EEZ were used to derive habitat-based models of animal density for two periods: 2002-2017 (Becker *et al.* 2021) and 2017-2020 (Becker *et al.* 2022). The most recent set of models include three notable changes from the 2002-2017 models: use of calibrated group size estimates, as in Bradford *et al.* (2021), exclusion of a spatial term on model selection, requiring more explicit reliance on environmental variables, and incorporating new approaches (Miller *et al.* 2022) for more comprehensively estimating uncertainty in model predictions that account for the combined uncertainty around all parameter estimates~~the overall period~~. The modeling framework incorporated Beaufort-specific trackline detection probabilities for striped dolphins from Barlow *et al.* (2015). ~~The m~~Models were ~~then~~ used to predict density and abundance for

Table 1. Model-based ~~L~~line-transect abundance estimates for striped dolphins in the~~derived from surveys of the entire~~ Hawaiian Islands EEZ in 2002~~,~~ and 2010 (Becker *et al.* 2021)~~,~~ and ~~and~~ 2017~~,~~ and 2020 (Becker *et al.* ~~2021,~~ 2022), derived from NMFS surveys in the central Pacific since 2000. The Becker *et al.* (2022) analysis incorporates a more comprehensive model-based approach to estimating model uncertainty, such that the CVs and 95% confidence limits for 2002/2010 and 2017/2020 are not directly comparable

Year	Model-based Abundance	CV	95% Confidence Limits
2020	64,343	0.28	37,822-109,462
2017	59,493 ~~35,179~~	0.28 ~~0.23~~	35,050-100,981 ~~22,416-55,209~~
2010	36,886	0.22	24,004-56,681
2002	35,817	0.22	23,384-54,861

each survey year based on the environmental conditions within that year (see Forney *et al.* 2015, Becker *et al.* 2016).~~The modeling framework incorporated Beaufort-specific trackline detection probabilities for striped dolphins from Barlow et al. (2015).~~ When model-based estimates are available for 2017 from both analyses, the results are largely similar for most species; however, striped dolphins are a notable exception, with 2017 estimates from Becker *et al*, (2022) nearly double those from Becker *et al.* (2021). Although Becker *et al.* (2022) attribute this change to the use of new calibrated group size, detailed review of the functional form of the model predictors reveal a shift from a linear decline in density with depth in Becker *et al.* (2021) to a thresholded form in Becker *et al.* (2022), with density constant at depths less than 3000m, leading to higher densities in shallow depths than the previous models. Bradford *et al.* (2021) produced design-based abundance estimates for striped dolphins in~~for each~~ 2002, 2010, and 2017~~survey year~~ that can be used as a point of comparison to the model-based estimates for those years. There is substantial variability within and between the design and model-based estimates across the time series~~While on average the estimates are broadly similar between the two approaches, the annual design-based estimates show much greater variability between years than do the model-based estimates~~ (Figure 2), suggesting additional survey data are needed to develop a well-parameterized model for this species. ~~The model-based approach reduces variability through explicit examination of habitat relationships across the full dataset, while the design-based approach evaluates encounter data for each year separately and thus is more susceptible to the effects of encounter rate variation. Model based estimates are based on the implicit assumption that changes in abundance are attributed to environmental variability alone. There are insufficient data to explicitly incorporate a trend term into the model due to the insufficient sample size to test for temporal effects.~~ Despite the substantial variability in the abundance estimates for this species, ~~not fully accounting for inter-annual variation in total abundance,~~ the model-based estimates are considered the best available estimate for each survey year. Becker *et al.* (2022) and Bradford *et al.* (2022) evaluated seasonal changes in the abundance of striped dolphins within the main Hawaiian Islands using summer-fall data from 2017 and winter survey data from 2020. Seasonal predictions using the model showed no reliance on dynamic variables, and design-based estimates were broadly similar (with broad and overlapping confidence intervals).Previously published ~~design-based~~ abundance estimates for the Hawaiian Islands EEZ ~~from 2002 and 2010 surveys~~ (e.g. Barlow 2006, Becker *et al.* 2012, Forney *et al.* 2015, Bradford *et al.* 2017) used a subset of the dataset used by Becker *et al.* (2021, 2022) and Bradford *et al.* (2021) to derive line-transect

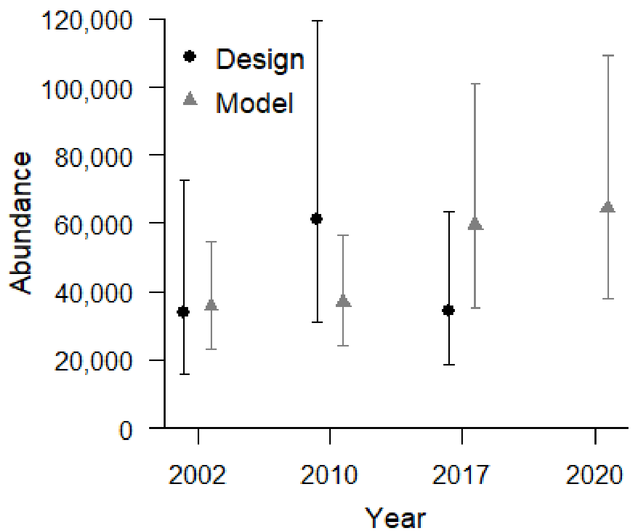

Figure 2. Comparison of design-based (black circles, Bradford *et al.* 2021) and model-based (gray triangles, Becker *et al.* 2021, 2022) estimates of abundance for striped dolphins for each survey year (2002, 2010, 2017, 2020).

parameters, such that these estimates have been superseded by the estimates presented here. The best estimate of abundance is based on the 2020~~2017~~ survey, or 64,343 (CV=0.28)~~35,179 (CV=0.23)~~ striped dolphins.

Population estimates are available for Japanese waters (Miyashita 1993) and the eastern tropical Pacific (Wade and Gerrodette 1993), but it is not known whether any of these animals are part of the same population that occurs around the Hawaiian Islands.

Minimum Population Estimate

The minimum population estimate~~size~~ is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2020~~2017~~ abundance estimate (from Becker *et al.* 2022), or 51,055~~29,058~~ striped dolphins.

Current Population Trend

The model-based abundance estimates for striped dolphins provided by Becker *et al.* (2021, 2022) are highly variable and do not explicitly allow for examination of population trend~~ other than that driven by environmental factors~~. Model-based examination of striped dolphin population trends including sighting data beyond the Hawaiian Islands EEZ will be required to more fully examine trend for this stock.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Hawai'i stock of striped dolphins is calculated as the minimum population size within the U.S. EEZ of the Hawaiian Islands (51,055~~29,058~~) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.5 (for a stock of unknown status with no known fishery mortality and serious injury within the Hawaiian Islands EEZ; Wade and Angliss 1997), resulting in a PBR of 511~~291~~ striped dolphins per year.

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Fishery Information

Information on fishery-related mortality and serious injury of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaiian fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. Entanglement in gillnets and hooking or entanglement in various hook and line fisheries have been reported for small cetaceans in Hawaii (Nitta and Henderson, 1993). In 2021, a striped dolphin stranded on Maui with scarring on its rostrum consistent with a previous hooking and scarring on its peduncle consistent with a previous entanglement, although these findings were not considered to be related to the cause of death (Bradford and Lyman in press). No estimates of human-caused mortality or serious injury are currently available for nearshore hook and line or gillnet fisheries because these fisheries are not observed or monitored for protected species bycatch.

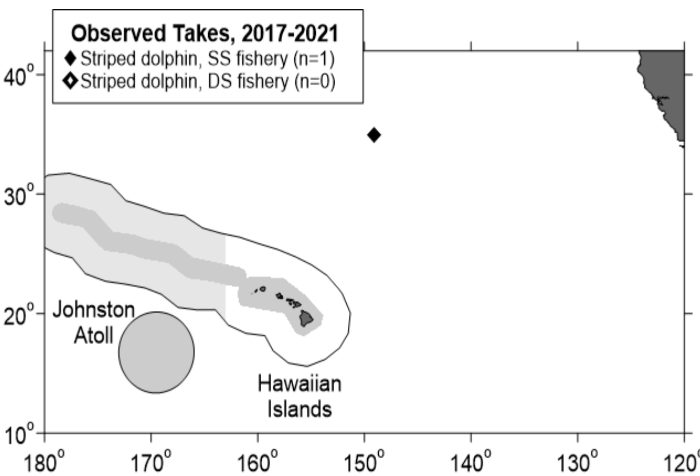

Figure 3. Location~~s~~ of a striped dolphin take~~s~~ within the shallow-set fishery (filled diamond) in Hawaii-based longline fisheries, ~~2014-2018~~2017-2021. Solid lines represent the U.S. EEZs. Gray shading notes areas closed to longline fishing.

There are currently two distinct longline fisheries based in Hawai'i: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas. Between 2017~~2014~~ and 2021~~2018~~, one~~two~~ striped dolphin~~s were~~ was observed hooked or entangled in the SSLL fishery (100% coverage) outside of the U.S. EEZ, and no ~~one~~ striped dolphins ~~was~~were observed hooked or entangled ~~seriously injured~~ in the DSLL fishery (~~18~~15-21% observer coverage) (Figure 3, Bradford 2018, 2020, 2021, 2023, in review). The striped dolphin was considered not seriously injured based on an evaluation of the observer's description of the interaction and following the most recently developed criteria for assessing serious injury in marine mammals (NMFS 2023b). ~~All striped dolphin interactions occurred outside of the U.S. EEZs.~~

The total estimated number of dead or seriously injured dolphins is calculated based on observer coverage rate, the location of the observed take (inside or outside of the EEZ), and the ratio of observed dead and seriously injured whales versus those judged to be not seriously injured. Observer coverage is measured on a per-trip basis throughout the calendar year as described by McCracken (2019). In years with large fluctuations in observer coverage, such as during the early days of the COVID-19 pandemic when observer coverage dropped to less than 10% during the second quarter of the year, the annual bycatch estimation process may be subset into several periods, as described in McCracken and Cooper (2022a). Average 5-yr estimates of annual mortality and serious injury for 2017-2022~~2014-2018~~ are 0.2~~0.4 (CV = 1.0)~~ dolphins outside of the U.S. EEZ~~s~~, and 0~~zero~~ within the Hawaiian Islands EEZ (Table 2). ~~Three additional unidentified cetaceans were taken in the DSLL fishery, some of which may have been striped dolphins.~~

Table 2. Summary of available information on incidental mortality and serious injury (MSI) of striped dolphin (Hawai‘i stock) in commercial longline fisheries, within and outside of the U.S. EEZ~~s~~ (McCracken and Cooper ~~2022a,~~ 2022b~~2019~~). Mean annual takes are based on 2017-2021~~2014-2018~~ data unless otherwise indicated. Information on all observed takes (T) and ~~combined mortality events & serious injuries (~~MSI~~)~~ is included. Total takes were prorated to deaths, serious injuries, and non-serious injuries based on the observed proportions of each outcome.

				~~Observed total interactions (T) and mortality events, and serious injuries (MSI), and total estimated mortality and serious injury (M&SI) of striped dolphins~~			
				Outside U.S. EEZ~~s~~		**Hawaiian Islands EEZ**	
Fishery Name	**Year**	**Data Type**	**Percent Observer Coverage**	**Observed~~.~~ T/MSI**	**Estimated M&SI (CV)**	**Obsserved~~.~~ T/MSI**	**Estimated M&SI (CV)**
Hawai‘i-based deep-set longline fishery	~~2014~~	Observer data	~~21%~~	~~0~~	~~0 (-)~~	~~0~~	~~0 (-)~~
	~~2015~~		~~21%~~	~~1/0~~	~~3 (1.1)~~	~~0~~	~~0 (-)~~
	~~2016~~		~~20%~~	~~0~~	~~0 (-)~~	~~0~~	~~0 (-)~~
	2017		20%	0	0 (-)	0	0 (-)
	2018		18%	0	0 (-)	0	0 (-)
	2019		21%	0	0 (-)	0	0 (-)
	2020		15%	0	0 (-)	0	0 (-)
	2021		18%	0	0 (-)	0	0 (-)
Mean Estimated Annual Take (CV) 2017-2021					**0 (-)~~0.4 (1.0)~~**		**0 (-)**
Hawai‘i-based shallow-set longline fishery	~~2014~~	Observer data	~~100%~~	~~2/2‡~~	~~2~~	~~0~~	~~0~~
	~~2015~~		~~100%~~	~~0~~	~~0~~	~~0~~	~~0~~
	~~2016~~		~~100%~~	~~1/1~~	~~2~~	~~0~~	~~0~~
	2017		100%	1/0	1	0	0
	2018		100%	0	0	0	0
	2019		100%	0	0 (-)	0	0 (-)
	2020		100%	0	0 (-)	0	0 (-)
	2021		100%	0	0 (-)	0	0 (-)
Mean Annual Takes (100% coverage) 2017-2021					**0.2~~0.5~~**		**0**
Minimum total annual takes within U.S. EEZ (2017-2021)							**0 (-)**

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

STATUS OF STOCK

The Hawai‘i stock of striped dolphins is not considered strategic under the 1994 amendments to the MMPA. The status of striped dolphins in Hawaiian waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Striped dolphins are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. Given the absence of recent recorded fishery-related mortality or serious injuries in the U.S. EEZ ~~waters~~, total fishery mortality and serious injury for striped dolphins can be considered insignificant and approaching zero. Several serious diseases have been found in stranded striped dolphins in Hawai‘i. One striped dolphin stranded in the main Hawaiian Islands tested positive for *Brucella* (Chernov 2010), ~~and~~ two for *Morbillivirus* (Jacob *et al.* 2016), and one for beaked whale circovirus (Clifton *et al.* 2023). *Brucella* is a bacterial infection that if common in the population may limit recruitment by compromising male and female reproductive systems, and can also cause neurological disorders that may result in death (Van Bressem *et al.* 2009). Although M*orbillivus* is known to trigger lethal disease in cetaceans (Van Bressem *et al.* 2009), its impact on the health of the stranded animals is not known as it was found in only a one tested tissue within each animal (Jacob *et al.* 2016). Beaked whale circovirus has been only recently described in cetaceans, with effects on the brain, lungs, and lymph system that may result in immunosuppression. Its role in the death of the striped dolphin was not clear, although all 6 tested tissues were positive for the disease. The presence of beaked whale circovirus and *Morbillivirus* each in 10 species (Clifton *et al.* 2023, Jacob *et al.* 2016) and *Brucella* in 3 species (Cherbov 2010, West unpublished data) raises concerns about the history and prevalence of these diseases in Hawaiʻi and the potential population impacts on Hawaiian cetaceans. It is not known if any of these diseases ~~*Brucella* or *Morbillivirus*~~ are common in the Hawaiʻi stock of striped dolphins.

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

STOCK DEFINITION AND GEOGRAPHIC RANGE

False killer whales are found worldwide in tropical and warm-temperate waters (Stacey *et al.* 1994). In the North Pacific, this species is well known from southern Japan, Hawai‘i, and the eastern tropical Pacific. False killer whales have been ~~were~~ encountered during periodic ~~three~~ shipboard line-transect surveys of the U.S. Exclusive Economic Zone (EEZ) around the Hawaiian Islands ~~in 2002, 2010, and 2017~~ (Figure 1~~; Barlow 2006, Bradford *et al.* 2014, Yano *et al.* 2018~~), and focused studies near the main and Northwestern Hawaiian Islands (NWHI) indicate that false killer whales occur in nearshore waters throughout the Hawaiian archipelago (Baird *et al.* 2008, 2013). This species also occurs in the U.S. EEZ ~~waters~~ around Palmyra and Johnston Atolls (e.g., Barlow *et al.* 2008) and American Samoa (Johnston *et al.* 2008, Oleson 2009).

Genetic, photo-identification, and telemetry studies indicate there are several ~~three~~ demographically independent populations of false killer whales throughout the Pacific and three in Hawaiian waters. Genetic analyses indicate restricted gene flow between island-associated populations of false killer whales sampled near the main Hawaiian Islands (MHI), the ~~Northwestern Hawaiian Islands (~~NWHI~~)~~, versus those ~~and~~ in pelagic waters of the Eastern (ENP) and Central North Pacific (CNP) (Chivers *et al.* 2010; Martien 2014),~~. Martien *et al.* (2014) analyzed mitochondrial DNA (mtDNA) control region sequences and genotypes from 16 nuclear DNA (nuDNA) microsatellite loci from 206 individuals from the MHI, NWHI, and offshore waters of the CNP and ENP and showed highly significant differentiation between populations confirming limited gene flow in both sexes.~~ The mtDNA analysis reveals strong phylogeographic patterns consistent with local evolution of haplotypes unique to false killer whales occurring nearshore within the Hawaiian Archipelago, while

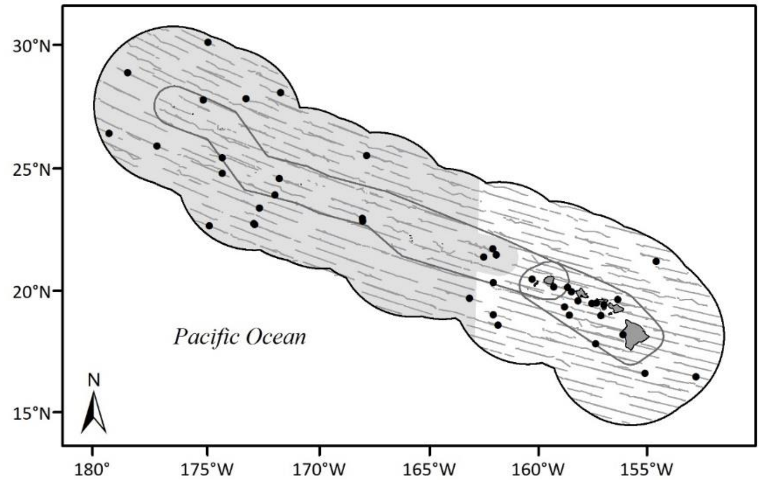

Figure 1. False killer whale sighting locations (circles) and survey effort (gray lines) during the 2002 (~~diamond~~Barlow 2006), 2010 (~~circle~~Bradford *et al.* 2017), and 2017 (~~square~~Yano *et al.* 2018) shipboard surveys of the U.S. EEZ ~~waters surrounding~~around the Hawaiian Islands (~~Barlow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

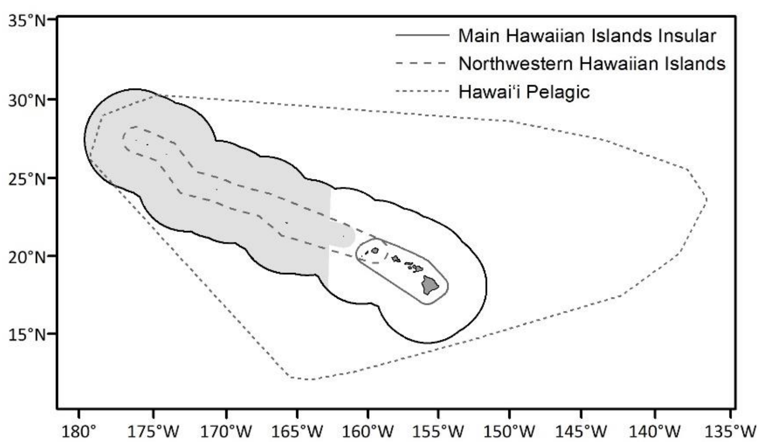

Figure 2. Main Hawaiian Islands insular and Northwestern Hawaiian Islands false killer whale stock boundaries (gray lines). Outer black line represents ~~approximate boundary of survey area and~~the U.S. EEZ.

the nuDNA analysis suggests NWHI false killer whales are at least as differentiated from MHI animals as they are from offshore animals. Photo-ID and social network analyses of individuals seen near the MHI indicate a tight social network with no connections to false killer whales seen near the NWHI or offshore waters, and satellite telemetry collected from 27 tagged MHI false killer whales shows movements restricted to the MHI (Baird *et al.* 2010, 2012). Further analysis of photographic and genetic data from individuals seen near the MHI suggests the occurrence of 4~~three~~ separate social clusters (Mahaffy *et al.* 2023)~~Baird *et al.* 2012, Martien *et al.* 2019~~. Parentage analysis of

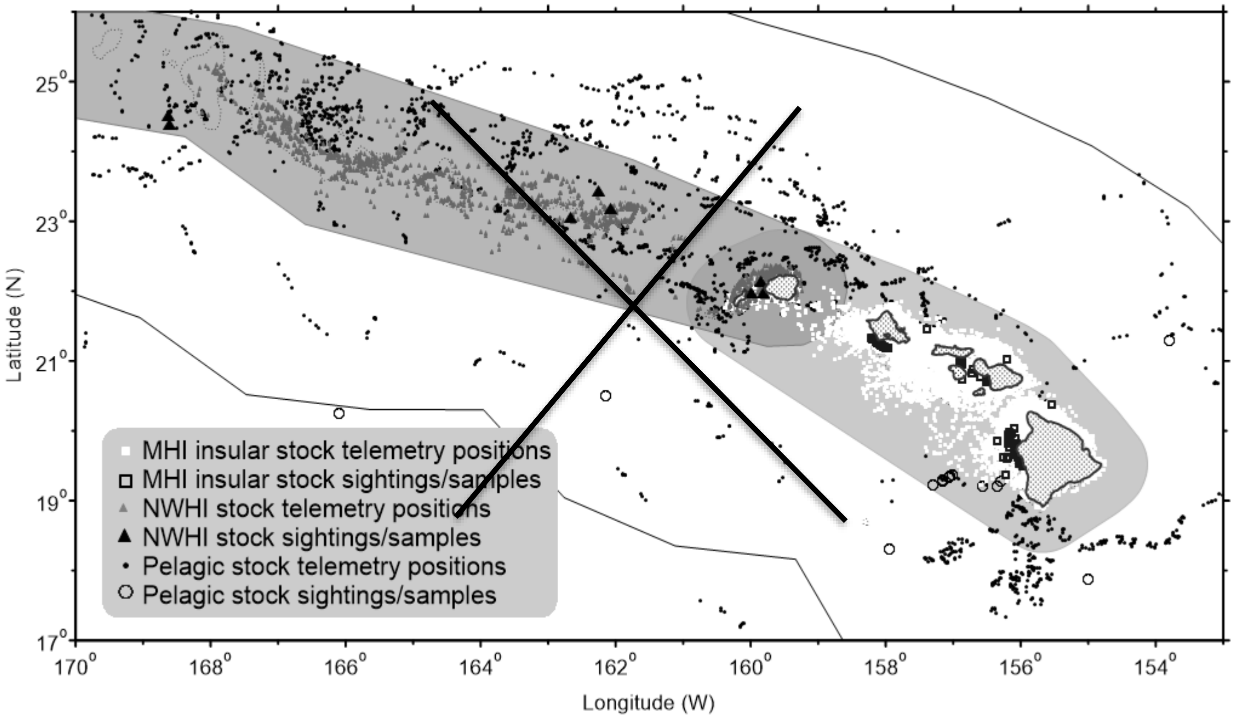

Figure 23. ~~Sighting, biopsy sample, and telemetry record locations of false killer whales identified as being part of the MHI insular (squares), NWHI (triangles), or pelagic (circles) stocks. The MHI stock area is shown in light gray; the NWHI stock area is shown in dark gray; the pelagic stock area includes the entire EEZ (reproduced from Bradford *et al.* 2015, with pelagic stock boundary revision described in Bradford *et al.* 2020). The MHI insular, pelagic, and NWHI stocks overlap around Kauai and Niihau.~~

sampled individuals reveals natal group fidelity of males and females and mating within the natal group 36-64% of the time (Martien *et al.* 2019). Additional evidence for the separation of false killer whales in Hawaiian waters into three separate stocks is summarized by Oleson *et al.* (2010, 2012).

~~Fishery observers have collected tissue samples for genetic analysis from cetaceans incidentally caught in the Hawaii-based longline fisheriesy since 2003. Between 2003 and 2010, eight false killer whale samples, 4 collected outside the Hawaiian Islands EEZ and 4 collected within the EEZ, but more than 100 nautical miles (185km) from the main Hawaiian Islands, were determined to have Pacific pelagic haplotypes (Chivers *et al.* 2010).~~ Outside of the Hawaiian insular waters structure is also evident. ~~At the broadest scale~~, ~~s~~Significant differences in both mtDNA and nuDNA are evident between pelagic false killer whales in the ENP and CNP ~~strata~~ (Chivers *et al.* 2010, Martien *et al.* 2014) and telemetry data from 10 pelagic false killer whales tagged within the Hawaiian Islands EEZ indicates pelagic animals there also use waters to the east of the EEZ (Andrerson *et al.* 2021, Oleson *et al.* 2023). ~~Sample distribution east and west of Hawaiian waters is insufficient to determine whether the sampled strata represent one or more stocks, and where pelagic stock boundaries may occur.~~ Large gaps genetic sample distribution throughout the tropical Pacific preclude finer delineation of population structure and boundaries for pelagic populations.

The stock range and boundaries for ~~of the three~~ Hawai‘~~ian~~ insular stocks of false killer whales are reviewed in Bradford *et al.* (2015), and the management area for Hawai‘i pelagic false killer whlaes is reviewed in Oleson *et al.* (2023)~~and revised for the pelagic stock in Bradford *et al.* (2020)~~ (Figure 2). The three stocks have partially overlapping ranges within the Hawaiian Islands EEZ. MHI insular false killer whales have been satellite tracked as far as 115 km from the main Hawaiian Islands. NWHI false killer whales have been seen up to 93 km from the NWHI and near-shore around Kaua‘i and O‘ahu (Baird *et al.* 2012, Bradford *et al.* 2015).~~, while~~Hawai‘i pelagic stock animals have been satellite tracked to within 5.6 km of the main Hawaiian Islands, ~~and~~ throughout the NWHI, and east outside of the EEZ to 138^0 W. ~~NWHI false killer whales have been seen up to 93 km from the NWHI and near-shore around Kauai and Oahu (Baird *et al.* 2012, Bradford *et al.* 2015).~~ Stock boundary descriptions are complex, but can be summarized as follows. The MHI insular stock boundary is derived from a Minimum Convex Polygon (MCP) bounded around a 72-km radius of the MHI, resulting in a boundary shape that reflects greater offshore use in the leeward portion of the MHI. The NWHI stock boundary is defined by a 93-km radius around the NWHI, with this radial boundary extended to the southeast to encompass Kauaʻi and Niʻihau. The NWHI boundary is latitudinally expanded at the eastern end of the NWHI to encompass animal movements observed outside of the 93-km radius (Figure 2). The Hawaiʻi pelagic stock has no inner ~~or outer~~ boundary within the EEZ. The management area for the Hawaiʻi pealgic stock is defined by an MCP around all genetic, telemetry, sighting, and bycatch location data known or assumed to be of Hawaiʻi pelagic stock animals with a 35km buffer around the points. This management area extends throughout most of the Hawaiian Islands EEZ, east to 132^0 W and south to 12^0 N with a complex shape (Figure 2). ~~The 2015 boundary revision placed an inner boundary at 11km from shore around each of the MHI, though this boundary was removed, given new telemetry data indicating use of waters within 5.6 km the MHI (Bradford *et al.* 2020).~~ The construction of these stock boundaries results in multiple stock overlap zones. The entirety of the MHI insular stock area is an overlap zone between the MHI insular and Hawaiʻi pelagic stocks. The entirety of the NWHI stock range is an overlap zone between NWHI and Hawaiʻi pelagic false killer whales. All three stocks overlap out to the MHI insular stock boundary between Kauaʻi and Nihoa and to the NWHI stock boundary between Kauaʻi and Oʻahu (see Figure 2).

~~The pelagic stock includes animals found within the U.S EEZ around Hawaii and in adjacent international waters. New model-based line-transect abundance estimates for the central Pacific enable examination of the status of the broader population of false killer whales relative to human-caused impacts from U.S. fisheries operating in international waters.~~ The Palmyra Atoll stock of false killer whales is ~~still~~ considered to be a separate stock because comparisons amongst false killer whales sampled at Palmyra Atoll and those sampled from the MHI insular stock and the pelagic ENP reveal restricted gene flow, although the sample size remains too low for robust comparisons (Chivers *et al.* 2010). ~~The status of Hawaii pelagic stock is evaluated based on data from U.S. EEZ waters ofaround the Hawaiian Islands (NMFS 2005), and abundance estimates for the broader central Pacific (including Palmyra Atoll) are provided for comparison to U.S. fisheries impacts on the high-seas~~.

For the Marine Mammal Protection Act (MMPA) stock assessment reports, there are five Pacific Islands Region management stocks: 1) the Main Hawaiian Islands insular stock, which includes animals inhabiting waters within a modified 72 km radius around the ~~main Hawaiian Islands~~MHI, 2) the Northwestern Hawaiian Islands stock, which includes animals inhabiting waters within a 93 km radius around the NWHI and Kauaʻi, with a latitudinal expansion of this area at the eastern end of the range, 3) the Hawaiʻi pelagic stock, which includes false killer whales inhabiting waters of the U.S. EEZ around Hawaiʻi and adjacent high seas waters, as defined by the Hawaiʻi pelagic false killer whale management area (Oleson *et al.* 2023), 4) the Palmyra Atoll stock, which includes animals found within the U.S. EEZ of Palmyra Atoll, and 5) the American Samoa stock, which includes animals found within the U.S. EEZ of American Samoa. Estimates of abundance, potential biological removal, and status determinations for the first three stocks are presented below. Palmyra Atoll and American Samoa stocks appear in separate reports.

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Fishery Information

Interactions with false killer whales, including depredation of pelagic fish catch, are identified in logbooks and NMFS observer records from Hawaiʻi pelagic longline fishing trips (Nitta & Henderson 1993, Oleson *et al.* 2010, PIRO 2015). False killer whales have been observed feeding on a variety of large pelagic fish, including mahi mahi (*Coryphaena hippurus*), yellowfin tuna (*Thunnus albacares*), big eye tuna (*T. obesus*), albacore *(T. alalunga)*, wahoo (*Acanthocybium solandri*), skipjack (*Katsuwonus pelamis*), and broadbill swordfish (*Xiphias gladius*) (Baird 2016), and they are reported to take large fish from troll lines of commercial and recreational fishermen (Shallenberger 1981). There are anecdotal reports of marine mammal interactions in the commercial Hawaiʻi shortline fishery that sets gear at Cross Seamount and possibly around the main Hawaiian Islands. The commercial shortline fishery is licensed to sell catch through the State of Hawai'i Commercial Marine License program, and until recently, no reporting systems existed to document marine mammal interactions. Baird and Gorgone (2005) documented high rates of dorsal fin disfigurements consistent with injuries from unidentified fishing line for MHI insular stock false killer whales. Evaluation of additional individuals with dorsal fin injuries and disfigurements suggests that the interaction rate between false killer whales and various forms of hook and line gear may vary by population and social cluster, with the highest rates in the MHI insular stock (Baird *et al.* 2014). The commercial or recreational fishery or fisheries responsible for these injuries is unknown, though through examination of satellite telemetry dataset and commercial logbook effort data, it is clear that there are regions where such interactions are far more likely, including the Kohala coast of Hawaiʻi Island and the waters extending from the southeast end of Oʻahu around to the north side of Maui and to the southwest side of Lānaʻi (Baird *et al.* 2021). A stranded MHI insular false killer whale in October 2013 had five fishing hooks and fishing line in its stomach and another stranded animal in September 2016 had one fishing hook in its stomach (Bradford & Lyman 2018). Although the fishing gear is not believed to have caused the death of either whale, examinations confirm that MHI insular false killer whales consume previously hooked fish or are interacting with MHI hook and line fisheries. Many of the hooks within the whale's stomach were not consistent with those currently allowed for use within the commercial longline fisheries and could originate from a variety of nearshore fisheries. No estimates of human-caused mortality or serious injury are currently available for nearshore hook and line or other fisheries because these fisheries are not monitored for protected species bycatch.

Because of high rates of false killer whale mortality and serious injury in Hawaiʻi-based longline fisheries, a Take Reduction Team was established in January 2010 (75 FR 2853, 19 January, 2010). The Team was charged with developing recommendations to reduce incidental mortality and serious injury of the Hawaiʻi pelagic, MHI insular and Palmyra stocks of false killer whales in Hawaiʻi-based longline fisheries. The Team submitted a draft Take Reduction Plan (TRP) to NMFS, and NMFS published a final TRP based on the Team's recommendations (77 FR 71260, 29 November, 2012). Take reduction measures include gear requirements, time-area closures (the Southern Exclusion Zone, or SEZ), and measures to improve captain and crew response to hooked and entangled false killer whales. The seasonal contraction of the Longline Exclusion Zone (LLEZ) around the MHI was also eliminated. The TRP became effective December 31, 2012, with gear requirements effective February 27, 2013. Adjustments to bycatch estimation methods were implemented for 2013 to account for changes in fishing gear and captain training intended to reduce the false killer whale serious injury rate (McCracken 2015).

There are two distinct longline fisheries based in Hawaiʻi: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas, but are prohibited from operating within the Papahānaumokuākea Marine National Monument (PMNM) and within the LLEZ around the main Hawaiian Islands and the Pacific Remote Islands and Atolls (PRIA) MNM around Johnston Atoll. The PMNM originally included the waters within a 50 nmi radius around the NWHI. ~~As of~~In August, 2016, the PMNM area was expanded to extend to the 200 nmi EEZ boundary west of 163° W. Between 2017~~2015~~ and 2021~~2019~~, one~~no~~ false killer whales was~~were~~ observed hooked or entangled in the SSLL fishery (100% observer coverage) within the U.S. EEZ of the Hawaiian Islands, and 54~~48~~ false killer whales were observed taken in the DSLL fishery (~~18~~15-21% observer coverage) within the Hawaiian Islands EEZ ~~waters~~ or adjacent high-seas waters (Bradford ~~2018a,~~ 2018~~b~~, 2020, 2021, ~~in press~~2023, in review) (Figure 3). The severity of injuries resulting from interactions with longline gear is based on an evaluation of the observer's description of each interaction and follows the most recently developed criteria for assessing serious injury in marine mammals (NMFS ~~2012~~ 2023b). In the DSLL fishery, 18~~13~~ false killer whales were taken within the Hawaiian Islands EEZ, including one within the overlap area of the pelagic and NWHI stocks. Stock identity is not known for any of the whales taken within the EEZ, though those outside of the stock overlap area are assumed to be Hawaiʻi pelagic stock animals. Of the 18 whales, 2 were found dead, 14 ~~Eleven of the 13 whales~~ were considered seriously injured, and 2~~two~~ non-seriously injured based on the information provided by the observer. Outside of the Hawaiian Islands EEZ, 5~~five~~ whales were observed dead, 23~~21~~ were considered seriously injured, 6~~seven~~ were considered not seriously injured, and 2~~two~~ had injuries with a severity that could not be determined based on the information provided by the observer. Two~~Three~~ additional unidentified "blackfish" (unidentified cetaceans known to be either false killer whales or short-finned pilot whales) were also taken ~~within~~ in the DSLL fishery outside of the Hawaiian Islands EEZ, with one considered seriously injured and~~,~~ one not

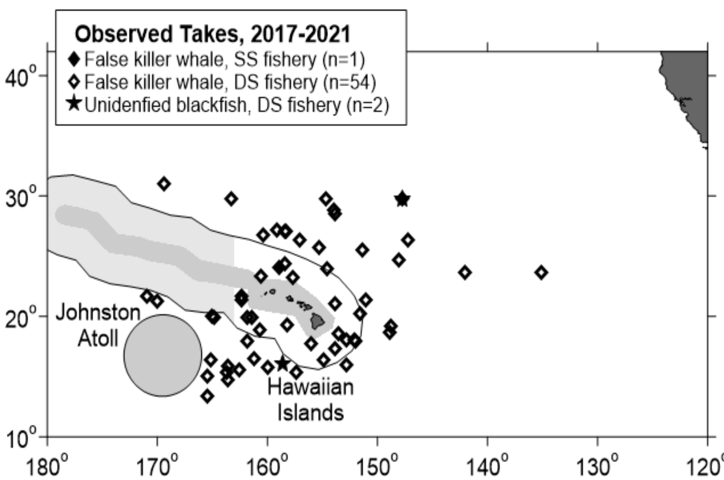

Figure 3. Locations of observed false killer whale takes within the shallow-set fishery (filled diamond), deep-set fishery (open diamonds), ~~(black symbols)~~ and possible takes (blackfish) of this species (~~open symbols~~closed stars) in the Hawaii-based longline fisheries, 2017-2021~~2015-2019~~. Some take locations overlap. Stock boundaries for false killer whales are not shown. ~~One take occurred inside the NWHI pelagic stock overlap area.~~ Solid lines represent the U.S. EEZ. Gray shading notes areas closed to ~~commercial fishing, with the PMNM Expansion area closed since August 2016~~longline fishing.

seriously injured~~, and one that could not be determined based on the information provided by the observer~~. The SEZ, a large triggered closure area south of the MHI implemented under the TRP, was closed following the trigger of 2 serious injuries within the Hawaiian Islands EEZ in November 2018. This closure remained in effect through the remainder of calendar year 2018. Following re-opening of the SEZ on January 1, 2019, the SEZ was again closed in February 2019 following a serious injury and a mortality~~two serious injuries~~ within the Hawaiian Islands EEZ. Following the closure there were 3 additional serious injuries within the Hawaiian Islands EEZ in 2019. The SEZ remained closed until August 2020. Following an increase in the trigger to 4 whales in 2021, an SEZ closure was triggered after a fourth and fifth serious injuries within the EEZ were reported in December 2021, though SEZ was not closed given the closure would not have been effective until after the automatic reopening date at the start of the new calendar year ~~the timing of the serious injuries~~.

The total estimated number of dead or seriously injured whales is calculated based on observer coverage rate, the location of the observed take (inside or outside of the EEZ), and the ratio of observed dead and seriously injured whales versus those judged to be not seriously injured. Observer coverage is measured on a per-trip basis throughout the calendar year as described by McCracken (2019). In years with large fluctuations in observer coverage, such as during the early days of the COVID-19 pandemic when observer coverage dropped to less than 10% during the second quarter of the year, the annual bycatch estimation process may be subset into several periods, as described in McCracken & Cooper (2022a). Prior to the implementation of the FKW TRP, for the period 2008 to 2012, the rate of dead and seriously injured false killer whales was 93% (McCracken 2014). The implementation of weak hooks under the TRP was intended to reduce the serious injury rate in the deep-set fishery, and as such the proportion of dead and

Table 1. Summary of available information on incidental mortality and serious injury (MSI) of false killer whales (FKW) and unidentified blackfish (UB, false killer whale or short-finned pilot whale) in commercial longline fisheries, by stock and EEZ area, as applicable (McCracken & Cooper 2022b, 2023~~2020~~). ~~5-yr m~~Mean annual takes are presented for 2017-2021~~2015-2019~~. Information on observed takes (T) and ~~combined mortality and serious injury~~ MSI is included. ~~Unidentified blackfish~~ UB are pro-rated as either FKW~~false killer whales~~ or short-finned pilot whales based on distance from shore (McCracken 2010). CVs are estimated based on the combined variances of annual FKW~~false killer whale~~ and UB~~blackfish~~ take estimates and the relative density estimates for each stock within the overlap zones. Values of '0' presented with no further precision are based on observation at 100% coverage and are not estimates.

Fishery Name	Year	Data Type	Percent Observer Coverage	Observed takes: FKW T/MSI, UB T/MSI – Outside U.S EEZ	Observed takes: FKW T/MSI, UB T/MSI – Within Hawaiian Islands EEZ	Estimated M&SI (CV): Hawaiʻi Pelagic Stock – Management Area Outside U.S EEZ	Estimated M&SI (CV): Hawaiʻi Pelagic Stock – Management Area Within Hawaiian Islands EEZ	Estimated M&SI (CV): MHI insular Stock	Estimated M&SI (CV): NWHI Stock
Hawaii-based deep-set longline fishery	~~2015~~	Observer data	~~21%~~	~~5/4~~ ~~1/1~~‡	~~0~~ ~~0~~	~~22.3 (0.4)~~	~~0 (-)~~	~~0 (-)~~	~~0 (-)~~
	~~2016~~		~~20%~~	~~9/8~~‡ ~~0~~	~~1/1~~ ~~0~~	~~27.9 (0.3)~~	~~4.0 (0.8)~~	~~0 (0.8)~~	~~0 (2.1)~~
	2017		20%	~~4~~6/~~4~~5† 0	2/1 0	29.7~~28.5~~ (0.4)	8.4 (0.7) ~~8.1 (0.6)~~	0.1 (0.8~~0.6~~)	0 (2.1~~2.0~~)
	2018		18%	8/5 1/1~~0~~	4/4 0	30.6~~29.7~~ (0.4)	12.3 (0.5) ~~11.9 (0.4)~~	0.1 (0.6~~0.5~~)	0 (2.0)
	2019		21%	9/7 1/0	6/5 0	38.7~~37.2~~ (0.3)	26.0 (0.4) ~~25.0 (0.4)~~	0.0 (0.5)	0.6 (2.0)
	2020		15%	3/2† 0	1/1 0	14.2 (0.5)	5.1 (0.9)	0.0 (0.9)	0 (2.2)
	2021		18%	10/9† 0	5/5 0	37.0 (0.4)	32.1 (0.4)	0.2 (0.5)	0.1 (2.0)
Mean Estimated Annual Take (CV) 2017-2021~~2015-2019~~						**30.0~~28.8~~ (0.2)**	**16.8 (0.2) ~~9.8 (0.3)~~**	**0.1 (0.3) ~~0.03 (0.4)~~**	**0.2 (1.6) ~~0.1 (1.8)~~**
Hawaii-based shallow-set longline fishery	~~2015~~	Observer data	~~100%~~	~~0~~ ~~0~~	~~0~~ ~~0~~	~~0~~	~~0~~	~~0~~	~~0~~
	~~2016~~		~~100%~~	~~0~~ ~~0~~	~~0~~ ~~0~~	~~0~~	~~0~~	~~0~~	~~0~~

	2017		100%	0 0	0 0	0	0	0	0
	2018		100%	0 0	0 0	0	0	0	0
	2019		100%	0 0	0 0	0	0	0	0
	2020		100%	0 0	1/1 0	0	1	0	0
	2021		100%	0 0	0 0	0	0	0	0
Mean Annual Takes (100% coverage) 2017-2021~~2015-2019~~						**0**	**0.2~~0~~**	**0**	**0**
Minimum total annual takes within U.S EEZ (2017-2021~~2015-2019~~)							**17.0 (0.2) ~~9.8 (0.3)~~**	**0.1 (0.3) ~~0.03 (0.4)~~**	**0.2 (1.6) ~~0.1 (1.8)~~**

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

seriously injured whales versus non-serious injuries is calculated annually based on the injury status of observed takes since the implementation of the TRP in 2013 (McCracken 2019).

Complete assessment of human-caused mortality within the full Hawai‘i pelagic false killer whale management area requires information on bycatch in foreign fleet fishing operations. Foreign longline fleets operate within the tropical Pacific including immediately outside of the Hawaiian Islands EEZ. Although the magnitude of foreign longline effort near the Hawaiian Islands EEZ is thought to be relatively low compared to that of the Hawai‘i-based fleet, there is considerable effort to the southwest of the EEZ and north of 30^0 on the east side of the islands (based on Global Fishing Watch data). The Western and Central Pacific Fisheries Commission (WCPFC) has collated 76 interactions with false killer whales in the western and central Pacific across the member fleets including reports from the Hawai‘i-based vessels from 2015 to 2020 (Williams et al. 2021). However, the WCPFC has not developed estimates of total bycatch for any segement of the fleet “given the low levels and imbalanced nature of observer coverage” (Peatman and Nicols 2020). Commercial fishing within the eastern portion of the Hawai‘i pealgic false killer whale management area is managed by the Inter-American Tropical Tuna Commission, though similar concerns about observer coverage have so far precluded any bycatch estimates for false killer whales in this region. The mortality rate of bycaught animals in foreign longline fleets may also be higher than in the U.S fleet given the bycatch mitigation measures in place for the Hawai‘i-based fleet, leading to additional uncertainty in the magnitude of the impact on the stock (Oleson *et al.* 2023).

Biological samples or individual animal photographs are required to assign a take to a specific stock. Very few observed takes are identified to stock, as collection of such information is very rare. The pelagic stock is known to interact with longline fisheries based on a small number of genetic samples obtained by fishery observers (e.g. Chivers *et al.* 2010) both inside and outside of the Hawaiian Islands EEZ. No samples or photographs have been collected that may be used to conclusively assign takes to stock within the MHI insular-pelagic overlap zone or the NWHI-pelagic stock overlap zone. However, MHI insular and NWHI false killer whales have been documented via telemetry to move far enough offshore to reach longline fishing areas (Bradford *et al.* 2015), and MHI insular stock animals have high rates of dorsal fin disfigurements consistent with injuries from unidentified fishing line (Baird & Gorgone 2005, Baird *et al.* 2014). When takes cannot be assigned to stock, annual bycatch estimates are prorated to stock using the following process. Takes of unidentified blackfish are prorated to false killer whale and short-finned pilot whale based on distance from shore (McCracken 2010), given patterns of previous bycatch for each species. Following proration of unidentified blackfish takes to species, Hawaiian Islands EEZ and high-seas estimates of false killer whale take are calculated by summing the annual false killer whale take and the annual blackfish take prorated as false killer whale within each region (McCracken 2020). Takes within the shallow-set longline fishery are assigned to the stock area in which they were observed. Estimated takes in the deep-set fishery within the Hawaiian Islands EEZ are apportioned to each stock area by first allocating take to each area based on relative annual fishing effort (by set) in that area. If an observed take occurred within the MHI-pelagic or NWHI-pelagic overlap zones, the take was assigned to that zone and the remaining estimated bycatch was assigned to stock areas as previously described. For both the shallow-set and deep-set fisheries, stock area bycatch estimates are then multiplied by the relative density of each stock within the stock area to estimate stock-specific bycatch for each year. Uncertainty in stock-specific bycatch estimates combines variances of total annual false killer whale bycatch and the fractional variance of false killer whale density according to which stock is being estimated. Enumeration of fishing effort within stock overlap zones is assumed to be known without error. Proration of unidentified blackfish takes and of false killer whale takes within the stock overlap zones introduces unquantified uncertainty into the bycatch estimates, but until methods of determining stock identity for animals observed taken within the overlap zone are available, and all animals taken can be identified to species and stock (e.g., with photos or tissue samples), these proration approaches are needed to ensure that potential impacts to all stocks are assessed in the overlap zones. Based on this approach, estimates of annual mortality and serious injury of false killer whales, by stock and EEZ area are shown in Table 1.

MAIN HAWAIIAN ISLANDS INSULAR STOCK

POPULATION SIZE

Bradford *et al.* (2018) used encounter data from dedicated and opportunistic surveys for MHI insular false killer whales from 2000 to 2015 to generate annual mark-recapture estimates of abundance. Due to spatiotemporal biases imposed by sampling constraints, annual estimates reflected the abundance of MHI insular false killer whales within the surveyed area in that year, and therefore c~~sh~~ould not be considered indicative of total population size every year. A new Bayesian pseudo~~-~~spatial analysis framework developed by Badger *et al.* (in review-a) used available telemetry data to develop utilization distributions for each MHI insular social cluster and then adjust abundance estimates based on the degree of overlap between survey effort~~s~~ and the distribution of each cluster. Based on this new approach, annual abundance estimates were derived for the full stock range for 1999 to 2021 (Badger *et al.* in review-b). The abundance estimate for 2021 was 138 (CV=0.08) false killer whales. Annual estimates over the 23-year survey period ranged from 136 to 210 animals. ~~The abundance estimate for 2015 was 167 (CV = 0.14). Annual estimates over the 16 year survey period ranged from 144 to 187 animals and are similar to multi-year aggregated estimates published previously (Oleson *et al.* 2010).~~

Minimum Population Estimate

The minimum population estimate for the MHI insular stock of false killer whales is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2021~~2015~~ abundance estimate (from Badger *et al.* in review-b~~Bradford *et al.* 2018~~), or 129~~149~~ false killer whales.

Current Population Trend

Using stock-wide annual abundance estimates for 1999 to 2021, Badger *et al.* (in review-b) evaluated the trend of the MHI insular stock and found an annual average decline of -3.51% (95% credible intervals -8.40 to +2.04) over the entire time series, and -5.53% (95% credible intervals -9.91 to -1.61) for the last 10 years (2011-2021). This decline appears to be attributed largely to social clusters 3 and 4, with some evidence that cluster 1 is increasing. The overall results generally align with previous analyses that have indicated a history of stock decline in recent decades. Reeves *et al.* (2009) suggested that the MHI insular stock of false killer whales may have declined between 1989 and 2007, based on sightings data collected near Hawai‘i using various methods. Baird (2009) reviewed trends in sighting rates of false killer whales from aerial surveys conducted using consistent methodology around the main Hawaiian Islands between 1994 and 2003 (Mobley *et al.* 2000). Sighting rates during these surveys showed a statistically significant decline that could not be attributed to any weather or methodological changes. The Status Review of MHI insular false killer whales (Oleson *et al.* 2010) presented a quantitative analysis of extinction risk using a Population Viability Analysis (PVA). The modeling exercise was conducted to evaluate the probability of actual or near extinction, defined as a population reduced to fewer than 20 animals, given measured, estimated, or inferred information on population size and trends, and varying impacts of catastrophes, environmental stochasticity and Allee effects. All plausible models indicated the probability of decline to fewer than 20 animals within 75 years was greater than 20%. Though causation was not evaluated, all plausible models indicated the population had declined since 1989, at an average rate of -9% per year (95% probability intervals -5% to -12.5%), though some two-stage models suggested a lower rate of decline (Oleson *et al.* 2010).~~Annual abundance estimates in Bradford *et al.* (2018) are not appropriate for evaluating population trends, as the study area varied by year, and each annual estimate represents only animals present in the study area within each year.~~

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the MHI insular false killer whale stock is calculated as the minimum population estimate (132~~149~~) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.1 (for a stock listed as Endangered under the ESA and with minimum population size less than 1500 individuals; Taylor *et al.* 2000) resulting in a PBR of 0.26~~0.3~~ false killer whales per year, or approximately one animal every 4~~3.3~~ years.

STATUS OF STOCK

The status of MHI insular stock of false killer whales relative to OSP is unknown, although this stock appears to have declined during the past four~~two~~ decades (Badger *et al.* in review-b, Oleson *et al.* 2010, Reeves *et al.* 2009; Baird 2009). MHI insular false killer whales are listed as "endangered" under the Endangered Species Act (1973) (77 FR 70915, 28 November, 2012). The Status Review report produced by the Biological Review Team (BRT) (Oleson *et al.* 2010, amended in Oleson *et al.* 2012) found that Hawaiian insular false killer whales are a Distinct Population Segment (DPS) of the global false killer whale taxon. Of the 29 identified threats to the population, the BRT considered the effects of small population size, including inbreeding depression and Allee effects, exposure to environmental contaminants (Ylitalo *et al.* 2009), competition for food with commercial fisheries (Boggs & Ito, 1993, Reeves *et al.* 2009), and hooking, entanglement, or intentional harm by fishermen to be the most substantial threats to the population. Because MHI insular false killer whales are formally listed as "endangered" under the ESA, they are automatically considered as a "depleted" and "strategic" stock under the MMPA. For the 5-yr period prior to the implementation of the TRP, the average estimated mortality and serious injury to MHI insular stock false killer whales (0.21 animals per year) exceeded the PBR (0.18 animals per year. Prior to the TRP, a seasonal contraction to the LLEZ potentially exposed a significant portion of the offshore range of the stock to longline fishing. Following implementation of the TRP, a significant portion of the recognized stock range is inside of the expanded year-round LLEZ around the MHI, providing significant protection for this stock from longline fishing. For the most recent 5-yr period, the estimate of mortality and serious injury (0.1~~0.03~~) is below the PBR (0.26~~0.30~~). The total fishery mortality and serious injury for the MHI insular stock of false killer whales cannot be considered to be insignificant and approaching zero, as it is ≥ 10% of PBR. The stock is declining (Badger *et al* in review-b), though the cause of that decline has not been thoroughly assessed.~~Effects of other threats have yet to be assessed, e.g., nearshore hook and line fishing and environmental contamination~~.

OTHER FACTORS THAT MAY BE CAUSING A DECLINE OR IMPEDING RECOVERY

There is significant geographic overlap between various nearshore fisheries and evidence of interactions with hook-and-line gear (Baird *et al.* 2015, 2021), such that these fisheries may pose a threat to the stock. Six~~Five~~ MHI insular false killer whales stranded between 2010 and 2021~~2016~~, including 4 from cluster 3 (Baird *et al.* 2023~~PIRO MMRN~~), a high rate for a single social cluster. High concentrations of polychlorinated biphenyls (PCBs) exceeding those proposed to cause adverse health effects (Kannan *et al.* 2020) were measured from 29 of 41 sampled individuals in the MHI insular stock (Kratofil *et al.* 2020). PCB concentrations from four stranded individuals within this population all revealed levels more than twice the highest suggested health threshold for PCBs, and had the highest levels of any sampled whales in the study. Differences in contaminant loads for various contaminant classes are also evident among social clusters suggesting differences in exposure or consumption of contaminated prey based on preferred foraging regions (Kratofil *et al.* 2020).

HAWAII PELAGIC STOCK

POPULATION SIZE

Encounter data from shipboard line-transect surveys conducted throughout the central Pacific were used to estimate the density and abundance of pelagic false killer whales~~ across the central Pacific, including within the Hawaiian Islands EEZ~~ (Bradford *et al*. 2020; Table 2). Density data from this larger modeled area were used to extract abundance within the Hawai'i pelagic false killer whale management area (Oleson *et al.* 2023), and within the Hawaiian Islands EEZ portion of the management area. The abundance of Hawai'i pelagic false killer whales in the full Management Area is 5,528 (CV = 0.35), and within the EEZ portion of the managment area is 2,038 (CV = 0.35).

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

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

Bradford *et al*. (2020) ~~also~~ produced design- and model-based abundance estimates for false killer whales within each survey year for the full Hawaiian Islands EEZ (prior to the delineation of the management area). ~~, and these can be used as a point of comparison to model-based estimates.~~ While on average, the estimates are broadly similar between the two approaches, annual design-based estimates show much greater variability between years than the model-based estimates (Figure 4). The model-based approach reduces variability through explicit examination of habitat relationships across the full dataset, while the design-based approach evaluates encounter data for each year separately and thus is more susceptible to the effects of encounter rate variation. Bradford *et al*. (2020) found through simulation that the low sighting rate in 2002 and high sighting rate in 2017 could be explained by encounter rate variation. Although a 'year' covariate was tested during model development, it was not selected as a significant variable. Despite not fully accounting for inter-annual variation in total abundance, the model-based estimates are considered the best approach.~~available estimate for each survey year. Current model-based estimates for the central Pacific include animals that are considered part of the Palmyra Atoll stock, as well as animals that may be part of an eastern Pacific stock on the eastern edge of the modeled area, and therefore are likely an overestimate of the full Hawaii pelagic stock abundance.~~ Previous abundance estimates from the Hawaiian Islands EEZ and central Pacific (e.g., Barlow 2006, Barlow & Rankin 2007, Becker *et al.* 2012, Bradford *et al.* 2014, Forney *et al.* 2015) used~~ing~~ subsets of the ~~full~~ dataset and different line-transect parameters than those used by Bradford *et al.* 2020, such that these estimates have been superseded by the estimates presented here~~have been published previously. The estimate of 2,086 (CV=0.35) from the 2017 survey is considered the best available current estimate for false killer whales in the Hawaiian Islands EEZ (Bradford *et al*. 2020)~~.

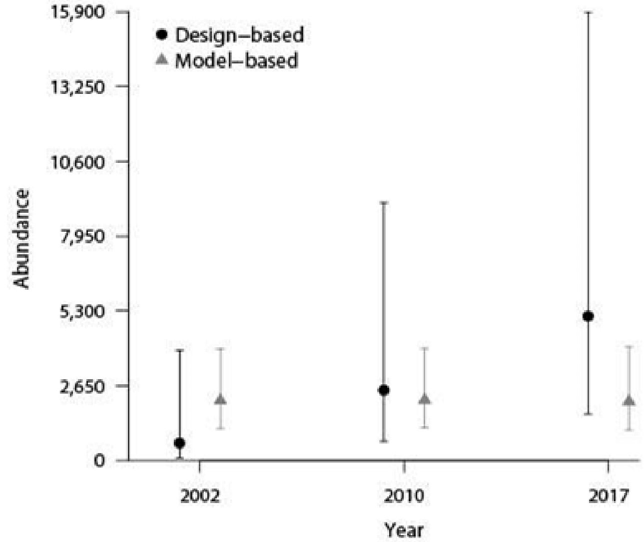

Figure 4. Comparison of design-based (circles) and model-based (triangles) (Bradford *et al*. 2020) estimates of abundance for false killer whales for each survey year (2002, 2010, 2017).

The reanalysis may still be subject to potential bias due to vessel attraction as described by Bradford *et al.* (2014), who reported that most (64%) false killer whale groups seen during the 2010 survey were seen moving toward the vessel when detected by the visual observers. Together with an increase in sightings close to the trackline, these behavioral data suggest vessel attraction is likely occurring and may be significant. Similar to the treatment of the detection function in Bradford *et al.* (2014, 2015), new model-based estimates use Beaufort-specific effective strip width estimates (following Barlow *et al*. 2015) derived from an analysis that used a half-normal model to minimize the effect of vessel attraction. The abundance estimate may still be positively biased due to vessel attraction because groups originally outside of the survey strip, and therefore unavailable for observation by the visual survey team, may have moved within the survey strip and been sighted. The acoustic data and visual data suggests vessel attraction (Bradford *et al.* 2014), though the extent of any bias created by this movement is unknown.

Minimum Population Estimate

The minimum population estimate for the Hawaiʻi pelagic stock of false killer whales~~size~~ is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2017 model-based abundance estimate. For the full management area the minimum population estimate is 4,152 Hawaiʻi pelagic false killer whales and for the Hawaiian Islands EEZ portion of the management area is 1,531 Hawaiʻi pelagic false killer whales ~~(Bradford *et al*. 2020), or 1,567 false killer whales. For the entire central Pacific study area, the minimum population size for 2017 is estimated to be 25,940 false killer whales.~~

Current Population Trend

Although a 'year' covariate was evaluated during model development and not included during the model selection process, the final model-based abundance estimates for false killer whales provided by Bradford *et al.* (2020) do not explicitly examine population trend other than that driven by environmental factors. In contrast, annual design-based estimates suggest an increase in population size within the Hawaiian Islands EEZ, however, these changes can be largely explained by random variability in encounter rate common for species like false killer whales with low density and patchy distribution. Examination of population trend for the Hawaii pelagic stock of false killer whales requires additional data inside and outside of the Hawaiian Islands EEZ.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

Within the Hawai‘i pelagic false killer whale management area t~~T~~he potential biological removal (PBR) level ~~for the Hawaii pelagic stock of false killer whales~~ is calculated as the minimum population estimate for the management area (4,152) ~~U.S. EEZ of the Hawaiian Islands (1,567)~~ times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.40~~0.50~~ (for a stock of unknown status and uncertain serious injury rate~~with a Hawaiian Islands EEZ mortality and serious injury rate CV <= 0.30~~; Wade & Angliss 1997), resulting in a PBR of 33~~16~~ false killer whales per year. Within the EEZ portion of the Hawai‘i pelagic false killer whale management area the potential biological removal (PBR) level is calculated as the minimum population estimate within the EEZ (1,531) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a stock of unknown status with a Hawaiian Islands EEZ mortality and serious injury rate CV <= 0.30; Wade & Angliss 1997), resulting in a PBR of 15 false killer whales per year. ~~For the entire central Pacific, based on the minimum population size of 25,940 false killer whales, and using the same recovery factor and maximum net growth rate as for the Hawaii pelagic stock, would yield a PBR of 259 false killer whales per year.~~

STATUS OF STOCK

The status of the Hawai‘i pelagic stock of false killer whales relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. ~~The mean concentration of polychlorinated biphenyls (PCBs) in all Hawaii false killer whale populations, including individuals from the pelagic stock (Kratofil *et al.* 2020) has been shown to exceed the level proposed to cause adverse health effects in other cetaceans (Kannan *et al.* 2020).~~ This stock is not listed as “threatened” or “endangered” under the Endangered Species Act (1973), nor designated as “depleted” under the MMPA. Following the NMFS Guidelines for Assessing Marine Mammal Stocks (NMFS ~~2005~~2023a), the status of this transboundary stock of false killer whales is assessed based on delineation of a management area defined using all available biological data for this population~~the estimated abundance and mortality and serious injury within the U.S. EEZ of the Hawaiian Islands because estimates of human-caused mortality and serious injury from all U.S. and non-U.S. sources in high seas waters are not available~~. The estimated mortality and serious injury within the ~~Hawaii EEZ~~ the managment area in ~~2018and~~ 2019 and 2021 were the highest recorded since before the TRP was implemented, with annual M&SI rates exceeding 60 animals per year~~, with the estimated take in 2019 more than double that in 2018~~. Take rates of false killer whales by the deep-set longline fishery outside of the EEZ continue to remain significantly higher since the TRP. ~~Model-based estimates of abundance and PBR for the central Pacific should be considered when evaluating stock status across the fishery area.~~ Total 5-year mortality and serious injury for 2017-2021~~2015-2019~~ across the management area (47~~9.8~~) is more~~less~~ than PBR (33~~16~~), therefore this stock is ~~not~~ considered a “strategic stock” under the MMPA. Total fishery mortality and serious injury for the Hawaii pelagic stock of false killer whales cannot be considered to be insignificant and approaching zero (i.e. less than 10% of PBR, or 1.6 animals per year).

OTHER FACTORS THAT MAY BE CAUSING A DECLINE OR IMPEDING RECOVERY

The mean concentration of polychlorinated biphenyls (PCBs) in all Hawai‘i false killer whale populations, including individuals from the pelagic stock (Kratofil *et al.* 2020) has been shown to exceed the level proposed to cause adverse health effects in other cetaceans (Kannan *et al.* 2020).

NORTHWESTERN HAWAIIAN ISLANDS STOCK

POPULATION SIZE

Encounter data from shipboard line-transect surveys of the entire Hawaiian Islands EEZ were reevaluated for each survey year, resulting in the following abundance estimates of Northwestern Hawaiian Islands false killer whales (Bradford *et al*. 2020; Table 3).

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

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

The updated design-based abundance estimates use sighting data from throughout the central Pacific to estimate the detection function and use Beaufort sea-state-specific trackline detection probabilities for false killer whales using the methods of Barlow *et al*. (2015). Although a previous 2010 estimate for this stock was published using a subset of this data (Bradford et al. 2014), Bradford *et al*. (2020)~~,~~ uses a consistent approach for estimating all abundance parameters and resulting estimates are considered the best available. There were no sightings of false killer whales in the NWHI stock area in 2002. The reanalysis may still be subject to potential bias due to vessel attraction as described by Bradford *et al.* (2014), who reported that most (64%) false killer whale groups seen during the 2010 HICEAS survey were seen moving toward the vessel when detected by the visual observers. Together with an increase in sightings close to the trackline, these behavioral data suggest vessel attraction is likely occurring and may be significant. Bradford *et al.* (2014, 2015, 2020) used a half-normal model to minimize the effect of vessel attraction, because groups originally outside of the survey strip, and therefore unavailable for observation by the visual survey team, may have moved within the survey strip and been sighted. There is some suggestion of such attractive movement within the acoustic and visual data (Bradford *et al.* 2014), though the extent of any bias created by this movement is unknown. The best estimate of current abundance is 477 (CV=1.71) false killer whales from the 2017 survey (Bradford *et al*. 2020).

Minimum Population Estimate

The minimum population size is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2017 abundance estimate for the Northwestern Hawaiian Islands stock (Bradford *et al.* 2020), or 178 false killer whales. ~~This estimate has not been corrected for vessel attraction and may be positively biased.~~

Current Population Trend

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

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Northwestern Hawaiian Islands false killer whale stock is calculated as the minimum population estimate (178) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.40 (for a stock of unknown status, with a Hawaiian Islands EEZ mortality and serious injury rate CV > 0.8; Wade & Angliss 1997), resulting in a PBR of 1.43 false killer whales per year.

STATUS OF STOCK

The status of false killer whales in Northwestern Hawaiian Islands waters relative to OSP is unknown, and insufficient data exists to evaluate abundance trends. The mean concentration of polychlorinated biphenyls (PCBs) in all Hawaii false killer whale populations (Kratofil *et al.* 2020), including individuals from the NWHI stock, has been shown to exceed the level proposed to cause adverse health effects in other cetaceans (Kannan *et al.* 2020). Biomass of some false killer whale prey species may have declined around the Northwestern Hawaiian Islands (Oleson *et al.* 2010, Boggs & Ito 1993, Reeves *et al.* 2009), though waters within the original PMNM have been closed to commercial longlining since 1991 and to other fishing since 2006. This stock is not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor as "depleted" under the MMPA. The rate of mortality and serious injury to NWHI false killer whales (0.16~~0.1~~) is less than the PBR (1.43 animals per year), though ~~and~~ cannot be considered to be insignificant and approaching zero (<10% of PBR). A very small portion of the recognized stock range lies outside of the newly expanded PMNM and the expanded LLEZ, such that this stock is likely not exposed to high levels of fishing effort because commercial and recreational fishing is prohibited within Monument waters and longlines are excluded from the majority of the stock range.

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Revised ~~5/5/2021~~ 08/16/2023

SHORT-FINNED PILOT WHALE (*Globicephala macrorhynchus*): Hawaiʻi Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Short-finned pilot whales are found in all oceans, primarily in tropical and warm-temperate waters. They are commonly sighted during periodic ~~Summer/fall~~ shipboard surveys of the waters within the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands, ~~resulted in 25 sightings in 2002, 36 in 2010, and 35 in 2017~~ including a high~~er~~ frequency of encounters nearshore within the Northwestern Hawaiian Islands (Figure 1~~; Barlow 2006, Bradford et al. 2017, Yano *et al.* 2018~~).

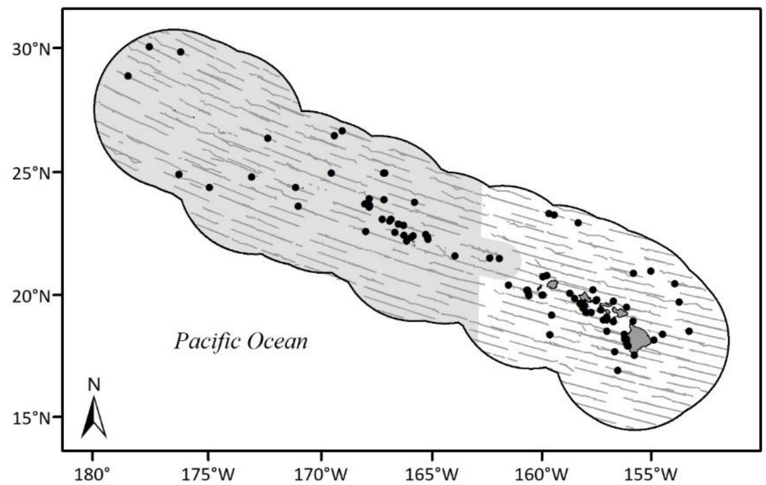

Figure 1. Short-finned pilot whale sighting locations (circles) and survey effort (~~light~~ gray lines) during the 2002 (~~diamonds~~Barlow 2006), 2010 (~~circles~~Bradford *et al.* 2017), and 2017 (~~squares~~Yano *et al.* 2018) shipboard surveys of the U.S. EEZ ~~waters surrounding~~around the Hawaiian Islands (~~Barlow 2006, Bradford et al. 2017, Yano *et al.* 2018~~outer black line). ~~Outer solid line represents approximate boundary of survey area and U.S. EEZ. Gray shading indicates area of t~~The Papahānaumokuākea Marine National Monument in the western portion of the EEZ is shaded gray. ~~with the lighter gray shading denoting the full 2016 Expansion area. Dotted line represents the 1000 m isobath.~~

Two forms of short-finned pilot whales have been identified in Japanese waters based on pigmentation patterns and differences in the shape of the heads of adult males (Kasuya *et al.* 1988). Genetic analysis of samples from throughout the global range of short-finned pilot whales suggest three types within the species, an Atlantic type, a western/central pacific and Indian Ocean (Naisa) type, and an eastern tropical Pacific and northern Japan (Shiho) type. Significant differentiation in mtDNA control region sequences further suggest that the three forms represent two subspecies, the shiho short-finned pilot whale ~~an~~ and the naisa short-finned pilot whale, with evidence of further divergence among the naisa types in the Atlantic and Pacific (Van Cise *et al.* 2019). The pilot whales in Hawaiian waters are of the naisa type. The shiho and naisa forms appear also to be distinguishable based on the acoustic features of their whistle and burst-pulse sounds, providing further evidence for divergence between these subspecies (Van Cise *et al.* 2017b).

Photo-identification, telemetry, acoustic, and genetic studies suggest that at least two demographically-independent populations of short-finned pilot whales reside in Hawaiian waters. Resighting and social network analyses of individuals photographed off Hawaii Island suggest the occurrence of one large and several smaller social clusters that use those waters, with some individuals within the smaller social clusters commonly resighted off Hawaiʻi Island (Mahaffy *et al.* 2015). Further, two groups of 14 individuals have been seen off~~at~~ Hawaiʻi Island and elsewhere in the main Hawaiian Islands, one off Oʻahu and the other off Kauaʻi, indicating some degree of connectivity within the main Hawaiian Islands (MHI). Satellite telemetry data from over 60 individuals tagged throughout the MHI~~main Hawaiian Islands~~ also support the occurrence of at least two populations (Baird 2016, Oleson *et al.* 2013). An assessment of foraging hotspots off Hawaiʻi Island revealed tight association between satellite-tagged short-finned pilot whales and the 1000-2500m depth range (Abecassis *et al.* 2015). Further, ~~More recently,~~ Van Cise *et al.* (2017a) used nuclear SNPs to assess population structure within ~~Hawaii~~ short-finned pilot whales around the Hawaiian Archipelago and found evidence for an island-associated population in the MHI. Although there was some support for separation of short-finned pilot whales in the northwestern Hawaiian Islands (NWHI) from other pelagic animals, additional genetic samples may be required to test this separation further. In addition, genetic data combined with social affiliation and habitat associations suggest the MHI population is further divided into social groups, and these groups may even rise to the level of demographic-independence between those found primarily near Hawaiʻi Island and those near Oʻahu and Kauaʻi (Van Cise *et al.* 2017a). Differences in the acoustic features of short-finned pilot whale social clusters recorded within the MHI further supports the existence of ~~several DIPs~~ demographically-independent populations within the MHI (Van Cise *et al.* 2017b). Formal assessment of demographic-independence has not been completed, but division of this population into one or more~~a separate~~ island-associated stocks may be warranted in the future.

For the Marine Mammal Protection Act (MMPA) stock assessment reports, short-finned pilot whales within the Pacific U.S. EEZ are divided into two discrete, non-contiguous areas: 1) Hawaiian waters (this report), and 2) waters off California, Oregon and Washington. The Hawaiʻi stock includes animals found both within the Hawaiian Islands EEZ and in adjacent high seas waters; however, because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of this stock is evaluated based on data from the U.S. EEZ ~~waters of~~around the Hawaiian Islands (NMFS 2005).~~. The status of the Hawaii stock is evaluated based on abundance, distribution, and human-caused impacts within the Hawaiian Islands EEZ, as such datasets are largely lacking for high seas waters (NMFS 2005).~~

POPULATION SIZE

Encounter data from shipboard line-transect surveys of the ~~entire~~ Hawaiian Islands EEZ were~~was~~ recently reevaluated, resulting in updated model-based abundance estimates of short-finned pilot whales in the entirety of the Hawaiian Islands EEZ (Becker *et al*. 2021, 2022; Table 1).

Table 1. Model-based ~~L~~line-transect abundance estimates for short-finned pilot whales in the~~derived from surveys of the entire~~ Hawaiian Islands EEZ in 2002 and~~,~~ 2010 (Becker *et al.* 2021)~~,~~ and 2017~~,~~ and 2020 (Becker *et al*. ~~2021,~~ 2022). ~~.~~derived from NMFS surveys in the central Pacific since 2000. The Becker *et al.* (2022) analysis incorporates a more comprehensive model-based approach to estimating model uncertainty, such that the CVs and 95% confidence limits for 2002/2010 and 2017/2020 are not directly comparable.

Year	Model-based Abundance	CV	95% Confidence Limits
2020	19,242	0.23	12,289-30,129
2017	17,237 ~~12,607~~	0.23 ~~0.18~~	11,009-26,989 ~~8,8263-18,008~~
2010	15,343	0.17	11,039-21,326
2002	15,198	0.17	10,900-21,191

Sighting data from 2002 to 2020~~2017~~ within the Hawaiian Islands EEZ were used to derive habitat-based models of animal density for two periods: 2002-2017 (Becker *et al.* 2021) and 2017-2020 (Becker *et al.* 2022). The most recent set of models include three notable changes from the 2002-2017 models: use of calibrated group size estimates, as in Bradford *et al.* (2021), exclusion of a spatial term on model selection, requiring more explicit reliance on environmental variables, and incorporating new approaches (Miller *et al.* 2022) for more comprehensively estimating uncertainty in model predictions that account for the combined uncertainty around all parameter estimates~~the overall period~~. The modeling framework incorporated Beaufort-specific trackline detection probabilities for short-finned pilot whales from Barlow *et al.* (2015). ~~The m~~Models were ~~then~~ used to predict density and abundance for each survey year based on the environmental conditions within that year (see Forney *et al*. 2015, Becker *et al*. 2016). ~~The modeling framework incorporated Beaufort-specific trackline detection probabilities for short-finned pilot whales from Barlow *et al.* (2015).~~ Bradford *et al*. (2021)

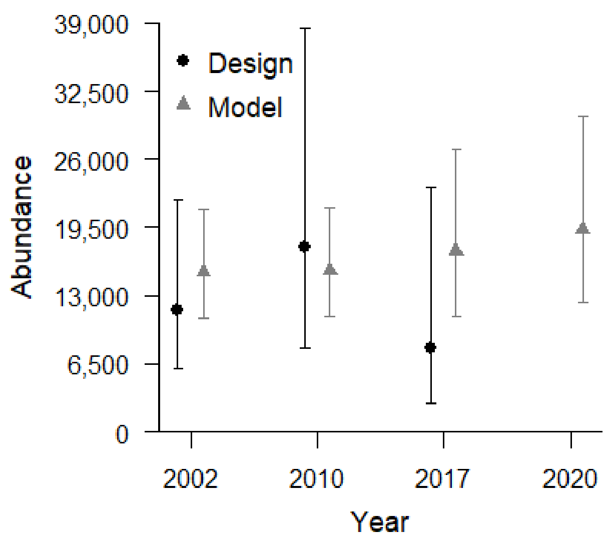

Figure 2. Comparison of design-based (black circles, Bradford *et al.* ~~in review~~2021) and model-based (gray triangles, Becker *et al*. ~~in review~~2021, 2022) estimates of abundance for short-finned pilot whales for each survey year (2002, 2010, 2017, 2020).

produced design-based abundance estimates for short-finned pilot whales in 2002, 2010, and 2017~~for each survey year~~ that can be used as a point of comparison to the model-based estimates for those years. While on average the estimates are broadly similar between the two approaches, the annual design-based estimates show much greater variability between years than do the model-based estimates (Figure 2). The model-based approach reduces variability through explicit examination of habitat relationships across the full dataset, while the design-based approach evaluates encounter data for each year separately and thus is more susceptible to the effects of encounter rate variation. Model based-estimates are based on the implicit assumption that changes in abundance are attributed to environmental variability alone. ~~There are insufficient data to e~~Explicitly incorporatinge a trend term into the model is not possible due to the insufficient sample size to test for temporal effects. Despite not fully accounting for inter-annual variation in total abundance, the model-based estimates are considered the best available estimate for each survey year. Becker *et al.* (2022) and Bradford *et al.* (2022) evaluated seasonal changes in the abundance of short-finned pilot whales within the main Hawaiian Islands using summer-fall data from 2017 and winter survey data from 2020. Although the model identified moderately lower densities of short-finned pilot whales in the MHI in winter, the design-based analysis showed a 7-fold increase in density during the same period, though confidence limits partly overlap for both analyses. The disparate results may demonstrate the impacts of encounter rate variation on the annual design-based estimates, though also suggest additional data will be needed to understand habitat relationships and seasonal movements of this species in Hawaiian waters. Previously published ~~design-based~~ abundance estimates for the Hawaiian Islands EEZ ~~from 2002 and 2010 surveys~~ (e.g. Barlow 2006, Becker *et al.* 2012, Forney *et al.* 2015, Bradford *et al*. 2017) used a subset of the dataset used by Becker *et al.* (2021, 2022) and Bradford *et al.* (2021) to derive line-transect parameters, such that these estimates have been superseded by the estimates presented here. The best estimate of abundance is based on the 2020~~2017~~ survey, or 19,242 (CV=0.23)~~12,607 (CV=0.18)~~ short-finned pilot whales.

Minimum Population Estimate

The minimum population estimate~~size~~ is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2020~~2010~~ abundance estimate ~~for the Hawaiian Islands EEZ~~ (from Becker *et al.* 2022) or 15,894~~10,847~~ short-finned pilot whales.

Current Population Trend

The model-based abundance estimates for short-finned pilot whales provided by Becker *et al.* (2021, 2022) do not explicitly allow for examination of population trend other than that driven by environmental factors. Model-based examination of short-finned pilot whale population trends including sighting data beyond the Hawaiian Islands EEZ will be required to more fully examine trend for this stock.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Hawaiʻi short-finned pilot whale stock is calculated as the minimum population estimate (15,894~~10,847~~) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.5~~0.40~~ (for a species of unknown status with no known fishery mortality within the U.S. EEZ of the Hawaiian Islands~~with a Hawaiian Islands EEZ fishery mortality and serious injury rate CV > 0.80~~; Wade and Angliss 1997), resulting in a PBR of 159~~87~~ short-finned pilot whales per year.

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Fishery Information

Information on fishery-related mortality of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaiian fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. Entanglement in gillnets and hooking or entanglement in va–rious hook and line fisheries have been reported for small cetaceans in Hawaiʻi (Nitta & Henderson, 1993). Short-finned pilot whales have been observed with fishing gear trailing from their mouths~~,~~ or have stranded with gear and other debris in their stomach, though the specific gear types have not been identified (Baird 2016, Bradford and Lyman 2018, 2019). ~~In 2014, a short-finned pilot whale was found stranded on Oahu with large amounts of debris in its stomach, including approximately 20 lbs. of fishing line, nets, and plastic drogues, though this gear was judged not to be the cause of death (Bradford and Lyman 2018).~~ In 2017, two short-finned pilot whales stranded together as part of a mass stranding event on Kauai. One of the whales had 12-15 lbs of nylon line and plastic present within its forestomach and the other had~~s~~ scarring on the upper right jaw consistent with previous fisheries interaction, though in neither case were these findings considered to be related

Table 2. Summary of available information on incidental mortality and serious injury (MSI) of short-finned pilot whales (GM~~Hawaii stock~~) and including those presumed to be short-finned pilot whales based on assignment of unidentified blackfish (UB) to this species in commercial longline fisheries, within and outside of the U.S. EEZ~~s~~ (McCracken & Cooper 2022b ~~2019~~). Mean annual takes are based on ~~2014-2018~~2017-2021 data unless otherwise indicated. Information on all observed takes (T) and ~~combined mortality events & serious injuries (~~MSI~~)~~ is included. Total takes were prorated to deaths, serious injuries, and non-serious injuries based on the observed proportions of each outcome. ~~Unidentified blackfish~~UB are prorated as either false killer whales or short-finned pilot whales according to their distance from shore (McCracken 2010). CVs are estimated based on the combination of annual short-finned pilot whale and blackfish variances and do not yet incorporate additional uncertainty introduced by prorating the unidentified blackfish.

Fishery Name	Year	Data Type	Percent Observer Coverage	~~Observed total interactions (T) and mortality events, and serious injuries (MSI), and total estimated mortality and serious injury (MSI) of short-finned pilot whales (GM)~~			
				Outside U.S. EEZ~~s~~		Hawaiian Islands EEZ	
				Observ~~.~~ed GM T/MSI Observ~~.~~ed UB T/MSI	Estimated MSI (CV)	Observ~~.~~ed GM T/MSI Observ~~.~~ed UB T/MSI	Estimated MSI (CV)
Hawaiʻi-based deep-set longline fishery	~~2014~~	Observer data	~~21%~~	~~0~~ ~~0~~	~~0 (-)~~	~~0~~ ~~0~~	~~0 (-)~~
	~~2015~~		~~21%~~	~~0~~ ~~1/1‡~~	~~0.7 (0.9)~~	~~1/1~~ ~~0~~	~~4.3 (0.9)~~
	~~2016~~		~~20%~~	~~0~~ ~~0~~	~~0 (-)~~	~~0~~ ~~0~~	~~0 (-)~~
	2017		20%	0 0	0 (-)	0 0	0 (-)
	2018		18%	0 1/1	0.9~~0.8~~ (0.8)	0 0	0 (-)
	2019		21%	0 1/0	0.4 (1.1)	0 0	0 (-)
	2020		15%	0 0	0 (-)	0 0	0 (-)
	2021		18%	1/1 0	5.4 -1.0)	0 0	0 (-)
Mean Estimated Annual Take (CV) 2017-2021					**1.3 (1.6)** ~~**0.3 (0.9)**~~		0 (-)~~**0.9 (1.1)**~~
Hawaiʻi-based shallow-set longline fishery	~~2014~~	Observer data	~~100%~~	~~0~~ ~~0~~	~~0~~	~~0~~ ~~0~~	~~0~~
	~~2015~~		~~100%~~	~~0~~ ~~0~~	~~0~~	~~0~~ ~~0~~	~~0~~
	~~2016~~		~~100%~~	~~0~~ ~~0~~	~~0~~	~~0~~ ~~0~~	~~0~~
	2017		100%	0 0	0	0 0	0
	2018		100%	0 0	0	0 0	0
	2019		100%	0 0	0	0 0	0
	2020		100%	0 0	0	0 0	0
	2021		100%	0 0	0	0 0	0
Mean Annual Takes (100% coverage) 2017-2021					**0**		**0**
Minimum total annual takes within U.S. EEZ (2017-2021)							0 (-)~~**0.9 (1.1)**~~

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

to the cause of death (Bradford and Lyman 2019). In 2020, a short-finned pilot whale was observed off Hawaiʻi Island with trailing line from its mouth, suggesting the whale was hooked in the mouth or had ingested the hook (Bradford and Lyman 2023), an injury that is considered serious according to criteria for assessing serious injury in marine mammals (NMFS 2023). No estimates of human-caused mortality or serious injury are currently available for nearshore hook and line or gillnet fisheries because these fisheries are not observed or monitored for protected species bycatch.

There are currently two distinct longline fisheries based in Hawaiʻi: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas, but are prohibited from operating within the Papahānaumokuākea Marine National Monument (PMNM) and within the Longline Exclusion Zone around the MHI and the Pacific Remote Islands and Atolls (PRIA) MNM around Johnston Atoll. The PMNM originally included the waters within a 50 nmi radius around the NWHI. In August, 2016, the PMNM area was expanded to extend to the 200 nmi EEZ boundary west of 163° W. ~~, a region that extends 50 nmi from shore around the Northwestern Hawaiian Islands, and within the Longline Exclusion Area, a region extending 25-75 nmi from shore around the main Hawaiian Islands. Commercial fishing has also been banned within the expanded PMNM since August 2016.~~ Between 2017-2021~~2014 and 2018~~, no short-finned pilot whales were observed hooked or entangled in the SSLL fishery (100% observer coverage), and one was observed taken in the DSLL fishery (~~18~~15-21% observer coverage) (Figure 3, ~~Bradford 2018a, 2018b, 2020, Bradford and Forney 2017,~~ McCracken and Cooper, 2022b~~2019~~), outside~~inside~~ the Hawaiian Islands EEZ. Based on an evaluation of the observer's description of the interaction and following the most recently developed criteria for assessing serious injury in marine mammals (NMFS ~~2012~~2023), this~~twothis~~ short-finned pilot whale~~s~~ was~~werewas~~ considered seriously injured. Two additional unidentified "blackfish" (unidentified cetaceans known to be either false killer whales or short-finned pilot whales) were taken during ~~2014-2018~~2017-2021 ~~(Bradford 2018b, 2020, Bradford and Forney 2017~~, McCraken and Cooper, 2022b), both within the DSLL fishery. Both of the blackfish~~DSLL~~ interactions occurred outside the Hawaiian Islands EEZ, with one considered seriously injured and one considered non-seriously injured~~whose injury status could not be determined based on the information provided by the observer~~. Takes of unidentified blackfish are prorated to false killer whale and short-finned pilot whale based on distance from shore (McCracken 2010), given patterns of previous bycatch for each species. ~~Unidentified blackfish are prorated to each stock based on distance from shore (McCracken 2010). The distance-from-shore model was chosen following consultation with the Pacific Scientific Review Group, based on the model's performance and simplicity relative to a number of other more complicated models with similar output (McCracken 2010).~~ Proration of unidentified blackfish takes introduces unquantified uncertainty into the bycatch estimates, but until all animals taken can be identified to species (e.g., photos, tissue samples), this approach ensures that potential impacts to all stocks are assessed.

The total estimated number of dead or seriously injured dolphins is calculated based on observer coverage rate, the location of the observed take (inside or outside of the EEZ), and the ratio of observed dead and seriously injured whales versus those judged to be not seriously injured. Observer coverage is measured on a per-trip basis throughout the calendar year as described by McCracken (2019). In years with large fluctuations in observer coverage, such as during the early days of the COVID-19 pandemic when observer coverage dropped to less than 10% during the second quarter of the year, the annual bycatch estimation process may be subset into several periods, as described in McCracken & Cooper (2022a). Average 5-yr estimates of annual mortality and serious injury for 2017-2021~~2014-2018~~ are 1.3 (CV=1.6)~~1.5 (CV = 0.9)~~ short-finned pilot whales outside of the U.S. EEZ~~s~~, and 0 ~~0.9 (CV = 1.1)~~ within the Hawaiian Islands EEZ (Table 2, McCracken and Cooper 2002b). Two additional unidentified cetaceans, ~~considered to be likely~~ likely to be blackfish based on the observer's description, ~~Four additional unidentified cetaceans~~ were taken in the DSLL fishery and~~, some of which~~ may have been short-finned pilot whales.

STATUS OF STOCK

The Hawaiʻi stock of short-finned pilot whales is not considered strategic under the 1994 amendments to the MMPA. The status of short-finned pilot whales in Hawaiian waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Short-finned pilot whales are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. ~~The estimated rate of mortality and serious injury within the Hawaiian Islands EEZ (0.9 animals per year) is less than the PBR (87). Based on the available data, which indicate total fishery-related takes are less than 10% of PBR,~~ In the past 5 years, one short-finned pilot whale was observed in nearshore waters seriously injured by fishing gear, although the source of the gear is unknown (Bradford and Lyman 2023). There is no systematic monitoring for interactions with protected species within near-shore fisheries that may take this species, thus total mean annual takes (0.2 yr) are undetermined. Given the absence of recent recorded longline fishery-related mortality or serious injuries and low levels of nearshore fisheries interactions within the U.S. EEZ~~s~~, the total fishery mortality and serious injury for short-finned pilot whales can be considered to be insignificant and approaching zero. Two short-finned pilot whales were found stranded in separate incidents following Navy sonar training exercises in Hawai‘i in 2014 (Bradford and Lyman 2018). Examination of the whales could ~~did~~ not conclusively link these stranding to use of sonar, though other blackfish have shown sensitivity to sonar training events in Hawaiian waters (Southall *et al*. 2006) and elsewhere (Brownell *et al*. 2009). Two of five short-finned pilot whales that died in a mass stranding on Kauai in 2017 had tissues infected with beaked whale circovirus (Clifton *et al.* 2023), which can lead to serious illness and immunosuppression, though it is not clear what effect that infection had in these strandings.

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Revised ~~5/5/2021~~ 08/16/2023

BRYDE'S WHALE (*Balaenoptera edeni*): Hawaiʻi Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Bryde's whales occur in tropical and warm temperate waters throughout the world. Leatherwood *et al.* (1982) described the species as relatively abundant in summer and fall on the Mellish and Miluoki banks northeast of Hawaiʻi and around Midway Island. Ohsumi and Masaki (1975) reported the tagging of "many" Bryde's whales between the Bonin and Hawaiian Islands in the winters of 1971 and 1972 (Ohsumi 1977). ~~Summer/fall~~ Periodic shipboard surveys of the waters within the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands have regularly encountered ~~resulted in 13~~ Bryde's whales throughout the EEZ ~~– sightings throughout the study area in 2002, 30 in 2010, and 2 in 2017~~ (Figure 1)~~; Barlow 2006, Bradford *et al.* 2017, Yano *et al.* 2018)~~. There is currently no biological basis for defining separate stocks of Bryde's whales in the central North Pacific. Bryde's whales were seen occasionally off southern California (Morejohn and Rice 1973) in the 1960s, but their seasonal occurrence has increased since at least 2000 based on detection of their distinctive calls (Kerosky *et al.* 2012).

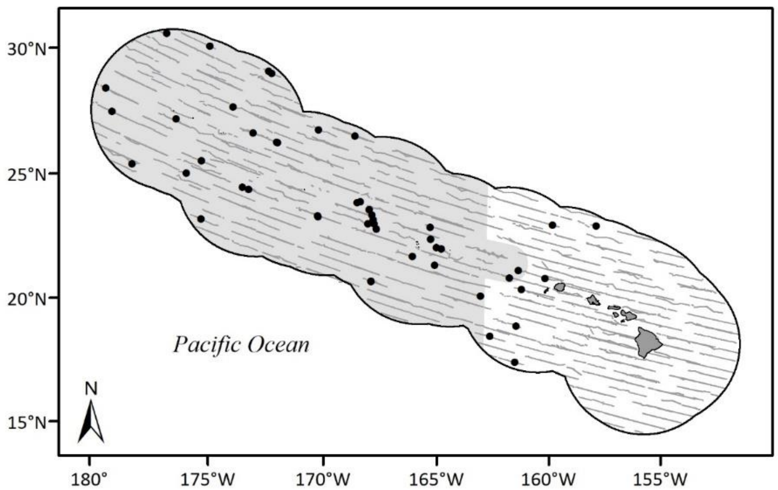

Figure 1. Bryde's whale sighting locations (circles) and survey effort (gray lines) during the 2002 (~~diamonds~~Barlow 2006), 2010 (~~circle~~Bradford *et al.* 2017), and 2017 (~~square~~Yano *et al.* *2018*) shipboard surveys of the U.S. EEZ ~~waters surrounding~~ around the Hawaiian Islands (~~Barlow 2006, Bradford *et al.* 2017, Yano *et al.* 2018~~outer black line). ~~Outer line represents approximate boundary of survey area and U.S. EEZ. Dark gray shading indicates area of t~~The ~~original~~ Papahānaumokuākea Marine National Monument in the western portion of the EEZ is shaded gray.~~, with the lighter gray shading denoting the full 2016 Expansion area. Dotted line represents the 1000 m isobath.~~

For the Marine Mammal Protection Act (MMPA) stock assessment reports, Bryde's whales within the Pacific U.S. EEZ are divided into two areas: 1) Hawaiian waters (this report), and 2) the eastern Pacific (east of 150ºW and including the Gulf of California and waters off California). The Hawaiʻ~~ian~~ stock includes animals found both within the Hawaiian Islands EEZ and in adjacent high seas waters; however, because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of this stock is evaluated based on data from the U.S. EEZ ~~waters~~ of the Hawaiian Islands (NMFS 2005).

POPULATION SIZE

Encounter data from shipboard line-transect surveys of the ~~entire~~ Hawaiian Islands EEZ were~~was~~ recently reevaluated for each survey year, resulting in updated model-based abundance estimates of Bryde's whales in the entirety of the Hawaiian Islands EEZ (Becker *et al.* 2021, 2022) (Table 1).

Sighting data from 2002 to 2020~~2017~~ within the Hawaiian Islands EEZ were used to derive habitat-based models of animal density for two periods: 2002-2017 (Becker et al. 2021) and 2017-2020 (Becker *et al*. 2022). The most recent set of models include three notable changes from the 2002-2017 models: use of calibrated group size estimates, as in Bradford *et al*. (2021), exclusion of a spatial term on model selection, requiring more explicit reliance on environmental variables, and incorporating new approaches (Miller *et al*. 2022) for more comprehensively estimating uncertainty in model predictions that account for the combined uncertainty around all parameter estimates. The modeling framework incorporated Beaufort-specific trackline detection probabilities for Bryde's whales from Barlow *et al*. (2015). ~~The m~~Models were ~~then~~ used to predict density and abundance for each survey year based on the environmental conditions within that year (see Forney *et al.* 2015, Becker *et al.* 2016). ~~The modeling~~

Table 1. Model-based line-transect abundance estimates for Bryde's whales in the ~~derived from surveys of the entire~~ Hawaiian Islands EEZ in 2002 and~~,~~ 2010~~, and 2017~~ (Becker *et al.* 2021) and 2017 and 2020 (Becker *et al.* 2022), derived from NMFS surveys in the central Pacific since 2000. The Becker *et al.* (2022) analysis incorporates a more comprehensive model-based approach to estimating model uncertainty, such that the CVs and 95% confidence limits for 2002/2010 and 2017/2020 are not directly comparable.

Year	Model-based Abundance	CV	95% Confidence Limits
2020	791	0.29	456-1,372
2017	679~~602~~	0.29 ~~0.22~~	392-1,175~~397-842~~
2010	822	0.20	554-1,220
2002	562	0.21	375-842

~~framework incorporated Beaufort-specific trackline detection probabilities for Bryde's whales from Barlow et al. (2015).~~ Bradford *et al.* (2021) produced design-based abundance estimates for Bryde's whales in~~for~~ 2002, 2010, and 2017 ~~each survey years~~ that can be used as a point of comparison to the model-based estimates for those years. While on average, the estimates are broadly similar between the two approaches, the annual design-based estimates show much greater variability between years than do the model-based estimates (Figure 2). The model-based approach reduces variability through explicit examination of habitat relationships across the full dataset, while the design-based approach evaluates encounter data for each year separately and thus is more susceptible to the effects of encounter rate variation. Bradford *et al.* (2021) found through simulation that the pronounced decrease in the design-based estimates between 2010 and 2017 could not be explained by encounter rate variation alone and likely reflected true changes in distribution of Bryde's whales in the study area between those survey years. The model based-estimates demonstrated a much smaller decrease between 2010 and 2017, but are based on the implicit assumption that changes in abundance are attributed to environmental variability alone. ~~There are insufficient data to include e~~Explicitly incorporating~~e~~ a trend term into the model is not possible due to the insufficient sample size to test for temporal effects. Despite not fully accounting for inter-annual variation in total abundance, the model-based estimates are considered the best available estimate for each survey year. Previously published abundance~~design-based~~ estimates for the Hawaiian Islands EEZ ~~from 2002 and 2010 surveys~~ (e.g. Barlow 2006, Becker *et al.* 2012, Forney *et al.* 2015, Bradford *et al.* 2017) used a subset of the dataset used by Becker *et al.* (2021, 2022) and Bradford *et al.* (2021) to derive line-transect parameters, such that these estimates have been superseded by the estimates presented here. The best estimate of abundance is based on the 2020 survey, or 791 (CV=0.29) Bryde's whales.

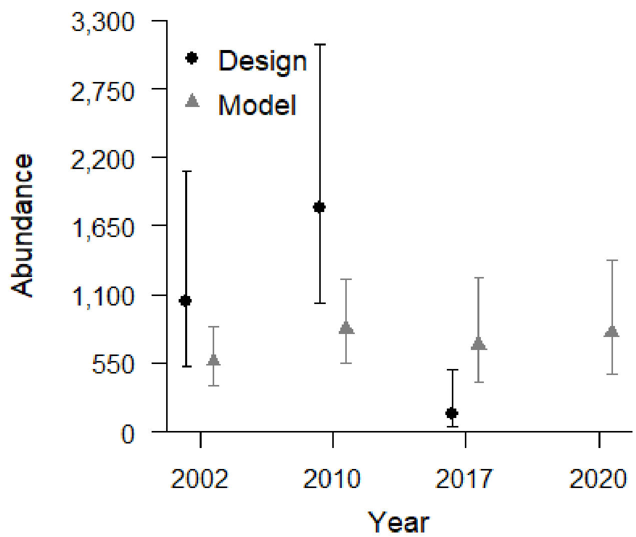

Figure 2. Comparison of design-based (black circles, Bradford *et al.* 2021) and model-based (gray triangles, Becker *et al.* 2021, 2022) estimates of abundance for Hawai'i Bryde's whales for each survey year (2002, 2010, 2017, 2020).

Tillman (1978) concluded from Japanese and Soviet CPUE data that the stock size in the North Pacific pelagic whaling grounds, mostly to the west of the Hawaiian Islands, declined from approximately 22,500 in 1971 to 17,800 in 1977. An estimate of 13,000 (CV=0.202) Bryde's whales was made from vessel surveys in the eastern tropical Pacific between 1986 and 1990 (Wade and Gerrodette 1993). The area to which this estimate applies is mainly east and somewhat south of the Hawaiian Islands, and it is not known whether these animals are part of the same population that occurs around the Hawaiian Islands.

Minimum Population Estimate

The ~~M~~minimum population estimate ~~size~~ is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2020~~2017~~ abundance estimate (Becker *et al.* 2022), or 623~~501~~ Bryde's whales.

Current Population Trend

The model-based abundance estimates for Bryde's whales provided by Becker *et al.* (2021, 2022) do not explicitly allow for examination of population trend other than that driven by environmental factors. Although annual encounter rate variation may have a large impact on abundance estimates for species with low density and patchy distribution, Bradford *et al.* (2021) suggest that the very high sighting rate in 2010 and very low sighting rate in 2017 cannot be explained through encounter rate variation alone and that there may be true fluctuations in Bryde's whale abundance within the Hawaiian Islands EEZ. Model-based examination of Bryde's whale population trends including sighting data beyond the Hawaiian Islands EEZ will be required to more fully examine trend for this stock.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

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

POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Hawaii stock of Bryde's whales is calculated as the minimum population size within the U.S EEZ of the Hawaiian Islands (623~~501~~) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a stock of unknown status with no known fishery mortality or serious injury within the Hawaiian Islands EEZ; Wade and Angliss 1997), resulting in a PBR of 6.2~~5.0~~ Bryde's whales per year.

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Fishery Information

There are currently two distinct longline fisheries based in Hawaii: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas, but are prohibited from operating within the Papahānaumokuākea Marine National Monument (PMNM) and within the Longline Exclusion Zone around the main Hawaiian Islands and the Pacific Remote Islands and Atolls (PRIA) MNM around Johnston Atoll. The PMNM originally included the waters within a 50 nmi radius around the Northwestern Hawaiian Islands. In August, 2016, the PMNM area was expanded to extend to the 200 nmi EEZ boundary west of 163° W. Between 2017~~2014~~ and 2021~~2018~~, no Bryde's whales were observed hooked or entangled in the SSLL fishery (100% observer coverage) or the DSLL fishery (15-21~~18-22~~% observer coverage) (~~Bradford 2018a, 2018b, 2020, Bradford and Forney 2017,~~ McCracken and Cooper 2022 ~~2019~~). Baleanopterid~~Large~~ whales have been observed entangled in longline gear off the Hawaiian Islands in the past (Bradford 2018~~Forney 2010~~).

Historical Mortality

Small numbers of Bryde's whales were taken near the Northwestern Hawaiian Islands by Japanese and Soviet whaling fleets in the early 1970s (Ohsumi 1977). Pelagic whaling for Bryde's whales in the North Pacific ended after the 1979 season (IWC 1981), and coastal whaling for this species ended in the western Pacific in 1987 (IWC 1989).

STATUS OF STOCK

The Hawaiʻi stock of Bryde's whales is not considered strategic under the 1994 amendments to the MMPA. The status of Bryde's whales in Hawaiian waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Bryde's whales are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. Given the absence of recent recorded fishery-related mortality or serious injuries within the Hawaiian Islands EEZ, the total fishery mortality and serious injury can be considered to be insignificant and approaching zero. The increasing level of anthropogenic noise in the world's oceans has been suggested to be a habitat concern for large whales (Richardson *et al.* 1995, Weilgart 2007).

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Appendix 2. Pacific reports revised in 2023 are highlighted. S=strategic stock, N=non-strategic stock. unk=unknown, undet=undetermined, n/a=not applicable.

Species (Stock)	N_{est}	CV N_{est}	N_{min}	R_{max}	Fr	PBR	Annual Human-Caused Mortality and Serious Injury	Annual Commercial Fishery Mortality and Serious Injury	Strategic Status	Recent Abundance Surveys			Revised
California sea lion (U.S.)	257,606	n/a	233,515	0.12	1	14,011	≥321	≥197	N	2008	2013	2014	2018
Harbor Seal (California)	30,968	n/a	27,348	0.12	1	1,641	43	30	N	2004	2009	2012	2014
Harbor Seal (Oregon/Washington Coast)	unk	unk	unk	0.12	1	undet	10.6	7.4	N	1999			2013
Harbor Seal (Washington Northern Inland Waters)	~~unk~~ 16,451	~~unk~~ 0.07	~~unk~~ 15,462	0.12	1	~~undet~~ 928	~~9.8~~ 40	~~2.8~~ 0	N	~~1999~~ 2013	2014	2019	~~2013~~ 2023
Harbor Seal (Southern Puget Sound)	~~unk~~ 2,417	~~unk~~ 0.08	~~unk~~ 2,253	0.12	1	~~undet~~ 135	~~3.4~~ 13.8	~~1~~ 0	N	~~1999~~ 2013	2014	2019	~~2013~~ 2023
Harbor Seal (Hood Canal)	~~unk~~ 3,363	~~unk~~ 0.16	~~unk~~ 2,940	0.12	~~1~~ 0.5	~~undet~~ 88	~~0.2~~ 2.0	~~0.2~~ 0	N	~~1999~~ 2010	2013	2019	~~2013~~ 2023
Northern Elephant Seal (California Breeding)	187,386	n/a	85,369	0.12	1	5,122	13.7	5.3	N	2005	2010	2013	2021
Guadalupe Fur Seal (Mexico)	34,187	n/a	31,019	0.137	0.5	1,062	≥3.8	≥1.2	S	2008	2009	2013	2019
Northern Fur Seal (California) (California)	14,050	n/a	7,524	0.12	1	451	1.8	≥0.8	N	2010	2011	2013	2015
Monk Seal (Hawai'i)	~~1,465~~ 1,564	~~0.03~~ 0.05	~~1,431~~ 1,444	0.07	0.1	~~5.0~~ 5.1	≥~~4.0~~ 5.4	≥~~2.2~~ 2.6	S	~~2018~~ 2019	~~2019~~ 2020	~~2020~~ 2021	~~2022~~ 2023
Harbor Porpoise (Morro Bay)	4,191	0.56	2,698	0.096	0.5	65	0	0	N	2008	2011	2012	2021
Harbor Porpoise (Monterey Bay)	3,760	0.561	2,421	0.058	0.5	35	≥0.2	≥0.2	N	2011	2012	2013	2021
Harbor Porpoise (San Francisco - Russian River)	7,777	0.62	4,811	0.061	0.5	73	≥0.4	≥0.4	N	2014	2016	2017	2021
Harbor Porpoise (Northern CA/Southern OR)	~~24,685~~ 15,303	~~0.41~~ 0.575	~~17,713~~ 9,759	0.04	1	~~354~~ 306	0	~~1~~ 0	N	~~2011~~ 2016	~~2014~~ 2021	~~2016~~ 2022	~~2021~~ 2023
Harbor Porpoise (Central Oregon) [new stock]	7,492	0.421	5,332	0.04	0.5	53	0	0	N	2016	2021	2022	2023
Harbor Porpoise (Northern OR/Washington Coast)	~~21,487~~ 22,074	~~0.44~~ 0.391	~~15,123~~ 16,068	0.04	0.5	~~151~~ 161	≥~~3.0~~3.2	≥~~3.0~~ 2.8	N	~~2002~~ 2016	~~2010~~ 2021	~~2011~~ 2022	~~2013~~ 2023
Harbor Porpoise (Washington Inland Waters)	11,233	0.37	8,308	0.04	0.4	66	≥7.2	≥7.2	N	2013	2014	2015	2016

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Species (Stock)	N_{est}	CV N_{est}	N_{min}	R_{max}	Fr	PBR	Annual Human-Caused Mortality and Serious Injury	Annual Commercial Fishery Mortality and Serious Injury	Strategic Status	Recent Abundance Surveys			Revised
Dall's Porpoise (California/Oregon/Washington)	16,498	0.61	10,286	0.04	0.48	99	≥0.66	≥0.66	N	2008	2014	2018	2021
Pacific white-sided Dolphin (California/Oregon/Washington)	34,999	0.222	29,090	0.04	0.48	279	7	4	N	2008	2014	2018	2021
Risso's Dolphin (California/Oregon/Washington)	6,336	0.32	4,817	0.04	0.48	46	≥3.7	≥3.7	N	2005	2008	2014	2016
Common Bottlenose Dolphin (California Coastal)	453	0.06	346	0.04	0.48	2.7	≥2.0	≥1.6	N	2009	2010	2011	2016
Common Bottlenose Dolphin (California/Oregon/Washington Offshore)	3,477	0.696	2,048	0.04	0.48	19.7	≥0.82	≥0.82	N	2008	2014	2018	2021
Striped Dolphin (California/Oregon/Washington)	29,988	0.3	23,448	0.04	0.48	225	≥4.0	≥4.0	N	2008	2014	2018	2021
Common Dolphin, short-Beaked (California/Oregon/Washington)	1,056,308	0.21	888,971	0.04	0.5	8,889	≥30.5	≥30.5	N	2008	2014	2018	2021
Common Dolphin, long-Beaked (California)	83,379	0.216	69,636	0.04	0.48	668	≥29.7	≥26.5	N	2008	2014	2018	2021
Northern right Whale Dolphin (California/Oregon/Washington)	29,285	0.72	17,024	0.04	0.48	163	≥6.6	≥6.6	N	2008	2014	2018	2021
Killer Whale (Eastern N Pacific Offshore)	300	0.1	276	0.04	0.5	2.8	0	0	N	2010	2011	2012	2018
Killer Whale (Eastern N Pacific Southern Resident)	~~74~~ 73	n/a	~~74~~ 73	0.035	0.1	0.13	~~≥0.4~~ 0	0	S	~~2019~~ 2020	~~2020~~ 2021	~~2021~~ 2022	~~2022~~ 2023
Short-finned pilot Whale (California/Oregon/Washington)	836	0.79	466	0.04	0.48	4.5	1.2	1.2	N	2005	2008	2014	2016
Baird's Beaked Whale (California/Oregon/Washington)	1,363	0.53	894	0.04	0.5	8.9	≥0.2	0	N	2008	2014	2018	2021
Mesoplodont Beaked whales (California/Oregon/Washington)	3,044	0.54	1,967	0.04	0.5	20	0.1	0.1	N	2005	2008	2014	2017
Cuvier's Beaked Whale (California/Oregon/Washington)	5,454	0.27	4,214	0.04	0.5	42	<0.1	<0.1	N	2008	2014	2016	2022

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Species (Stock)	N_{est}	CV N_{est}	N_{min}	R_{max}	Fr	PBR	Annual Human-Caused Mortality and Serious Injury	Annual Commercial Fishery Mortality and Serious Injury	Strategic Status	Recent Abundance Surveys			Revised
Pygmy Sperm Whale (California/Oregon/Washington)	4,111	1.12	1,924	0.04	0.5	19.2	0	0	N	2005	2008	2014	2016
Dwarf Sperm Whale (California/Oregon/Washington)	unk	unk	unk	0.04	0.5	undet	0	0	N	2005	2008	2014	2016
Sperm Whale (California/Oregon/Washington)	~~1,997~~ 2,606	~~0.57~~ 0.135	~~1,270~~ 2,011	0.04	0.1	~~2.5~~ 4.0	~~0.6~~ 0.52	~~0.64~~ 0.52	S	~~2005~~ 2008	~~2008~~ 2014	~~2014~~ 2018	~~2019~~ 2023
Gray Whale (Eastern N Pacific)	26,960	0.05	25,849	0.062	1	801	131	9.3	N	2011	2015	2016	2020
Gray Whale (Western N Pacific)	290	n/a	271	0.062	0.1	0.12	unk	unk	S	2014	2015	2016	2020
Humpback Whale (Central America / Southern Mexico - California-Oregon-Washington)	1,496	0.171	1,284	0.082	0.1	3.5	14.9	8.1	S	2019	2020	2021	2022
Humpback Whale (Mainland Mexico - California-Oregon-Washington)	3,477	0.101	3,185	0.082	0.5	43	22	11.4	S	2016	2017	2018	2022
Blue Whale (Eastern N Pacific)	1,898	0.085	1,767	0.04	0.2	4.1	≥ ~~19.5~~ 18.6	≥ ~~1.54~~ 0.61	S	2016	2017	2018	~~2021~~ 2023
Fin Whale (California/Oregon/Washington)	11,065	0.405	7,970	0.04	0.5	80	≥ ~~43.6~~ 43.4	≥ ~~0.64~~ 0.41	S	2008	2014	2018	~~2021~~ 2023
Sei Whale (Eastern N Pacific)	~~519~~ 864	~~0.4~~ 0.40	~~374~~ 625	0.04	0.1	~~0.75~~ 1.25	≥ ~~0.2~~ 0	≥ ~~0.2~~ 0	S	2005	2008	2014	~~2018~~ 2023
Minke Whale (California/Oregon/Washington)	915	0.792	509	0.04	0.4	4.1	≥ ~~0.59~~ 0.19	≥ ~~0.59~~ 0.17	N	2008	2014	2018	~~2021~~ 2023
Bryde's Whale (Eastern Tropical Pacific)	unk	unk	unk	0.04	0.5	undet	unk	unk	N	n/a	n/a	n/a	2015
Rough-toothed Dolphin (Hawai'i)	~~76,375~~ 83,915	~~0.41~~ 0.49	~~54,804~~ 56,782	0.04	0.5	~~548~~ 511	~~0~~ 3.2	~~0~~ 3.2	N	2002	2010	2017	~~2020~~ 2023
Rough-toothed Dolphin (American Samoa)	unk	unk	unk	0.04	0.5	undet	unk	unk	unk	n/a	n/a	n/a	2010
Risso's Dolphin (Hawai'i)	~~7,385~~ 6,979	~~0.22~~ 0.29	~~6,150~~ 5,283	0.04	0.5	~~61~~ 53	0	0	N	~~2002~~ 2010	~~2010~~ 2017	~~2017~~ 2020	~~2020~~ 2023

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Species (Stock)	N_{est}	CV N_{est}	N_{min}	R_{max}	Fr	PBR	Annual Human-Caused Mortality and Serious Injury	Annual Commercial Fishery Mortality and Serious Injury	Strategic Status	Recent Abundance Surveys			Revised
Common Bottlenose Dolphin (Hawai‘i Pelagic)	~~unk~~ 24,669	~~unk~~ 0.57	~~unk~~ 15,783	0.04	0.5	~~undet~~ 158	0	0	N	~~2002~~ 2010	~~2010~~ 2017	~~2017~~ 2020	~~2017~~ 2023
Common Bottlenose Dolphin (Kaua’i and Ni'ihau)	~~unk~~ 112	~~unk~~ 0.24	~~97~~ 92	0.04	0.5	~~1~~ 0.9	unk	unk	N	~~2003~~ 2016	~~2012~~ 2017	~~2015~~ 2018	~~2017~~ 2023
Common Bottlenose Dolphin (O'ahu)	~~unk~~ 112	~~unk~~ 0.17	~~n/a~~ 97	0.04	0.5	~~undet~~ 1.0	unk	unk	N	~~2002~~ 2016	~~2003~~ 2017	~~2006~~ 2017	~~2017~~ 2023
Common Bottlenose Dolphin (~~4 Islands Region~~Maui Nui)	~~unk~~ 64	~~unk~~ 0.15	~~n/a~~ 56	0.04	0.5	~~undet~~ 0.6	unk	unk	N	~~2002~~ 2016	~~2003~~ 2017	~~2006~~ 2018	~~2017~~ 2023
Common Bottlenose Dolphin (Hawai‘i Island)	~~unk~~ 136	~~unk~~ 0.43	~~91~~ 96	0.04	0.5	~~0.9~~ 1.0	~~unk~~ ≥ 0.2	unk	N	~~2002~~ 2016	~~2003~~ 2017	~~2006~~ 2018	~~2017~~ 2023
Pantropical Spotted Dolphin (Hawai‘i Pelagic)	~~39,798~~ 67,313	~~0.51~~ 0.27	~~26,548~~ 53,839	0.04	0.5	~~265~~ 538	0	0	N	~~2002~~ 2010	~~2010~~ 2017	~~2017~~ 2020	~~2020~~ 2023
Pantropical Spotted Dolphin (O‘ahu)	unk	unk	unk	0.04	0.5	undet	unk	unk	N			n/a	~~2017~~ 2023
Pantropical Spotted Dolphin (~~4 Islands Region~~ Maui Nui)	unk	unk	unk	0.04	0.5	undet	unk	unk	N			n/a	2017
Pantropical Spotted Dolphin (Hawai‘i Island)	unk	unk	unk	0.04	0.5	undet	≥ 0.2	≥ 0.2	N			n/a	2017
Spinner Dolphin (Hawai‘i Island)	665	0.09	617	0.04	0.5	6.2	≥ 1.0	unk	N	2010	2011	2012	2018
Spinner Dolphin (O’ahu / 4 Islands Region)	n/a	n/a	n/a	0.04	0.5	undet	≥ 0.4	unk	N	1998	2002	2007	2018
Spinner Dolphin (Kaua’i and Ni’ihau)	n/a	n/a	n/a	0.04	0.5	undet	unk	unk	N	1995	1998	2005	2018
Spinner Dolphin (Hawai‘i Pelagic)	unk	unk	unk	0.04	0.5	undet	0	0	N		2002	2010	2018
Spinner Dolphin (Kure / Midway)	unk	unk	unk	0.04	0.5	undet	unk	unk	N		1998	2010	2018
Spinner Dolphin (Pearl and Hermes Reef)	unk	unk	unk	0.04	0.5	undet	unk	unk	N			n/a	2018

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Species (Stock)	N_{est}	CV N_{est}	N_{min}	R_{max}	Fr	PBR	Annual Human-Caused Mortality and Serious Injury	Annual Commercial Fishery Mortality and Serious Injury	Strategic Status	Recent Abundance Surveys			Revised
Spinner Dolphin (American Samoa)	unk	unk	unk	0.04	0.5	undet	unk	unk	unk			n/a	2010
Striped Dolphin (Hawai'i Pelagic)	~~35,179~~ 64,343	~~0.23~~ 0.28	~~29,058~~ 51,055	0.04	0.5	~~291~~ 511	0	0	N	~~2002~~ 2010	~~2010~~ 2017	~~2017~~ 2020	~~2020~~ 2023
Fraser's Dolphin (Hawai'i)	40,960	0.7	24,068	0.04	0.5	241	0	0	N	2002	2010	2017	2020
Melon-headed Whale (Hawaiian Islands)	40,647	0.74	23,301	0.04	0.5	233	0	0	N	2002	2010	2017	2020
Melon-headed Whale (Kohala Resident)	unk	unk	unk	0.04	0.5	undet	0	0	N	2002	2010	2017	2020
Pygmy killer Whale (Hawai'i)	10,328	0.75	5,885	0.04	0.5	59	0	0	N	2002	2010	2017	2020
False killer Whale (NW Hawaiian Islands)	477	1.71	178	0.04	0.4	1.43	0.16	0.16	N	2002	2010	2017	~~2021~~ 2023
False killer Whale (Hawai'i Pelagic)	~~2,086~~ 5,528	0.35	~~1,567~~ 4,152	0.04	0.5	~~16~~ 33	~~9.8~~ 47	~~9.8~~ 47	~~N~~ S	2002	2010	2017	~~2021~~ 2023
False killer Whale (Main Hawaiian Islands Insular)	~~167~~ 138	~~0.14~~ 0.08	~~149~~ 129	0.04	0.1	~~0.3~~ 0.26	0.03	0.03	S	2013	2014	2015	~~2021~~ 2023
False killer Whale (Palmyra Atoll)	1,329	0.65	806	0.04	0.4	6.4	0.3	0.3	N			2005	2012
False killer Whale (American Samoa)	unk	unk	unk	0.04	0.5	undet	unk	unk	unk	n/a	n/a	n/a	2010
Killer Whale (Hawai'i)	161	1.06	78	0.04	0.5	0.8	0	0	N	2002	2010	2017	2020
Pilot Whale, short-finned (Hawai'i)	~~12,607~~ 19,242	~~0.18~~ 0.23	~~10,847~~ 15,894	0.04	0.4	~~87~~ 159	~~0.9~~ 0.2	~~0.9~~ 0	N	~~2002~~ 2010	~~2010~~ 2017	~~2017~~ 2020	~~2020~~ 2023
Blainville's Beaked Whale (Hawai'i Pelagic)	1,132	0.99	564	0.04	0.5	5.6	0	0	N	2002	2010	2017	2020
Longman's Beaked Whale (Hawai'i)	2,550	0.67	1,527	0.04	0.5	15	0	0	N	2002	2010	2017	2020
Cuvier's Beaked Whale (Hawai'i Pelagic)	4,431	0.41	3,180	0.04	0.5	32	0	0	N	2002	2010	2017	2020
Pygmy Sperm Whale (Hawai'i)	42,083	0.64	25,695	0.04	0.5	257	0	0	N	2002	2010	2017	2020
Dwarf Sperm Whale (Hawai'i)	unk	unk	unk	0.04	0.5	undet	0	0	N	2002	2010	2017	2020
Sperm Whale (Hawai'i)	5,707	0.23	4,486	0.04	0.2	18	0	0	S	2002	2010	2017	2020

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Species (Stock)	N_{est}	CV N_{est}	N_{min}	R_{max}	Fr	PBR	Annual Human-Caused Mortality and Serious Injury	Annual Commercial Fishery Mortality and Serious Injury	Strategic Status	Recent Abundance Surveys				Revised
Blue Whale (Central N Pacific)	133	1.09	63	0.04	0.1	0.1	0	0	S		2002	2010		2017
Fin Whale (Hawai‘i)	203	0.99	101	0.04	0.1	0.2	0	0	S	2002	2010	2017		2020
Bryde’s Whale (Hawai‘i)	~~602~~ 791	~~0.22~~ 0.29	~~501~~ 623	0.04	0.5	~~5~~ 6.2	0	0	N	~~2002~~ 2010	~~2010~~ 2017	~~2017~~ 2020		~~2020~~ 2023
Sei Whale (Hawai‘i)	391	0.9	204	0.04	0.1	0.4	0.2	0.2	S	n/a	2002	2010		2017
Minke Whale (Hawai‘i)	438	1.05	212	0.04	0.5	2.1	0	0	N	2002	2010	2017		2020
Humpback Whale (American Samoa)	unk	unk	150	0.106	0.1	0.4	0	0	S	2006	2007	2008		2009