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TO: The File

FROM: Sarah Malloy
Acting Regional Administrator

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Date: 2023.05.22 15:09:30 -10'00'

DATE: May 22, 2023

SUBJECT: Correction to the Incidental Take Statement Table in the Hawaii Seep-set Longline Fishery Biological Opinion

On May 18, 2023, following the completion of the biological opinion, we identified an error in the summary table of the Incidental Take Statement. Specifically, on page 323 in Table 54, we stated that the maximum five-year running sum for anticipated takes of North Pacific loggerhead sea turtles was 433. This was a typographical error. The actual five-year running sum for anticipated takes of North Pacific Loggerhead sea turtles is 43, as noted and analyzed elsewhere in the biological opinion. Accordingly, we have corrected the value for the five-year running sum for anticipated takes of North Pacific loggerhead sea turtles in Table 54 of this biological opinion to 43. This is a non-substantive change that does not require reinitiation of consultation.

The erratum page is enclosed with this memo. The cell containing corrected information is shaded gray.

Erratum

Endangered Species Act Section 7(a)(2) Biological Opinion on the Authorization of the Hawaii Deep-Set Longline Fishery

Pages 322-333

Table 54. Anticipated take for the DSLL fishery. These are the estimated number of captures, expanded from the observed number of incidental captures and inclusive of prorated numbers based on unidentified captures.

Species	Maximum 5-Year Running Sum
Giant manta ray	181
Indo-West Pacific scalloped hammerhead sharks	14
Oceanic whitetip shark	10,589
Green sea turtle (all species)	77
Leatherback sea turtle	92
North Pacific Loggerhead	43
Olive ridley sea turtle (all species)	592
Sperm whale	6
Main Hawaiian insular false killer whale	0.427 (1 observed)

Endangered Species Act (ESA) Section 7(a)(2) Biological Opinion

Action Agency:	National Marine Fisheries Service, Pacific Islands Region, Sustainable Fisheries Division
Federal Action:	Authorization of the Hawaii Deep-Set Longline Fishery
Consultation Conducted by:	National Marine Fisheries Service, Pacific Islands Region, Protected Resources Division
NMFS File No. (ECO):	PIR-2018-10461
PIRO Reference No.:	I-PI-18-1690-AG
Approved By:	MALLOY.SARAH.JOAN.1262526743 Digitally signed by MALLOY.SARAH.JOAN.1262526743 Date: 2023.05.18 08:03:27 -10'00' Sarah Malloy Acting Regional Administrator, Pacific Islands Region
Date Issued:	5/18/23

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1 INTRODUCTION

Section 7(a)(2) of the Endangered Species Act of 1973, as amended (ESA; 16 U.S.C. 1536(a)(2)) requires each federal agency to ensure that any action they authorize, fund, or carry out is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of critical habitat of such species. When a federal agency's action "may affect" a listed species or its designated critical habitat, that agency is required to consult formally with the National Marine Fisheries Service (NMFS) or the U.S. Fish and Wildlife Service (FWS), depending upon the endangered species, threatened species, or designated critical habitat that may be affected by the action (50 CFR 402.14(a)). Federal agencies are exempt from this general requirement if they have concluded that an action may affect, but "is not likely to adversely affect" endangered species, threatened species, or their designated critical habitat, and NMFS or the FWS concur with that conclusion (50 CFR 402.14 (b)).

Section 7(b)(3) of the ESA requires that at the conclusion of consultation, NMFS provides an opinion stating whether the Federal agency's action is likely to jeopardize ESA-listed species or destroy or adversely modify designated critical habitat. If NMFS determines that the action is likely to jeopardize listed species or destroy or adversely modify critical habitat, in accordance with the ESA section 7(b)(3)(A), NMFS provides a reasonable and prudent alternative that allows the action to proceed in compliance with section 7(a)(2) of the ESA. If incidental take[1] is expected, section 7(b)(4) requires NMFS to provide an incidental take statement that specifies the impact of any incidental taking and includes reasonable and prudent measures to minimize such impacts and terms and conditions to implement the reasonable and prudent measures. NMFS, by regulation has determined that an incidental take statement must be prepared when take is "reasonably certain to occur" as a result of the proposed action (50 C.F.R. 402.14(g)(7)).

For the actions described in this document, the action agency is NMFS' Pacific Islands Region (PIRO) Sustainable Fisheries Division (SFD), which proposes to authorize the operation of the Hawaii deep-set longline fishery (DSLL), as managed under the authority of the Magnuson-Stevens Fishery Conservation and Management Act, the Fishery Ecosystem Plan (FEP) for Pacific Pelagic Fisheries of the Western Pacific Region (Pelagic FEP; WPRFMC 2009a, 2009b) and other applicable laws. The consulting agency for this proposal is also NMFS, represented by PIRO's Protected Resources Division (PRD). This document represents NMFS' biological

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[1] Take" is defined by the ESA as harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct. NMFS defines "harass" as to "create the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavioral patterns which include, but are not limited to, breeding, feeding, or sheltering" (Application and Interpretation of the Term "Harass" Pursuant to the Endangered Species Act: NMFS Guidance Memo May 2, 2016). NMFS defines "harm" as "an act which actually kills or injures fish or wildlife." Such an act may include significant habitat modification or degradation where it actually kills or injures fish or wildlife by significantly impairing essential behavioral patterns, including breeding, spawning, rearing, migrating, feeding or sheltering. Take of species listed as endangered is prohibited at the time of listing, while take of threatened species may not be specifically prohibited unless NMFS has issued regulations prohibiting take under section 4(d) of the ESA.

opinion on the effects of the proposed action on the following endangered and threatened species and critical habitat that has been designated for those species (see Table 2): endangered black abalone, endangered white abalone, threatened chambered nautilus, threatened corals: *Acropora globiceps, A. jacquelineae, A. retusa, A. speciosa, Euphyllia paradivisa, Isopora crateriformis*; endangered Southern Resident killer whale, threatened Central California coast coho salmon, threatened Central Valley spring-run chinook salmon, endangered Sacramento River winter-run Chinook salmon, endangered California coast steelhead, threatened California Central Valley steelhead, threatened southern California coast steelhead, threatened southern North American green sturgeon, endangered eastern Pacific scalloped hammerhead shark, threatened giant manta ray, threatened Indo-West Pacific scalloped hammerhead shark, threatened oceanic whitetip shark, endangered hawksbill sea turtle, threatened Central North Pacific, East Indian-West Pacific, East Pacific, and Southwest Pacific green sea turtles; endangered Central West Pacific and Central South Pacific green sea turtles; endangered North Pacific loggerhead sea turtles; threatened and endangered populations of olive ridley sea turtles, endangered leatherback sea turtle, endangered Hawaiian monk seal, endangered blue whale, endangered fin whale, Mexico humpback whale, endangered North Pacific right whale, endangered sei whale, endangered sperm whale, and endangered main Hawaiian Islands (MHI) insular false killer whale (IFKW).

This biological opinion has been prepared in accordance with the requirements of section 7 of the ESA, the implementing regulations (50 CFR 402), agency policy, and guidance and information contained in NMFS's biological evaluation (NMFS 2018a), NMFS and FWS recovery plans and status reviews for sea turtles (NMFS 2010; NMFS and FWS 1998a, 1998b, 1998c, 1998d; 2007a, 2007b, 2007c, 2007d; 2011, 2013, 2014, 2020a, 2020b; Seminoff et al. 2015), and elasmobranchs (Miller and Klimovich 2017; Young et al. 2017), and the Western Pacific Fishery Management Council (Council) annual stock assessment and fishery evaluation (SAFE reports) for the Pacific Pelagic FEP for 2017 (WPRFMC 2018, 2019, 2020, 2021, 2022), and other sources of information as cited herein.

On July 5, 2022, in *Animal Legal Defense Fund v. Haaland, et. al.*, 4:19-cv-06812-JST, the U.S. District Court for the Northern District of California issued an order vacating the 2019 regulations that were revised or added to 50 CFR part 402 in 2019 ("2019 Regulations," see 84 FR 44976, August 27, 2019) without making a finding on the merits. On September 21, 2022, the U.S. Court of Appeals for the Ninth Circuit granted a temporary stay of the district court's July 5 order. On November 14, 2022, the Northern District of California issued an order granting the government's request for voluntary remand without vacating the 2019 regulations. The District Court issued a slightly amended order two days later on November 16, 2022. As a result, the 2019 regulations remain in effect, and we are applying the 2019 regulations here. For purposes of this consultation and in an abundance of caution, we considered whether the substantive analysis and conclusions articulated in the biological opinion and incidental take statement would be any different under the pre-2019 regulations. We have determined that our analysis and conclusions would not be any different.

1.1 Consultation History

On February 23, 2004, NMFS issued a final biological opinion on regulatory amendments to the Fishery Management Plan for the Pelagic Fisheries of the Western Pacific Region (NMFS 2004a). That biological opinion separately considered the deep-set and shallow-set components of the fisheries. It concluded that authorization of the Pelagics FMP, as modified by proposed sea turtle protection measures, was not likely to adversely affect humpback or sperm whales and was not likely to jeopardize the continued existence of threatened and endangered green turtles (*Chelonia mydas*), endangered leatherback turtles (*Dermochelys coriacea*), threatened loggerhead turtles (*Caretta caretta*), or threatened and endangered olive ridley turtles (*Lepidochelys olivacea*).

In 2004, the deep-set component of the Hawaii-based pelagic longline fishery was estimated to have exceeded the incidental take of olive ridley sea turtles that had been anticipated in the 2004 incidental take statement. Consistent with 50 CFR 402.16(a), NMFS reinitiated formal consultation on the deep-set component of the Hawaii-based pelagic longline fishery on February 17, 2005.

On October 5, 2005, NMFS issued a new biological opinion for the DSLL fishery that exempted the incidental take of green, leatherback, loggerhead, and olive ridley sea turtles (NMFS 2005).

On June 5, 2013, NMFS reinitiated consultation on the DSLL fishery after triggering two of the four reinitiation criteria pursuant to 50 CFR 402.16(a). Reinitiation was warranted because: (1) A new species was listed that may be affected by the action (MHI IFKW); and (2) new information revealed effects to listed species in a manner or to an extent not considered (an interaction between a sperm whale and gear associated with the fishery, NMFS 2014).

In February 2014, while the consultation was ongoing, the DSLL fishery exceeded the number of leatherback sea turtles specified in the incidental take statement, thereby triggering a third reinitiation criteria pursuant to 50 CFR 402.16(a) (NMFS 2014).

On September 10, 2014, NMFS published a final rule (79 FR 53852) that listed 20 new species of reef-building corals as threatened under the ESA. Of those, NMFS expects that seven occur in the Exclusive Economic Zone (EEZ). On October 6, 2014, NMFS determined that Pacific Island pelagic fisheries, including the deep-set and shallow-set fishery, would not affect ESA-listed species of shallow reef-building corals (NMFS 2014).

On September 19, 2014, NMFS' issued a biological opinion for the continued operation of the Hawaii DSLL fishery and concluded that the action was not likely to jeopardize the continued existence of humpback whales, sperm whales, MHI IFKWs, North Pacific loggerhead distinct population segment, leatherback sea turtles, olive ridley sea turtles, green sea turtles, and Indo-west Pacific scalloped hammerhead sharks. NMFS also concluded that the action was not likely to adversely affect hawksbill sea turtles, blue whales, fin whales, sei whales, North Pacific right whales, Hawaiian monk seals, Eastern Pacific scalloped hammerhead sharks, and critical habitat for Hawaiian monk seals and leatherback sea turtles.

On February 27, 2015, the Hawaii shallow set longline (SSLL) fishery interacted with a fin whale, which represented new information revealing effects of the action not considered in the previous biological opinion. On August 21, 2015, NMFS published a final rule revising critical habitat for Hawaiian monk seals. On September 10, 2015, NMFS SFD reinitiated consultation on the SSLL and DSLL fisheries pursuant to 50 CFR 402.16(a) to address these two reinitiation triggers. On September 16, 2015, NMFS determined the Hawaii SSLL and DSLL fisheries were not likely to adversely affect Hawaiian monk seal critical habitat and fin whales (NMFS 2015).

On April 6, 2016, NMFS and FWS issued a final rule to list 11 DPS of green sea turtles (8 threatened and 3 endangered) under the ESA (81 CFR 20058). This final rule removed the previous range-wide listing for the biological species.

On March 24, 2017, NMFS issued a supplement to its 2014 biological opinion in response to a request for consultation on April 13, 2016, because the DSLL fishery had exceeded the amount of take specified in the 2014 biological opinion for North Pacific loggerhead, olive ridley, and green sea turtles (NMFS 2017b). The 2016 listing of 11 green sea turtle DPS also triggered reinitiation criteria 402.16(a)(4). NMFS' 2017 supplemental biological opinion re-affirmed the conclusion of its 2014 biological opinion that the proposed action in 2017 was not likely to jeopardize the continued existence of North Pacific loggerhead, olive ridley, and green sea turtles in the wild. The 2017 supplemental biological opinion incidental take statement replaced the incidental take statement that accompanied the 2014 biological opinion for North Pacific loggerhead, olive ridley and green sea turtles. The revised incidental take statement anticipated and authorized 18 loggerhead interactions across 3 years, 141 for the endangered Mexico and threatened eastern Pacific populations of olive ridley sea turtles, and 42 for the threatened western Pacific population (combined total 183) and 36 green sea turtle interactions across three year split among six of the 11 listed DPS' (12 East Pacific green sea turtles; 6 each Central North Pacific, East Indian-west Pacific, and Southwest Pacific green sea turtles; and 3 each from Central West Pacific and Central South Pacific green sea turtles).

In January 2018, NMFS listed the giant manta ray and oceanic whitetip shark as threatened (83 FR 2916 and 83 FR 4153, respectively), and designated critical habitat for MHI IFKW (83 FR 35062) in July of the same year.

On October 4, 2018, SFD requested PRD to reinitiate formal consultation on the DSLL fishery because SFD determined that the number of interactions with East Pacific green turtles exceeded the amount specified in the incidental take statement contained in NMFS' 2017 biological opinion. NMFS reinitiated consultation pursuant to 50 CFR 402.16(a)(1 and 4).

On February 21, 2019, NMFS PRD acknowledged receipt of NMFS SFD's October 4, 2018, request to initiate consultation, noting that SFD had provided all necessary information to initiate consultation on October 4, 2018, and that consultation had begun on that same day (letter from A. Garrett to B. Harman; NMFS 2018b). NMFS PRD noted that the consultation was held in abeyance for 38 days due to a lapse in appropriations that led to a partial shutdown of the federal government). Consultation resumed on January 28, 2019. NMFS completed a draft biological opinion in September 2019 for internal review, and completed internal review in October 2019.

On October 14, 2019, Ryan Steen, attorney with the Hawaii Longline Association (HLA) emailed Michael Tosatto, PIRO Regional Administrator to request NMFS consider a new model and its implications for this consultation. The Hawaii Longline Association followed this email request with a letter on November 14, 2019, that effectively acknowledged a delay in the consultation to ensure that the new model, which had been run for the SSLL fishery (after the biological opinion was completed) would be run using data from the DSLL fishery. At its 180th meeting, October 22-24, 2019, the Council also "urged PIRO to delay the completion of the deep-set longline fishery consultation to ensure that all anticipated scientific information including the PIFSC take (sic) model are considered in the BiOp" (also repeated in letter from the Western Pacific Fishery Management Council to M. Tosatto, October 31, 2019).

On November 21, 2019, NMFS PIRO proposed, and on November 25, 2019, the Hawaii Longline Association (HLA) agreed to extend the consultation to allow NMFS to consider and incorporate the results of the new model as well as establish a series of regular calls to provide updates on NMFS progress.

On February 12, 2020, NMFS Pacific Islands Fishery Science Center (PIFSC) shared that the model run was sufficiently complete to provide an internal briefing on its results (email from C. Littnan to A. Garrett and B. Harman, Feb. 12, 2020). An internal briefing was held on February 26, 2020.

On February 13, 2020, at an internal briefing to present the results of the analyses of PIFSC's "take" model for the DSLL, PRD asked PIFSC to also develop a model run that examined the aggregate effect of the American Samoa longline fishery (ASLL), Hawaii DSLL and SSLL, as develop a couple more scenarios for the DSLL – that all interactions are adults, that all are adult females. PIFSC summarized this request in email on March 3, 2020 (C. Littnan to A. Garrett and B. Harman, email March 3, 2020).

On March 2, 2020, NMFS responded to the Council's letter and noted that NMFS had agreed to a consultation timeline with HLA that should allow the opportunity to consider the on-going work ("take" model) of PIFSC.

In July 2020, the Council sent NMFS a letter (to M. Tosatto, July 1, 2020) again requesting that NMFS consider the assessment of the population –level impacts of North Pacific loggerhead and Western Pacific leatherback turtle interactions (aka, PIFSC "take" model) in the DSLL and the ASLL consultations. In this same letter, the Council noted that "interactions in the DSLL and ASLL are low compared to other fisheries and thus any measures implemented in DSLL and ASLL are likely to have *only a limited effect* at the population level." The Council also noted that priority should be placed on improving handling and release methods when developing potential mitigation measures; hard caps and that trip limits are *not feasible* measures in the DSLL and ASLL without 100% coverage or an electronic monitoring program. It was during this summer that Council staff, SFD, and HLA started discussions to minimize the effects of the action.

On August 12, 2020, PIFSC provided PRD a with a summary of their analysis for endangered Western Pacific leatherbacks and North Pacific loggerheads that examined the effect of the combined mortality in the ASLL, SSLL, and DSLL fisheries.

On November 3, 2020, PRD met with PIFSC staff seeking some early feedback on the draft analysis for MHI IFKW in this consultation.

On November 27, 2020, the HLA announced that it would voluntarily convert the DSLL fishery from wire leaders to monofilament nylon leaders by July 1, 2021. This gear change was proposed in order to reduce the DSLL's effects on oceanic whitetip sharks and other protected species (HLA 2020).

On December 7, 2020, the Council commended the HLA for their proposal to replace wire leaders with monofilament, and other associated improvements to the fishery (letter to HLA from K. Simonds). The Council noted that they would initiate development of a regulatory amendment to prohibit the use of wire leaders in the DSLL for the March 2021 meeting.

On January 22, 2021, NMFS PRD requested additional information regarding the proposed gear change from wire to monofilament leaders including more information on the effect of the gear change to listed species. This gear change represented a significant change in the proposed action requiring new scientific analyses. As a result, the consultation was subsequently "paused" in NMFS' Environmental Consultation Organizer.

On February 9, 2021, the HLA provided notification to NMFS that they would assist the action agency in gathering and providing responsive information for the DSLL consultation. At their March 2021 meeting, the Council requested PIFSC to complete the analysis evaluating the effects of leader material on catchability of oceanic whitetip sharks, as well as target and other non-target species to inform the Council's final action. The preliminary results of their analysis was presented to the Oceanic Whitetip Shark Working Group on April 22, 2021, and the written report was provided to Council staff on May 10, 2021. The analysis found that prohibition of wire leaders in the DSLL was expected to reduce mortality of oceanic whitetip sharks by approximately 30% (Bigelow and Carvalho 2021).

On March 18, 2021, PRD requested additional data on the DSLL fishery from SFD to cover years 2018 to 2020.

At their June 2021 meeting (June 22-24, 2021), the Council took final action on regulatory amendments under the Pelagic FEP to improve post-hooking survivorship of oceanic whitetip sharks. Specifically, the Council recommended regulatory amendments to prohibit wire leaders in the Hawaii deep-set longline fishery; and require removal of trailing gear from oceanic whitetip sharks for all longline vessels operating under the Pelagic FEP.

On June 30, 2021, NMFS SFD provided additional information about the effects of wire versus nylon leaders on ESA-listed species in the fishery.

On July 12, 2021, NMFS issued a letter to HLA to thank them for discussing the conversion of gear in the Hawaii DSLL fishery. In that letter, NMFS proposed an extension to complete the biological opinion and regular meetings to “check in.” At that time, NMFS expected to provide a draft biological opinion by October 30, 2021.

On January 19, 2022, NMFS SFD issued a proposed rule to prohibit the use of wire leaders in the Hawaii DSLL and require the removal of fishing gear from any oceanic whitetip shark caught in all of the region’s domestic longline fisheries (87 FR 2742).

On April 28, 2022, NMFS published a final rule to prohibit the use of wire leaders in the Hawaii DSLL. The rule also requires, with limited exceptions for safety and data collection, that fishermen remove fishing gear from any oceanic whitetip shark caught in all longline fisheries operating under the FEP in accordance with the requirements set forth at 50 CFR 665.811(a). The rule is intended to reduce the amount of fishing gear (aka, trailing gear) attached to released oceanic whitetip sharks.

On September 28, 2022, NMFS issued a no-jeopardy supplemental biological opinion on the agency’s authorization of the DSLL fishery considering the potential effects to threatened oceanic whitetip sharks and giant manta rays. For all other species adversely affected by the Hawaii Deep-set fishery, the 2014 biological opinion, as supplemented in 2017, remained valid.

On February 2, 2023, NMFS PIFSC completed modifications to a revised population viability analysis (PVA) on the West Pacific leatherback sea turtle, incorporating DSLL capture data from 2018 to 2021 and updating how maturity status is determined within the model, allowing for small turtles to be mature. This was in response to concerns from PRD regarding the potential for the previous model to underestimate the reproductive value of larger juveniles based on estimated carapace lengths.

1.2 Description of the Proposed Action

NMFS proposes to authorize the Hawaii DSLL as it has been managed under the existing regulatory framework of the Pelagic FEP and other applicable laws (e.g., ESA, National Environmental Policy Act [NEPA] and Marine Mammal Protection Act [MMPA]). The Pelagic FEP and its implementing regulations establish the framework for the Council and NMFS to manage this fishery (WPRFMC 2009a; 50 CFR 665).

Domestic longline fishing in the Action Area consists of two separately managed fisheries, the Hawaii DSLL and the SSLL. The Hawaii DSLL targets primarily bigeye tuna (*Thunnus obesus*) and occasionally yellowfin tuna (*T. albacares*), while the SSLL fishery targets swordfish (*Xiphias gladius*). Both fish in the EEZ around the Hawaiian Islands and on the high seas. Longline fishing gear consists of a mainline that exceeds one nautical mile (nm) in length suspended horizontally in the water column, from which branch lines with baited hooks are attached (Figure 1). Longline fishing allows a vessel to distribute effort over a large area to harvest fish that are not concentrated in great numbers. Overall catch rates in relation to the number of hooks are generally low (less than 2%; NMFS 2018b). Longline fishing involves "setting" (deploying) a mainline horizontally at a preferred depth in the water column using floats spaced at regular intervals. Mainline lengths can be 18 to 60 nm long. Crewmembers attach branch lines, with clips, to the mainline at regular intervals, and each branch line has a single baited hook.

After it is set, the gear "soaks" (fishes) for several hours before being "hauled" (retrieved). In longlining, a "set" is the deployment and retrieval of a mainline, floats, and branch lines. Usually, crewmembers make one set per day. Hawaii DSLL fishing trips are usually 3-4 weeks in duration, with about 13 days spent fishing (NMFS 2001, 2005; Beverly and Chapman 2007; WPRFMC 2009a; WPRFMC 2020). Figure 1 illustrates the difference between Hawaii SSLL and DSLL fishery gear configuration. Vessels from the Hawaii DSLL fishery may transition to the SSLL in Hawaii; however, these vessels must return to port and exchange gear in order to participate as it is a separate fishery (NMFS 2019a).

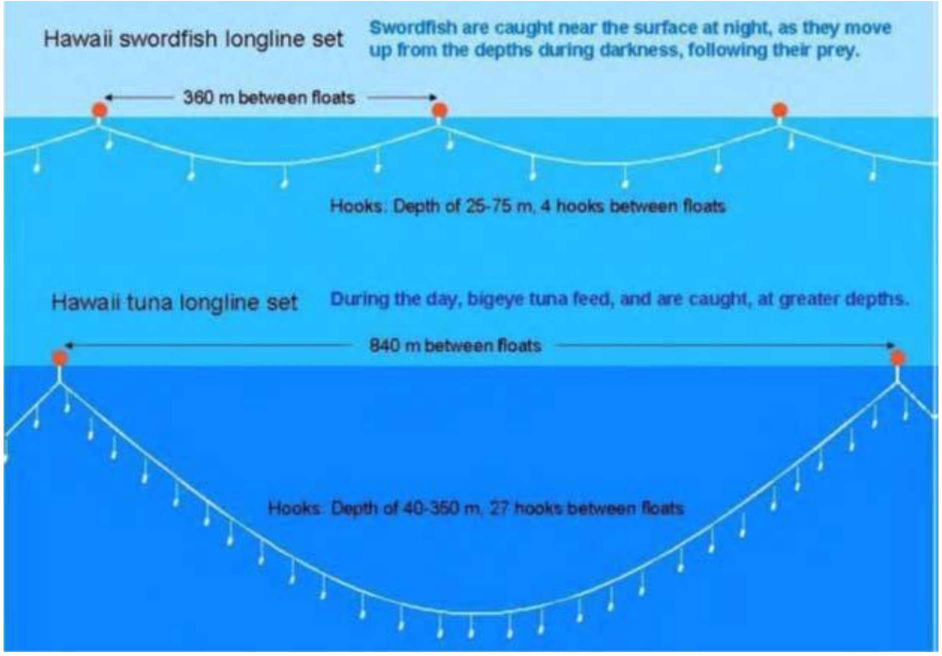


Figure 1. Generalized depiction of swordfish (shallow) and tuna (deep) longline gear configurations (NMFS 2018a).

The following summarize the current requirements that apply to the Hawaii DSLL fishery (50 CFR Parts 229, 300, 404, 600, and 665):

Fishing Permits and Certificates Required (on board each fishing vessel);

- Hawaii Longline Limited Entry Permit.
- Marine Mammal Authorization Program Certificate.
- High Seas Fishing Compliance Act Permit.
- Western and Central Pacific Fisheries Commission (WCPFC) Convention Area Endorsement.
- Protected Species Workshop Certificate.
- Western Pacific Receiving Vessel Permit, if applicable.
- State of Hawaii Commercial Marine License.

Reporting, Monitoring, and Gear Identification;

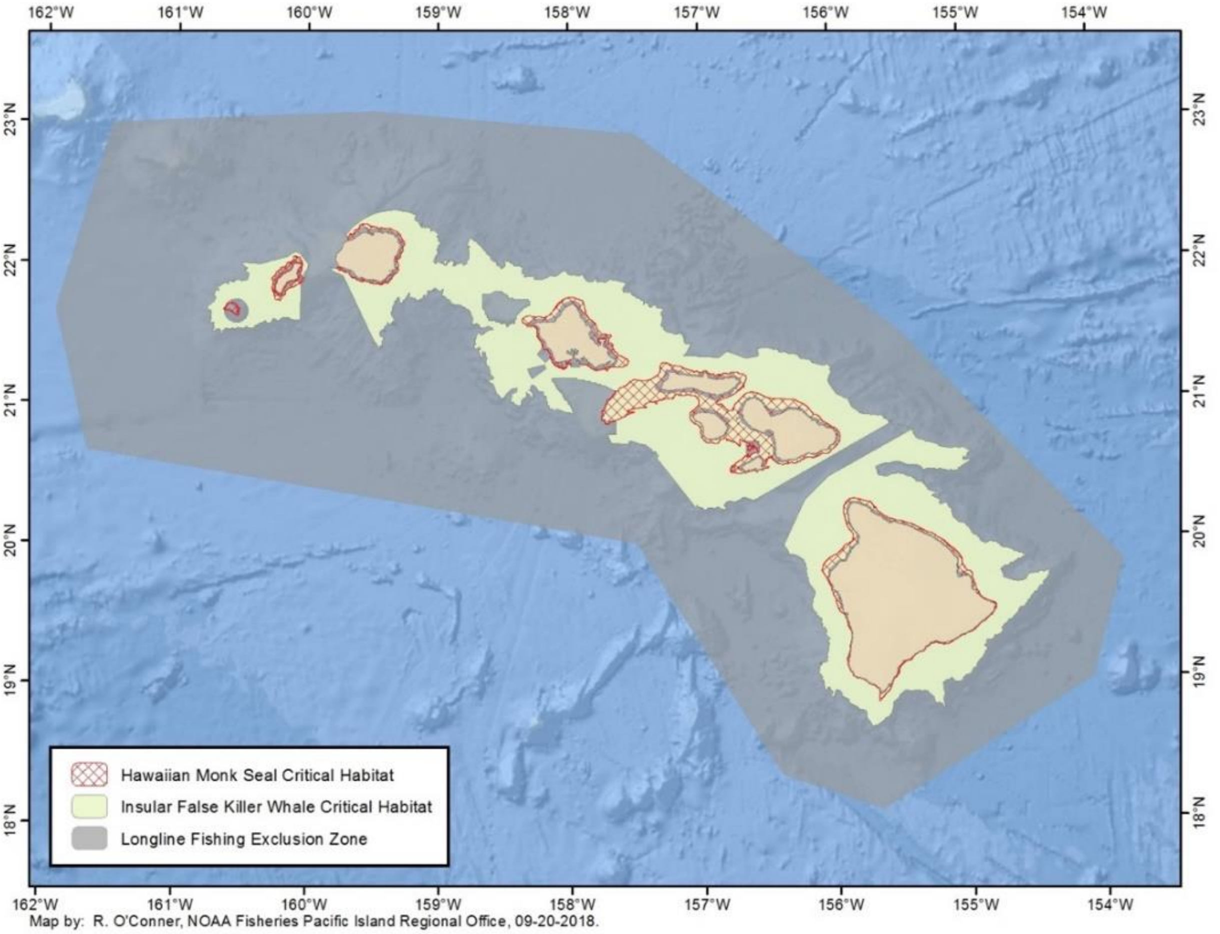
- NMFS Western Pacific Daily Longline Fishing Log for recording effort, catch, and other data.
- NMFS Pacific Transshipment Declaration Form, if applicable.
- Marine Mammal Authorization Program Mortality/Injury Reporting Form.
- Vessel monitoring system.
- Vessel and fishing gear identification.

Notification Requirement and Observer Placement;

- Notify the PIRO Observer Program Contractor at least 72 hours before departing on a fishing trip to declare the trip type (shallow-set or deep-set). Once the fishing trip begins, the operator may make sets only of the declared type. A vessel is required to have a NMFS fishery observer on board if assigned by NMFS. Since 2010, NMFS placed observers on approximately 20% of DSLL trips.
- Fisheries observer guidelines provided must be followed.

Prohibited Areas in Hawaii (see Figure 2 and Figure 4);

- Main Hawaiian Islands Longline Fishing Prohibited Area (exclusion zone).
- Northwest Hawaiian Islands Longline Protected Species Zone. In 1991, the Council voted to permanently establish a "protected species zone" that closed the portion of the EEZ within 50 nm of the center geographical positions of certain islands and reefs in the Northwestern Hawaiian Islands (NWHI), listed in the table below. Where the areas are not contiguous, parallel lines drawn tangent to and connecting those semicircles of the 50-nm areas that lie between Nihoa Island and Necker Island, French Frigate Shoals and Gardner Pinnacles, Gardner Pinnacles and Maro Reef, and Lisianski Island and Peal and Hermes Reef, delimit the remainder of the NWHI longline protected species zone.
- Pacific Remote Islands Marine National Monument: Prohibited commercial fishing in the Monument, which includes all United States EEZ waters out to 200 nm around Wake and Jarvis Island and Johnston Atoll and out to 50 nm around Howland, Baker, Jarvis Islands, Kingman Reef, and Palmyra Atoll (Figure 3).
- Papahanaumokuakea Marine National Monument: Prohibited commercial fishing in the Monument. Fishing vessels entering and leaving the boundaries of the Papahanaumokuakea Monument are required to notify the Papahanaumokuakea Monument office (Figure 3; 50 CFR 404).



Map by: R. O'Conner, NOAA Fisheries Pacific Island Regional Office, 09-20-2018.

Figure 2. Fine scale map of the MHI fishing exclusion zone and critical habitats.

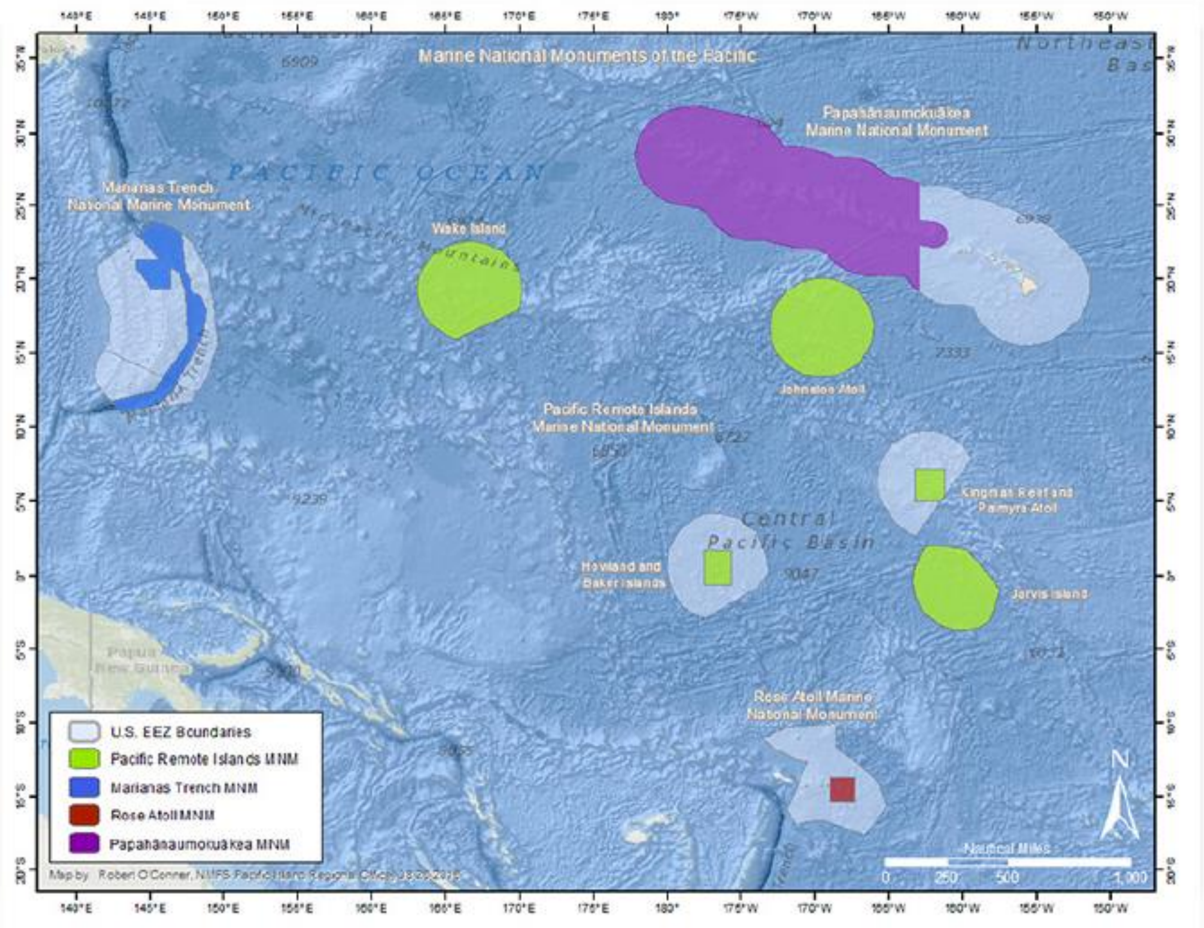


Figure 3. Boundaries of marine national monuments in the Pacific Islands where longline fishing is prohibited.

Protected Species Workshop;

- Each year, longline vessel owners and operators must complete a protected species workshop and receive a certificate.
- The vessel owner must have a valid protected species workshop certificate to renew a Hawaii longline limited entry permit.
- The vessel operator must have a valid protected species workshop certificate on board the vessel while fishing.

Sea Turtle, Seabird, and Shark Handling and Mitigation Measures;

- Vessel owners and operators are required to adhere to regulations for safe handling and release of sea turtles and seabirds.
- Vessel owners and operators must have on board the vessel all required turtle handling/dehooking gear specified in regulations.

- Vessel owners and operators can choose between side setting and stern setting, with additional requirements to reduce seabird interactions when fishing north of 23°N.
- All oceanic whitetip sharks and mobula rays caught in the WCPFC and the Inter-American Tropical Tuna Commission (IATTC) Convention Area, and all silky sharks caught in the WCPFC, must be released as soon as possible after catching them, taking reasonable steps for releasing them in a manner that will result in the least possible harm to the animal without compromising human safety.
- Hooked oceanic whitetip sharks must be brought to the side of the vessel and be released using a dehooker or line clippers to remove trailing gear from the animal. When using line clippers, cut the branch line as close to the hook as possible. Vessel owners, operators, and crew are required to release any oceanic whitetip shark and silky sharks; and take reasonable steps for their safe release.

Marine Mammal Handling and Mitigation Measures;

- Vessel owners and operators must follow the marine mammal handling guidelines provided at the protected species workshop.
- Vessel owners or operators must submit the marine mammal authorization program mortality/injury reporting form within 48 hours after the end of the fishing trip to NMFS to report injuries or mortalities of marine mammals (50 CFR 229.6). The crew must notify the operator if a marine mammal interaction occurs, so that the captain can supervise the handling and release.
- NMFS-approved marine mammal handling and release, as well as captain notification placards, must be posted on every vessel.

Gear Requirements for deep-set fishing north of the equator (0° latitude);

- Each float line must be at least 20 m (65 ft. 7 in) long.
- Attach at least 15 branch lines between two consecutive floats (basket gear requires at least 10 branch lines).
- No light sticks are allowed on the vessel.
- Any nylon monofilament line used in a branch line or leader must have a diameter (thickness) of 2.0 mm or larger. Any other line material used in a branch line or leader must have a breaking strength of 400 lb. or more.
- No metal wire line within 1 meter of the hook.
- Shark lines, a type of fishing gear used to target sharks, are prohibited.
- Use circle hooks with a maximum wire diameter of 4.5 mm. The hook shank must contain round wire that can be measured with a caliper. If the hook point is offset, it must be offset by no more than 10°.

Unless otherwise noted, the above regulations are at 50 CFR Part 665. A summary of regulations for Hawaii longline fisheries (SSLL and DSLL combined) is provided by the Summary of Hawaii Longline Fishing Regulations (NMFS 2020a).

In addition to the above, the Western and Central Pacific Fisheries Commission (WCPFC) adopted Conservation and Management Measure (CMM) 2019-05, which became effective on

January 1, 2021. NMFS' rule (88 FR 30671) on this CMM includes requirements for the handling of Mobulids;

- Prohibit targeted fishing or intentional setting on mobulid rays.
- Prohibit vessels from retaining, transshipping or landing any part or whole carcass of mobulid rays.
- Require vessels to promptly release mobulid rays, alive and unharmed, to the extent practical, as soon as possible and in a manner that will result in the least possible harm to the captured individual.

Under the proposed action, the Hawaii DSLL fishery also operates under a catch limit for bigeye tuna, established by the WCPFC, and implemented by NMFS through regulations at 50 CFR 300.224. When the United States fishery reaches the annual limit, NMFS prohibits the Hawaii DSLL and SSLL fisheries from retaining and landing bigeye tuna caught in the Western and Central Pacific Ocean (WCPO; west of 150° W). The Pelagics FEP also established a process under the authority of the Magnuson-Stevens Fishery Conservation and Management Act that allows NMFS to specify catch limits for pelagic fisheries, including bigeye tuna in the United States territories, and authorize the government of each United States territory to allocate a portion of its catch limit to United States fishing vessels permitted through specified fishing agreements (50 CFR 665.819). A limit for bigeye tuna in the eastern Pacific Ocean is established by the IATTC and implemented by NMFS through regulations at 50 CFR 300, applicable to vessels greater than 24 meters in length.

In 2021, NMFS specified a 2022 limit of 2,000 metric tons (t) of longline-caught bigeye tuna for each U.S. Pacific territory (American Samoa, Guam, and the Commonwealth of the Northern Mariana Islands (CNMI), the territories). NMFS also authorized each territory to allocate up to 1,500 t of its 2,000 t bigeye tuna limit in 2022 to U.S. longline fishing vessels through specified fishing agreements that meet established criteria. The overall allocation limit among all territories, however, may not exceed 3,000 t. As an accountability measure, NMFS would monitor, attribute, and restrict (if necessary) catches of longline-caught bigeye tuna, including catches made under a specified fishing agreement (86 FR 73990, December 29, 2021)

These catch limits and accountability measures are identical to those that NMFS has specified for U.S. Pacific territories in each year since 2014. The individual territorial allocation limit of 1,500 t is identical to what NMFS specified for 2020 and 2021. The overall allocation limit among all of the territories may not exceed 3,000 t for the year, which is consistent with previous years.

NMFS monitors catches of longline-caught bigeye tuna by the longline fisheries of each U.S Pacific territory, including catches made by U.S. longline vessels operating under specified fishing agreements. When NMFS projects that a territorial catch or allocation limit will be reached, NMFS, as an accountability measure, prohibits the catch and retention of longline-caught bigeye tuna by vessels in the applicable territory (if the territorial catch limit is projected to be reached), and/or vessels in a specified fishing agreement (if the allocation limit is projected to be reached).

In 2021, NMFS received separate specified fishing agreements between the CNMI and the Hawaii Longline Association (HLA) and between American Samoa and HLA, each of which included a specification of 1,500 t of bigeye tuna. NMFS began allocating catches to the CNMI on August 30, 2021, prior to the U.S. fishery reaching the WCPO bigeye tuna catch limit. Based on logbooks submitted by longline vessels, the CNMI allocation

The Hawaii DSLL fishery is a limited entry fishery, the Pelagics FEP authorizes 164 vessels to participate in the Hawaii longline fishery (DSLL or SSLL). Currently, NMFS has issued all 164 permits to eligible participants, however not all fishing permits are actively used each year. Therefore, the potential for increased participation in the fishery exists from the unused permits. In both 2020 and 2021, 146 vessels participated in the fishery but this number dropped to 144 in 2022 (see Table 11 in Section 3 – Environmental Baseline).

Prior to 2021, most vessels in the Hawaii DSLL fishery used steel wire leaders at the terminal portion of the branch line between the hook and the weighted swivel to reduce the risk of crew injuries resulting from the 'fly back' of weighted branch lines. On April 28, 2022, NMFS published a final rule prohibiting the use of wire leaders in the Hawaii DSLL fishery (87 FR 25153; 50 C.F.R. §§ 665.802(gg), (hh), 665.811). The rule also requires, with limited exceptions for safety and data collection, that fishermen remove fishing gear from any oceanic whitetip shark caught in all longline fisheries operating under the FEP to reduce the amount of fishing gear (trailing gear) attached to released oceanic whitetip sharks. The final rule specifically requires that the animals should be left in the water and a dehooker or line clippers shall be used to remove trailing gear.

1.2.1 Requirements of the False Killer Whale Take Reduction Plan

Under the proposed action, the Hawaii DSLL fishery is also subject to regulations implemented under the authority of the MMPA to conserve false killer whales (50 CFR 229). NMFS implemented the False Killer Whale Take Reduction Plan (FKWTRP) regulations on December 31, 2012 (77 FR 712590). Because the FKWTRP includes measures that affect the MHI IFKW, we discuss it here.

The FKWTRP implemented the following regulatory measures for the Hawaii DSLL fishery. All were effective on December 31, 2012, with the exception of the gear requirements, which went into effect on February 27, 2013:

- Requires circle hooks with 4.5 mm maximum wire diameter, sufficient round wire in the shank to be measured with a caliper, and 10 degree offset or less.
- Established a minimum 2.0 mm diameter for monofilament used in leaders or branch lines, and a minimum breaking strength of 400 pounds for any line used in the construction of a branch line if any other material is used.
- Established a year-round MHI longline fishing prohibited area in FKWTRP regulations, bounded by the same coordinates as the existing February-September boundary of the MHI Longline Prohibited Area (50 CFR 665.806(a)(2)). Under the authority of the MSA, regulations prescribing the pre-existing MHI were revised by NMFS to align boundaries of the MHI longline prohibited area and ensure existing applicable regulations for the longline fishery are consistent with FKWTRP and MMPA.
- Requires annual certification in marine mammal interaction mitigation techniques for longline vessel owners and operators.
- Requires posting of a marine mammal handling and release informational placard on longline vessels.
- Requires captains' supervision of marine mammal handling and release.

- Requires posting of a placard instructing crew to notify the captain of marine mammal interactions.
- Established a Southern Exclusion Zone (SEZ) and specific bycatch triggers for closure of this zone to the Hawaii DSLL (Figure 4).

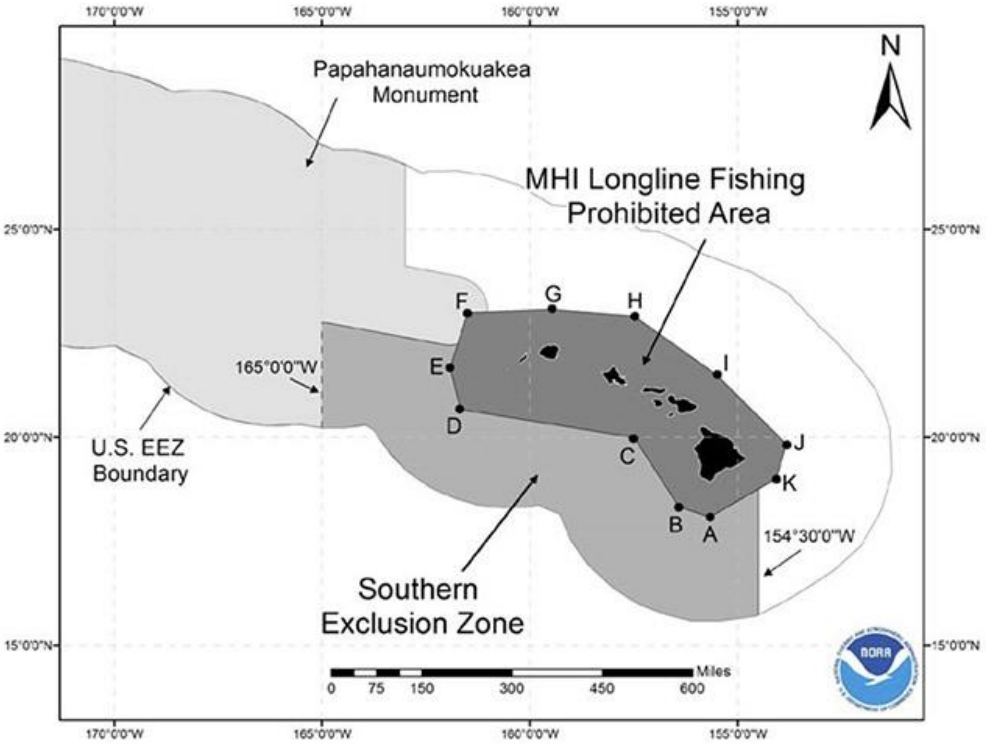


Figure 4. Map of the MHI longline fishing prohibited area, the FKWTRP southern exclusion zone, and the Papahanaumokuakea Monument.

The specified threshold level ("trigger") for closing the SEZ is calculated based on observed false killer whale mortalities or serious injuries in the DSLL fishery that occur in the EEZ around Hawaii. The trigger is calculated as the larger of these two values: (i) two observed mortality or serious injuries (M/SI) within the EEZ around Hawaii or (ii) the smallest number of observed false killer whale M/SI that, when extrapolated based on the percentage observer coverage in the deep-set longline fishery for that year, exceeds the Hawaii Pelagic false killer whale stock's potential biological removal level. In accordance with the FKWTRP regulations a closure of the SEZ is required through the end of the fishing year. The SEZ has been closed twice since implementation of the FKWTRP. The first closure of the SEZ occurred on July 24, 2018, and the SEZ was reopened on January 1, 2019. The SEZ was closed again on February 22, 2019, and reopened on August 25, 2020. In 2020, a new trigger was published to revise the trigger to four observed M/SI of false killer whales (85 FR 81184). In 2021, four observed mortalities or serious injuries of false killer whales occurred incidental to the Hawaii DSLL within the U.S. EEZ around Hawaii on January 18, 2021, March 26, 2021, April 17, 2021, and November 19, 2021. Because the injury determination of the fourth interaction meeting the trigger was not available until January 2022, the timeframe for closing the SEZ in 2021 had passed, and the SEZ was not closed.

1.3 Overview of NMFS Assessment Framework

Biological opinions address two central questions: (1) has a Federal agency insured that an action it proposes to authorize, fund, or carry out is not likely to jeopardize the continued existence of endangered or threatened species and (2) has a Federal agency insured that an action it proposes to authorize, fund, or carry out is not likely to result in the destruction or adverse modification of critical habitat that has been designated for such species. What follows summarizes how NMFS' generally answers these two questions; that is followed by a description of how this biological opinion will apply this general approach to the DSLL fishery.

Before we introduce the assessment methodology, we want to explain how we analyze an "effect." For this, we analyze the change or departure from a prior state or condition of a system caused by an action or exposure (Figure 5). Although Figure 5 depicts a negative effect, the definition itself is neutral: it applies it to activities that benefit endangered and threatened species as well as to activities that harm them. Whether the effect is positive (beneficial) or negative (adverse), an "effect" represents a change or departure from a prior condition (a in Figure 5); in consultations, the prior global condition of species and designated critical habitat is summarized in the Status of the Species narratives while their prior condition in a particular geographic area (the Action Area) is summarized in the Environmental Baseline section of this opinion. Extending this baseline condition over time to form a future without the project condition (line b in Figure 5); this is alternatively called a counterfactual because it describes the world as it might exist if a particular action did not occur. Although consultations do not address it explicitly, the future without the project is implicit in almost every effects analysis.

As Figure 5 illustrates, effects have several attributes: polarity (positive, negative, or both), magnitude (how much a proposed action causes individuals, populations, species, and habitat to depart from their prior state or condition) and duration (how long any departure persists). The last of these attributes—duration—implies the possibility of recovery which has the additional attributes recovery rate (how quickly recovery occurs over time; the slope of line **c** in the figure) and degree of recovery (complete or partial). For example, the recovery rate allows us to estimate how long it would take for a coral reef and associated benthic communities to recover.

As described in the following narratives, biological opinions apply this concept of effects to endangered and threatened species and designated critical habitat. Jeopardy analyses are designed to identify probable departures from the prior state or condition of individual members of listed species, populations of those individuals, and the species themselves. Destruction or adverse modification analyses are designed to identify departures in the area, quantity, quality, and availability of the physical and biological features that represent habitat for these species.

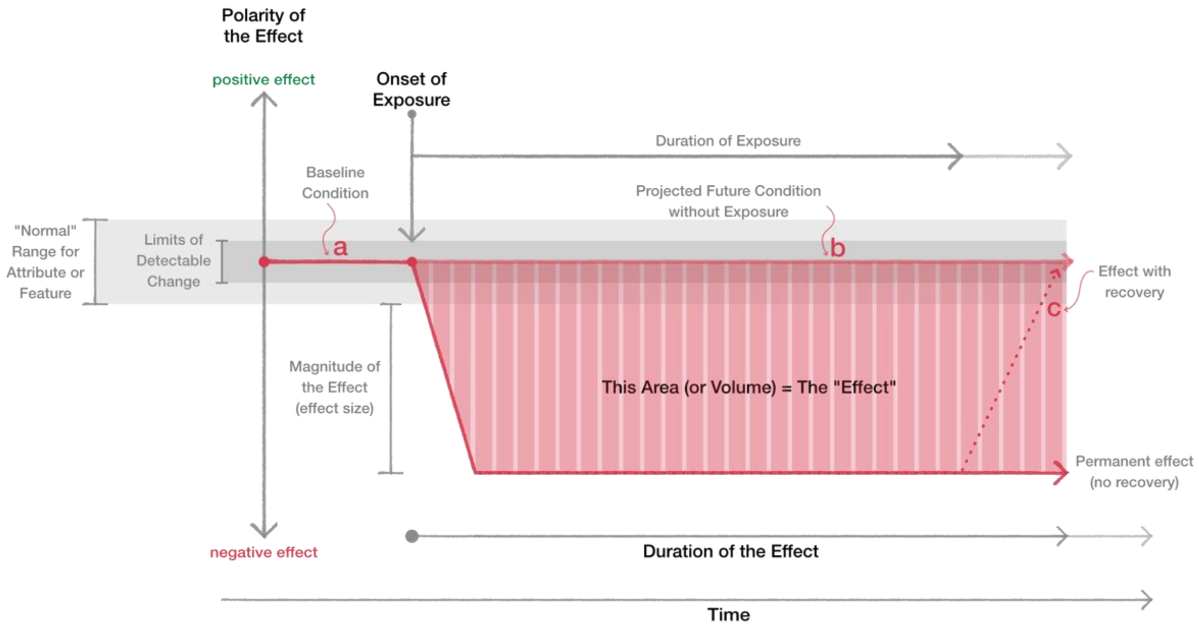


Figure 5. A schematic of the various elements encompassed by the word "effect." The vertical bars in the figure depict a series of annual "effects" (negative changes from a pre-existing or "baseline" condition) that are summed over time to estimate the action's full effect. See text for a more complete explanation of this figure.

1.3.1 Jeopardy Analyses

The ESA section 7 regulations define "jeopardize the continued existence of" as "to engage in an action that reasonably would be expected, directly or indirectly, *to reduce appreciably* the likelihood of *both the survival and recovery* of a listed species in the wild by reducing the *reproduction, numbers, or distribution* of that species" (50 CFR 402.02, *emphasis added*). The jeopardy standard is focused on the effects of the action when considered together with the species' status and all other threats acting on it. A federal action that adversely affects a declining population does not necessarily jeopardize that species unless the action itself is the cause of some active change of the species' status for the worse. See *National Wildlife Federation v. NMFS*, 524 F.3d 917, 930 (9th Cir. 2008). Minor reductions in the reproduction, numbers, or distribution of a species that are inconsequential at the species level will not be sufficient to jeopardize that species. In other words, a jeopardizing action requires that any reduction in the likelihood of survival or recovery be appreciable; i.e., material or meaningful from a biological perspective. See *Oceana v. Pritzker*, 75 F. Supp. 3d 469, 481-84 (DDC 2014)(holding that NMFS was within the bounds of its discretion to construe the word "appreciably" as entailing more than a bare reduction in the likelihood of survival and recovery, but rather "a considerable or material reduction in the likelihood of survival and recovery"). We note, however, that for a species that has a particularly dire pre-action condition, an action's even slight impacts may rise to the level of appreciable reduction (83 FR 35178, July 25, 2018). This definition requires our assessments to address four primary variables:

1. Reproduction
2. Numbers
3. Distribution
4. The probability of the proposed action will cause one or more of these variables to change in a way that represents an appreciable reduction in a species' likelihood of surviving and recovering in the wild.

Reproduction leads this list because it is "the most important determinant of population dynamics and growth" (Carey and Roach 2020). Reproduction encompasses the reproductive ecology of endangered and threatened species; specifically, the abundance of adults in their populations, the fertility or maternity (the number of live births rather than the number of eggs they produce) of those adults, the number of live young adults produce over their reproductive lifespans, how they rear their young (if they do), and the influence of habitat on their reproductive success, among others. Reducing one or more of these components of a population's reproductive ecology can alter its dynamics so reproduction is a central consideration of jeopardy analyses.

The second of these variables—numbers—receives the most attention in the majority of risk assessments and that is true for jeopardy analyses as well. Numbers or abundance usually represents the total number of individuals that comprise the species, a population, or a sub-population; it can also refer to the number of breeding adults or the number of individuals that become adults. For species faced with extinction or endangerment, several numbers matter: the number of populations that comprise the species, the number of individuals in those populations, the proportion of reproductively active adults in those populations, the proportion of sub-adults that can be expected to recruit into the adult population in any time interval, the proportion of younger individuals that can be expected to become sub-adults, the proportion of individuals in the different genders (where applicable) in the different populations, and the number of individuals that move between populations over time (immigration and emigration). Reducing these numbers or proportions can alter the dynamics of wild populations in ways that can reinforce their tendency to decline, their rate of decline, or both. Conversely, increasing these numbers or proportions can help reverse a wild population's tendency to decline or cause the population to increase in abundance.

The third of these variables—distribution— refers to the number and geographic arrangement of the populations that comprise a species. Jeopardy analyses must focus on populations because the fate of species is determined by the fate of the populations that comprise them: species become extinct with the death of the last individual of the last population. For that reason, jeopardy analyses may consider changes in the number of populations, which provides the strongest evidence of a species' extinction risks or its probability of recovery. Jeopardy analyses also may consider changes in the spatial distribution of the populations that comprise a species because such changes provide insight into how a species is responding to long-term changes in its environment (for example, to climate change). The spatial distribution of a species' populations also determines, among other things, whether all of a species' populations are affected by the same natural and anthropogenic stressors and whether some populations occur in protected areas or are at least protected from stressors that afflict other populations.

To assess whether reductions in a species' reproduction, numbers, or distribution that are caused by an action appreciably reduce the species' likelihood of surviving and recovering in the wild, NMFS' first assesses the status of the endangered or threatened species that may be affected by an action. That is the primary purpose of the narratives in the Status of Listed Resources sections of biological opinions. Those sections of biological opinions also present descriptions of the number of populations that comprise the species and their geographic distribution. Then NMFS' assessments must consider the status of those populations in the Environmental Baseline section. The baseline condition of the populations and individuals in an Action Area determines their probable responses to future actions.

To assess the effects of actions considered in biological opinions, NMFS' consultations use an exposure–response–risk assessment framework. The assessments that result from this framework begin by identifying the physical, chemical, or biotic aspects of proposed actions that are known or are likely to have individual, interactive, or cumulative direct and indirect effects on the environment (we use the term "potential stressors" for these aspects of an action). As part of this step, we identify the spatial extent of any potential stressors and recognize that the spatial extent of those stressors may change with time. The area that results from this step of our analyses is the Action Area for a consultation.

After they identify the Action Area for a consultation, jeopardy analyses then identify the listed species and designated critical habitat (collectively, "listed resources"; critical habitat is discussed further below) that are likely to occur in that Action Area. If we conclude that one or more species is likely to occur in an Action Area when the action would occur, jeopardy analyses try to estimate the number of individuals that are likely to be exposed to stressors caused the action: the intensity, duration, and frequency of any exposure (these represent our exposure analyses). In this step of our analyses, we try to identify the number, age (or life stage), and gender of the individuals that are likely to be exposed to an action's effects and the populations or subpopulations those individuals represent.

Once we identify the individuals of listed species that are likely to be exposed to an action's effects and the nature of that exposure, we examine the scientific and commercial data available to determine whether and how those individuals are likely to respond given their exposure (these represent our response analyses). Our individual-level assessments conclude with an estimate of the probable consequences of these responses for the "fitness" of the individuals exposed to the action. Specifically, we estimate the probability that exposed individuals will experience changes in their growth, development, longevity, and the number of living young they produce over their lifetime. These estimates consider life history tradeoffs, which occur because individuals must allocate finite resources to growth, maintenance and surviving or producing offspring; energy that is diverted to recover from disease or injury is not available for reproduction.

If we conclude that an action can be expected to reduce the fitness of at least some individuals of threatened or endangered species, our jeopardy analyses then estimate the consequences of those changes on the viability of the population(s) those individuals represent. This step of our jeopardy analyses considers the abundance of the populations whose individuals are exposed to an action; their prior pattern of growth and decline over time in the face of other stressors; the proportion of individuals in different ages and stages; gender ratios; whether the populations are "open" or "closed" (how much they are influenced by immigration and emigration); and their ecology (for example, whether they mature early or late, whether they produce many young or a small number of them, etc.). Because the fate of species is often determined by the fate of the population(s) that comprise them, this is a critical step in our jeopardy analyses.

Our risk analyses normally conclude by assessing how changes in the viability of populations of threatened or endangered species affect the viability of the species those populations comprise (measured using probability of demographic, ecological, or genetic extinction in 10, 25, 50 or 100 years). For the analysis in the biological opinion, we used 40 years for all species (except West Pacific leatherback sea turtles) because we have adequate information about population trends, interactions rates, and effects from interactions to reasonably predict the action's effects over that timeframe. For West Pacific leatherback sea turtles, we note 1) the population growth rate estimated by Martin et al. (2020a; -6.0% [95% CI: -24.1 to 12.2%]) represents a 36% spread and includes both positive and negative growth rates, and 2) the recent nest numbers from the Jamursba Medi and Wermon index beaches (Lontoh et al. In Prep, see Figure 24) show stable numbers of annual nests from 2017 to 2021, inconsistent with the projected mean decline of Martin et al. (2020a, though within the 95% CI) and used in the PVA of Siders et al. (2023).

New nesting data suggest a somewhat more favorable, stabilizing trend than the population growth rates projected by Martin et al. (2020a) and that was used by Siders et al. (2023) to estimate 2021 population sizes. However, this data is not yet available in sufficient detail that will allow an update the growth trend analysis of Martin et al. (2020a). To date we have only received the graph shown in Figure 24 and detailed data on nest counts per month are needed to update the model. In addition, the data from 2012 to 2021 that show stable to increasing trends are likely not of sufficient duration compared to the dramatic declines from the 1980's to 2012 to update our understanding of the current status of this population. From the 1980's to 2013, nesting at the index beaches declined by a factor of 13 (nest numbers in the 1980's were about 13 times higher than in 2013). In contrast, from 2013 to 2021, nesting at the index beaches increased by a factor of about 2. While this is encouraging, it does not ameliorate the historic declines. Thus, we consider the modeling results of Martin et al. (2020a) as the best available estimates for long-term West Pacific leatherback population trends, while acknowledging the uncertainty around these estimates and the need to update that model with the new data once it is available.

The resulting uncertainty of the current trend limits our ability to reliably predict the population status beyond 20 years, and even within those 20 years our assumed mean decline of 6% per year may be overly pessimistic. This causes collateral high uncertainty in our ability to predict the action's effects with reliability over longer timeframes, thus we used a 20–year analytic period for West Pacific leatherback turtles.

This step of our analyses considers data available on the particular populations and species affected by an action. However, this step of our analyses can also be informed by empirical information on (1) species that have become extinct—they became endangered but did not "survive" endangerment and, therefore, could not "recover" from it; (2) species whose abundance and distribution has declined and collapsed but whose future—their likelihood of continuing to persist over time (survive) or recovering them from endangerment—remains uncertain; (3) species that have declined and collapsed, but have begun the process of recovering from endangerment although they have not yet "recovered" in the wild; and (4) species that have survived endangered and subsequently recovered from it. The second of these categories includes species that have been extinct in the wild, but "survive" in captivity.

Destruction or adverse modification analyses

The section 7 regulations define "destruction or adverse modification" as "a direct or indirect alteration that appreciably diminishes the value of critical habitat for the conservation of a listed species. Such alterations may include, but are not limited to, those that alter the physical or biological features essential to the conservation of a species or that preclude or significantly delay development of such features" (50 CFR 402.02). This definition focuses on how federal actions affect the quantity, quality, and availability of the physical or biological features of the designated critical habitat.

NMFS uses the same exposure–response–risk assessment framework for designated critical habitat that it uses for jeopardy analyses. Exposure analyses first determine if designated critical habitat occurs in the Action Area for a consultation. If it does, those analyses identify the physical or biological features of critical habitat that are likely to be exposed to an action's effects.

Our analyses then consider how those features are likely to respond to that exposure, which requires us to consider the habitat's probable condition when the exposure occurs (that is, the impact of the Environmental Baseline on the value of the habitat); the ecology of the habitat at the time of exposure; where the exposure is likely to occur; and when the exposure is likely to occur; and the intensity, duration, and frequency of exposure.

If our analyses lead us to expect the quantity, quality, or availability of the physical or biological features of an area of designated critical habitat will decline because of a proposed action, we ask initially if those reductions are likely to be sufficient to reduce the value of the designated critical habitat for the conservation of listed species in the Action Area. By value, we mean the probability that the habitat designated in the Action Area will be occupied by and provide utility to individuals of the endangered or threatened species it was designated to help conserve. In this case, occupancy only means that individuals of the species are likely to use the habitat, even if they only use it intermittently; utility means that the individuals that occupy the habitat receive measurable improvement in their fitness (as defined earlier) as a result of using the habitat.

NMFS' destruction or adverse modification analyses are based on whether any reductions in the value of designated critical habitat in an Action Area is likely to be sufficient to reduce the value of the entire critical habitat designation. In this final step of our assessment, we combine information about the essential features of critical habitat that are likely to experience changes in quantity, quality, and availability given exposure to an action with information on the physical, chemical, biotic, and ecological processes that produce and maintain those constituent elements in the Action Area. We use the conservation value of the entire designated critical habitat (as described in the Status of the Listed Resources) as our point of reference for this comparison.

1.4 Application of this Approach in this Consultation

NMFS has identified several aspects of the DSLL fishery and fishing vessels that represent potential stressors to threatened or endangered species or designated critical habitat. The term stressor means any physical, chemical, or biological change that can induce a direct or indirect effect on the environment (Action Area) or that can induce an adverse response on threatened or endangered species and their critical habitat. Sources of the stressors are primarily vessels and vessel operations, and gear use. The specific stressors addressed in this consultation include:

- Interaction with, including incidental capture of, non-target species, listed species, or their prey,
- Derelict gear;
- Introduction of oily discharges, cardboard, plastics, and other waste into marine waters;
- Collisions with vessels;
- Vessel noise; and
- Vessel emissions.

1.4.1 Action Area

The Action Area includes all areas to be affected directly or indirectly by the Federal action and not merely the immediate area involved in the action (50 CFR §402.02). For this consultation, the action area encompasses all areas where DSLL vessels operate, including their fishing areas (Figure 6) and vessel paths during transits. The Hawaii DSLL fishery generally occurs within 300-400 nm of the MHI, primarily between the equator and 35° N with 98.6% of sets occurring within that zone. Historically, most of the fishing effort occurred south of Hawaii, but in recent years, there has been considerable fishing north of the Hawaiian Islands (NMFS 2018b).

All Hawaii-based DSLL vessels operate out of Honolulu Harbor. However, some DSLL trips originate from other ports such as Long Beach or San Francisco, California; or Pago Pago, American Samoa. We include the paths from these ports to the fishing grounds. Fishermen departing from California begin fishing on the high seas, outside the United States EEZ. Fishermen departing from American Samoa usually begin fishing near the Equator or in the North Pacific where they expect higher catch rates of bigeye tuna.

The Action Area also includes waters where vessels travel to and from fishing grounds but are prohibited (by Federal regulations and other applicable laws) from longline fishing, including the United States EEZ seaward of the State of California (50 CFR 660), the Papahanaumokuakea Marine National Monument (50 CFR 404; Presidential Proclamations 8031 and 9478), Pacific Remote Islands Marine National Monument and the MHI Longline Fishing Prohibited Area, which prohibits longline fishing year-round (50 CFR 665).

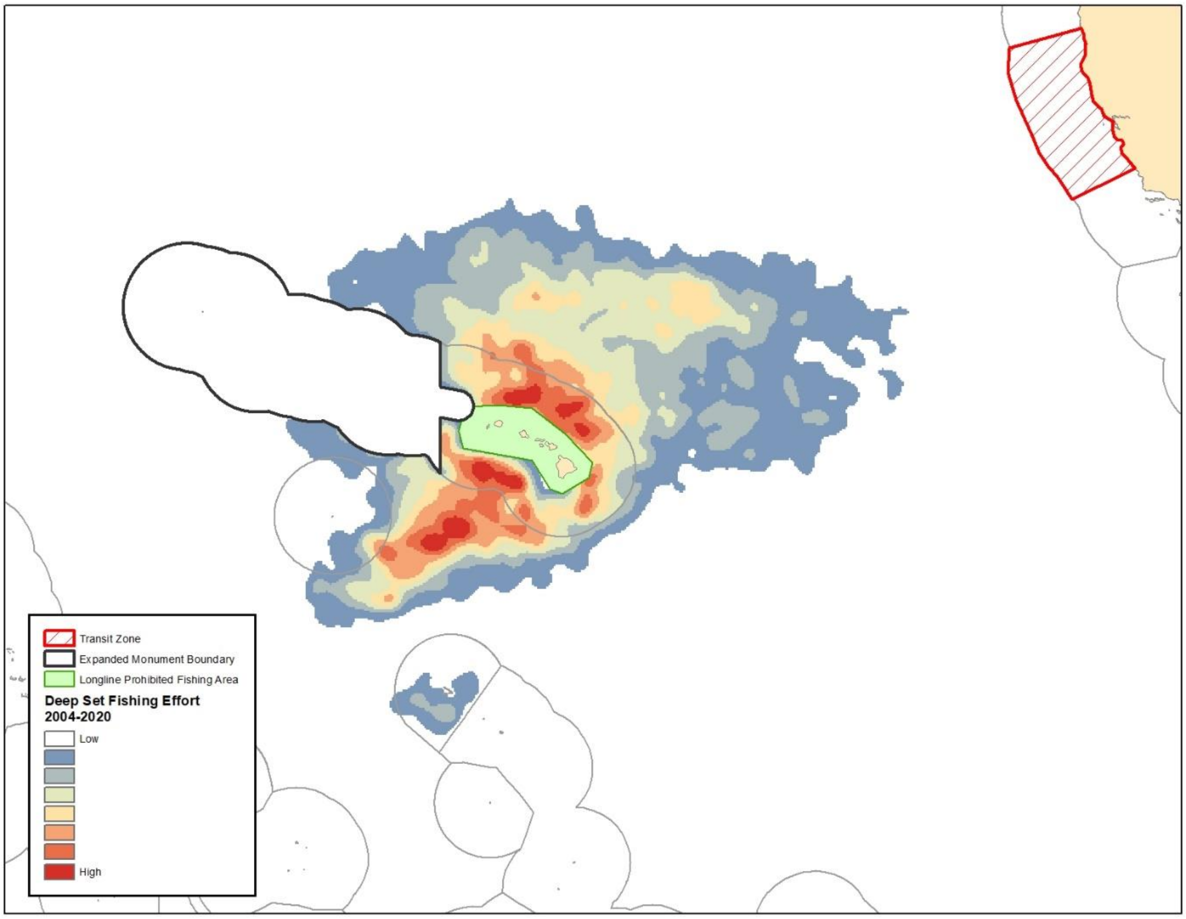


Figure 6. Location of Hawaii DSLL fishery as illustrated by effort (observed sets) from 2004 to 2020 (*n*= 65,245 sets). Colors from blue to red illustrate areas of lower to higher numbers of sets.

1.5 Approach to Evaluating Effects

After identifying the Action Area for this consultation, we identified those activities and associated stressors that are likely to co-occur with (a) individuals of endangered or threatened species or areas designated as critical habitat for threatened or endangered species; (b) species that are food for endangered or threatened species; or (c) species that prey on or compete with endangered or threatened species. The latter step represents our exposure analyses, which are designed to identify:

- The exposure pathway (the course the stressor takes from the source to the listed resource or its prey);
- The exposed listed resource (what life history forms or stages of listed species are exposed; the number of individuals that are exposed; which populations the individuals represent); and
- The timing, duration, frequency, and severity of exposure.

We also describe how the exposure might vary depending on the characteristics of the environment (for example, the occurrence of oceanic fronts or eddies) and seasonal differences in those characteristics, behavior of individual animals, etc. Our exposure analyses require knowledge of the action, and a species' population structure and distribution, migratory behaviors, life history strategy, and abundance.

We used available data to describe the Hawaii DSLL fishery location and its stressors. Interactions by hooking, entanglements, and landings represent the best data available on the Hawaii DSLL fishery because it has been collected under 20% observer coverage. From 2004 to 2019, annual observer coverage ranged from 19.6% to 26.0%, however, due to restrictions for NMFS deploying observers during the COVID 19 pandemic, observer coverage in 2020 was 15.2% and 17.84 % in 2021. Observer coverage for 2022 was 20.22%. In contrast, no data are available to characterize exposure to vessel strikes or discharges of waste in the DSLL.

We began considering the general location of exposure (coastal or pelagic), and whether there were unique temporal characteristics to their potential exposure (for instance, would exposure likely occur only when a vessel was transiting to and from harbor (e.g., IFKW critical habitat, which is wholly contained within the MHI longline exclusion zone)). We then evaluated the likelihood that each species would be exposed to the stressors described above. Where we concluded that the likelihood of exposure is extremely unlikely or the response will not rise to the scale of take, we do not include the stressor further in our exposure or response analyses (these determinations are explained in Appendix A). As a result, we focused our attention on the primary threat, the observed interactions, and characterizing the effects of those interactions on listed resources.

The stressors associated with the DSLL fishery produce responses that range from exposed but not adversely affected (such as opportunistic successful depredation of bait or catch), to accidentally being hooked and released alive unharmed, hooked and released injured, and death (immediate, or later in time following injury). Survival from injury is a function of an individual's prior health condition, environmental conditions, severity of injury, indicators of the severity of stress and injury (such as manner of capture, handling and release), and other variables (Swimmer and Gilman 2012; Hall and Roman 2013).

Figure 7 presents our conceptual model of how we translate an animal's interaction with the DSLL fishery into exposure, responses, and potential fitness consequences to the individual animals. In our response analyses, we present an extensive review of the best scientific and commercial data available to describe these relationships for each species with observed interactions in the DSLL fishery. Our response analyses (with information from our exposure analyses, and the Status of Listed Resources and the Environmental Baseline) provide the basis for how we characterize the risk that the proposed action poses those listed species and their designated critical habitat.

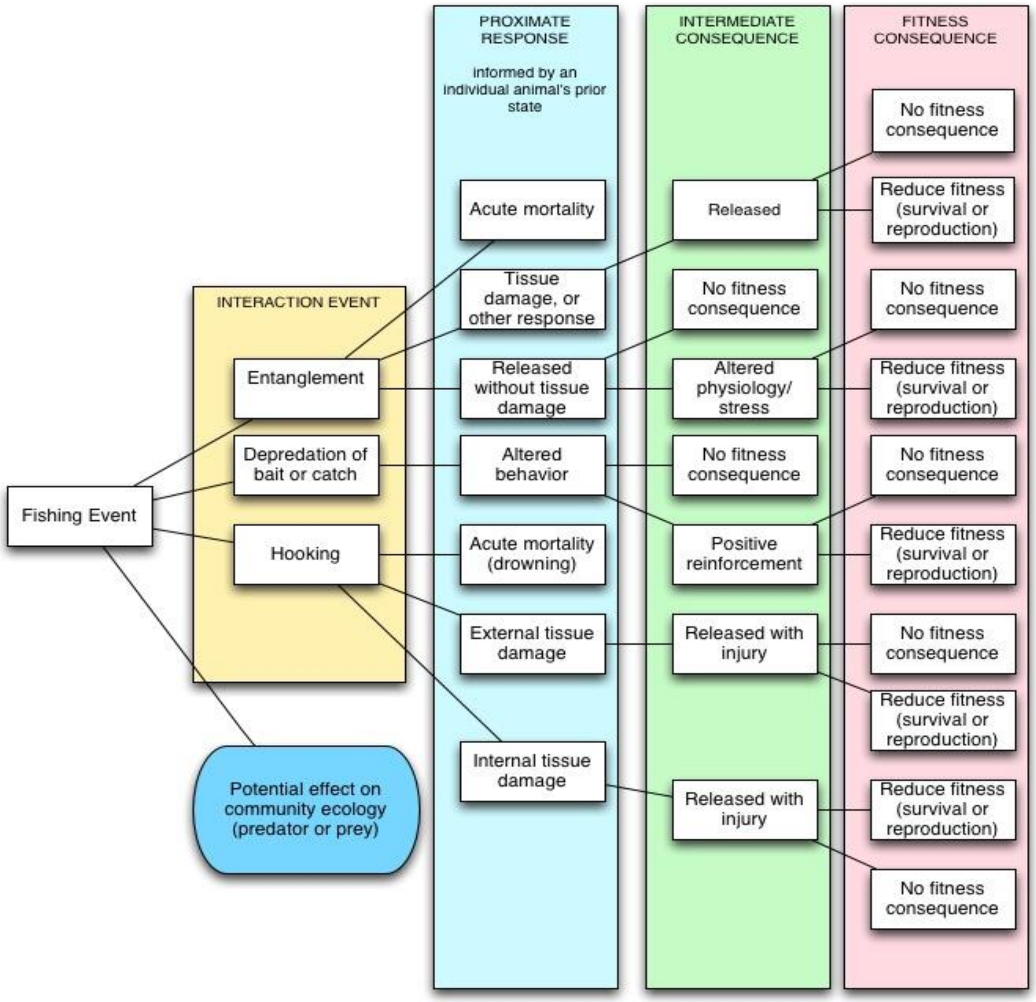


Figure 7. Conceptual model of the potential responses of an animal's exposure in the DSLL fishery and the pathways between exposure (the interaction event the individual animal has with the fishing gear), responses, and potential fitness consequences to the individual.

We lay the foundation for our risk assessment and our understanding of the animal's pre-existing physical, physiological, or behavioral state in the Status of Listed Resources and the Environmental Baseline using qualitative and quantitative analytical methods.

Next, we analyzed historic interactions rates, including the severity of those interactions, to inform our estimation of probable future interactions. The Hawaii-based longline fishery was split up to become the SSLL and DSLL fisheries in 2004. Therefore, the analysis presented in the biological evaluation (NMFS 2018b) used data from 2004 to 2017 to analyze effects of the action. We now have observed incidental captures in the DSLL through 2022, hence our analysis of the demographic, spatial and temporal effects of the fishery in this biological opinion focus on observed incidental captures from 2004 to 2022, as these are the years for which we have data and for which captures in the DSLL were separated from captures in the SSLL.

For estimated incidental captures (i.e. accounting for percent observer coverage) and for anticipated future incidental captures, McCracken (2019a) used data from 2002 to 2017 in her Bayesian inference model to estimate the anticipated annual mean and 95th percentile incidental captures based on the observed number of incidental captures given that the fishery has approximately 20% observer coverage. However, due to the COVID pandemic, observer coverage was less than 20% in 2020. While this full analysis has not been updated since then, McCracken and Cooper (2020a, 2020b) used the same model to estimate annual incidental captures based on observed captures for 2018-2019. Similarly, for sea turtles and sharks and rays, McCracken and Cooper (2021a, 2021b) applied the Bayesian inference model to estimate incidental captures in 2020 based on observed incidental captures. From these, we have a dataset of estimated captures from 2002 to 2021 however we note again that we focus our analysis from 2004 when the DSLL and SSLL were regulated as separate fisheries and therefore considered separate federal actions.

We now have finalized capture data through 2022 and have updated the estimates accordingly. In conducting our calculations, for the maximum 5-year running average and running sum (see Section 1.6.4), we used the estimated capture numbers from 2004 to 2022. However, we also report the annual mean and 95th percentile values from McCracken (2019a) with the caveat that they are based on data from 2002 to 2017.

Finally we note, during the course of this consultation we have continually included new data as they become available. All GIS mapping products relay data from 2004 to 2020. Data from 2021 and 2022 was not added due to the logistical constraints of rerunning all GIS products during the documents final review process. However, our inspection of the 2021 and 2022 data indicates they will not substantively change the GIS analyses presented herein.

1.5.1 Tipping Points

The Ninth Circuit Court of Appeals has said that in some circumstances, the Services are required to identify a ‘‘tipping point’’ beyond which the species cannot recover in making section 7(a)(2) determinations. Specifically, the Ninth Circuit has said that ‘‘when a proposed action will have significant negative effects on the species’ population or habitat, the duty to consider the recovery of the species necessarily includes the calculation of the species’ approximate tipping point.’’ *Oceana, Inc. v. Nat’l Marine Fisheries Serv*., 705 F. App’x 577, 580 (9th Cir. 2017) (citing *Nat’l Wildlife Fed’n v. Nat’l Marine Fisheries Serv*., 524 F.3d 917 (9th Cir. 2008)); see also *Wild Fish Conservancy v. Salazar*, 628 F.3d 513, 527 (9th Cir. 2010) (overturning jeopardy analysis based on purported NMFS failure to determine ‘‘when the tipping point precluding recovery … is likely to be reached’’). Neither the Act nor our regulations state any requirement for the Services to identify a ‘‘tipping point’’ as a necessary prerequisite for making section 7(a)(2) determinations. Section 7(a)(2) provides the Services with discretion as to how it will determine whether the statutory prohibition is exceeded. NMFS does not read the statutory language as requiring the identification of a tipping point, and the state of science often does not allow the Services to identify a ‘‘tipping point’’ for many species (84 FR 44987, August 27, 2019). NMFS has considered the Ninth Circuit’s guidance on this topic, and has determined that the circumstances in this consultation do not warrant the identification of tipping points for the subject species.

1.5.2 Simulations using Bayesian State-Space Population Viability Analyses for North Pacific Loggerhead Sea Turtles and the West Pacific Population of Leatherback Sea Turtles

A Bayesian state-space nesting trend analysis and Population Viability Analysis (PVA) was used to assess the current population status of two populations: the endangered West Pacific leatherback sea turtle and the endangered North Pacific loggerhead sea turtle to examine the long term viability of those populations (Martin et al. 2020a, 2020b); note that in this section we use the term 'population' to refer to both the North Pacific loggerhead sea turtle and the West Pacific leatherback sea turtle, throughout the rest of this biological opinion, the North Pacific loggerhead sea turtle is referred to as a species because that is how it is listed under the ESA). The model was subsequently updated with 2017 to 2021 DSLL capture data for leatherback sea turtles (Siders et al. 2023). The purpose of the analyses are to bring together current knowledge of the ecology of the species of concern using nest count data to forecast the population trends and extinction risks over defined time frames. The Bayesian analysis addresses some of the shortcomings of poor data, imprecise parameter estimates, and sampling and process variance in demographic parameters by exploring the full range of probability distributions for parameters of interest.

Specifically, the Bayesian state-space model estimated the population's mean log growth rate and variance in mean log growth (with 95% credible intervals) and the population's finite rate of increase (λ), also with 95% credible interval for the estimate. For the West Pacific leatherback sea turtle, some nesting data was missing and a Bayesian state-space model was used to impute the missing data prior to estimating the population's growth rate. The PVA model estimated: (a) the time until the population declines to 50%, 25%, and 12.5% of current abundance estimates; (b) the median time for each species to reach those thresholds; and (c) the species' probability of reaching those thresholds (50%, 25%, and 12.5% of current abundance estimates) in 5, 10, 25, 50, 100-year time intervals with associated 95% credible intervals.

Annual nest counts for each population were used as the foundational data for these analyses because this represents the most reliable long term census data for these populations. Ideally, we would have key demographic parameters on each of the populations to support robust analyses of abundance and trends, such as: age at maturity, survivorship across life stages, adult and hatchling sex ratios, clutch frequency, nesting success and remigration intervals for each subpopulation that comprise the population. However, comprehensive population-specific data sets are not available for either population.

Nest counts provide an index of female abundance for the specific nesting beaches, however assumptions must be made on clutch frequency and, because females do not return to nest every year, remigration interval to convert nest counts to adult female abundance. Results of the PVA are also only applicable to the portion of each population, or subpopulation, for which we have nest counts. No data are available to provide an index of abundance for other age classes. Due to this data limitation, estimating *r* from these data does not represent the true population growth rate for the population of interest but only the adult female portion of the population. We also note that the updated leatherback PVA of Siders et al. (2023) did not have new nesting data available to incorporate in the model and they therefore projected populations based on the end of the time series in Martin et al. (2020a; 2017) to the end of the updated fishery capture data time series (2021) to estimate annual numbers of nesting females.

Modeling assumptions made by Martin et al. (2020a, 2020b) and Siders et al. (2023; projections only) for the imputation, trend, and projections are as follows:

Imputation:

1. The true number of nests per month is distributed normally, where the monthly means can be modeled with a discrete Fourier series with a fixed annual frequency. In other words, the number of peaks within a year is constant over the years. The variance around the means is assumed constant over the years.

2. The cyclical nature of nesting throughout the year is sufficiently captured with an autoregressive model using a lag of one month (AR1 model) where the relationship between the numbers of nests in two months is modeled by the Fourier series.

Trend:

1. The proportion of total nests observed on a given beach is constant through time.

2. The time series of observed number of annual nesters are representative of the populations, as are the population growth rates observed for nesters. Note that index beaches are typically identified for long-term monitoring because they represent the population well.

3. The clutch frequency is constant through time.

4. The population growth rate is density-independent (i.e., no carrying capacity or Allele effects in the timeframe considered).

5. The population growth rate is the same across the different beaches (i.e., there is a single biological process resulting in a single trend).

6. The process and observation variances are estimable (with so few beaches, the prior has a lot of influence).

7. The index beaches were observed with constant effort or were effort-corrected by the data collectors prior to being provided to PIFSC to form the time series used in these analyses.

8. The remigration interval is constant through time.

Projections:

1. The population will face the same threats in the future as it did in the past. The PVA applies the population growth rate from the past, which was influenced by some particular magnitude of threats, to the future trajectory of the population. It does not

> account for the impacts of new or increasing threats, such as the loss of nesting habitat from climate change.[2]

The PVA model was extended to include a "mortality[3]" model to assess the effect of the DSLL fishery on North Pacific loggerhead sea turtles (Martin et al. 2020b) and the West Pacific population of leatherback sea turtles (Martin et al. 2020b; Siders et al. 2023). Martin et al. (2020b) and Siders et al. (2023) used turtle lengths and assigned at-vessel and post-interaction mortality rates from the observer data in the DSLL fishery to parameterize a multivariate normal distribution that characterized historical length-mortality interactions for the fishery, and this distribution was used to generate estimates for the length and mortality of turtles that the fishery would interact with in the future.

Due to the limited number of known lengths and assigned mortality rates for the DSLL fishery, the length-mortality distributions used in the model were much less informed (i.e., less data-rich) compared to the SSLL, thereby leading to greater uncertainty about historical and future population impacts. As a result, Martin et al. (2020b) and Siders et al. (2023) added stochasticity to the simulations for the population impact of both historical and future fishery interactions. Siders et al. (2023) updated the model of Martin et al. (2020b) and the impact of the DSLL on West Pacific leatherback sea turtles. Key updates included updating the model with observed and estimated leatherback captures from 2017 to 2021, along with subsequent updates to anticipated captures and fishery mortality rates and a revised approach to how maturity was assigned. Siders et al. (2023) increased the survival of smaller (younger) leatherback sea turtles by proportioning annual survival between juvenile and adult survival rates, with adult survival rates representing a larger proportion of total survival rates with increasing size (Siders et al. 2023).

Because the PVA considers only adult nesting females, each sea turtle interacting with the fishery was converted to an adult nester equivalent (ANE) to assess the impact to the population from, for adult-sized turtles, the loss of any adult females including the future productivity that female would have contributed to the population; and for juvenile-sized turtles, the future productivity that individual may have contributed to the population. Converting adult-sized turtles to ANE requires an estimate of the population sex ratio to determine the probability that the captured individual was female, and future productivity requires an estimate of adult survival rate and remigration interval. In addition to population sex ratio, adult survival rate and remigration interval, converting juvenile-sized turtles to ANE also requires an estimate of the time remaining until the turtle reaches maturity, and the probability that the turtle survives to maturity.

Martin et al. (2020a) details the size-at-age relationship, age at maturity, sex ratio, remigration interval and survival rates used for each population. Siders et al. (2023) used the same size-at-age relationship, sex ratio, remigration interval, and survival rates as Martin et al. (2020a, 2020b) but they increased the minimum size at maturity while allowing for the possibility of smaller turtles to be mature. To account for unobserved historical interactions in the DSLL fishery in the

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[2] Note that these are the assumptions used by Martin et al. (2020a, 2020b) and Siders et al. (2023). Our analyses in this biological opinion consider the impacts of climate change on all species adversely affected by the action.

[3] Referred to as "take" in Martin (2020b).

historical take analysis, estimates of total fishery and species specific interactions were taken from the Annual Stock Assessment and Fishery Evaluation Report for U.S. Pacific Island Pelagic Fisheries Ecosystem Plan 2018 (WPRFMC 2019 and McCracken references therein; Martin et al. 2020a) and updated information from 2019 to 2021 was taken from WPRFMC (2020, 2021, 2022), McCracken (2020), and McCracken and Cooper (2021a, 2022a), as well as information on size and release condition from the observer database. The estimated unobserved interactions (historical) were the difference between the estimated total interactions and the known observed interactions. Each individual of the anticipated interaction received a length and fishery mortality from a bivariate normal distribution, a back-calculated age with years until maturity based on a von Bertalanffy growth model, juvenile survival rate, sex, and a remigration interval. From this, the probability of being female, the probability of surviving until maturity (a combination of size and age at capture and juveniles survival rate), and the probability of the interaction resulting in a mortality were estimated to yield an ANE for each interaction (Martin et al. 2020a, 2020b).

In order to not double count the impacts of the fishery, the anticipated total historical captures and mortalities attributed to the fishery were ‘removed’ and new population growth rates, r, were calculated to estimate the population growth rate of each population without the fishery. The PVA model was then run both with and without the added fishery mortalities, and the metrics described above were compared between the mortality and no mortality models to assess the impact of the fishery. Those metrics include: (a) the time until the population declines to 50%, 25%, and 12.5% of current abundance estimates; (b) the median time for each species to reach those thresholds; and (c) the species’ probability of reaching those thresholds (50%, 25%, and 12.5% of current abundance estimates) in 5, 10, 25, 50, 100-year time intervals with associated 95% credible intervals (Martin et al. 2020b; Siders et al. 2023).

1.5.3 Climate Change

Future climate will depend on warming caused by past anthropogenic emissions, future anthropogenic emissions and natural climate variability. NMFS’ policy (NMFS 2016b) is to use climate indicator values projected under the Intergovernmental Panel on Climate Change (IPCC)'s Representative Concentration Pathway (RCP) 8.5 when data are available or best available science that is as consistent as possible with RCP 8.5. RCP 8.5, like the other RCPs, were produced from integrated assessment models and the published literature; RCP 8.5 is a high pathway for which radiative forcing reaches >8.5 W/m^2 by 2100 (relative to pre-industrial values) and continues to rise for some amount of time. Since NMFS (2016a), RCP have been updated to a new version called SSP5-8.5 (ICCP 2021). A few projected global values under SSP5 8.5 are noted in Table 1.

Presently, the IPCC predicts that climate-related risks for natural and humans systems are higher for global warming of 1.5 ºC but lower than the 2 ºC presented in Table 1 (IPCC 2021). Changes in parameters will not be uniform, and IPCC projects that areas like the equatorial Pacific will likely experience an increase in annual mean precipitation under scenario 8.5, whereas other mid-latitude and subtropical dry regions will likely experience decreases in mean precipitation. Sea level rise is expected to continue to rise well beyond 2100 and while the magnitude and rate depends upon emissions pathways, low-lying coastal areas, deltas, and small islands will be at greater risk (IPCC 2021).

Table 1. Projections for certain climate parameters under Representative Concentration Pathway 8.5 (values for temperature from Table 4.2 in IPCC 2021; values for sea level from Chapter 9 of IPCC 2021).

Projections	**Scenarios (Mean and likely range)**	
	Years 2041-2060 (mid-term projections)	**Years 2081-2100 (long-term projections)**
Global mean surface temperature anomalies **(ºC)**	1.7 (1.2-2.5)	4.0 (2.7-5.7)
Global mean sea level increase (m)	0.23 (0.20-0.29)	0.77 (0.63-1.01)

Given the limited data available on sea turtle populations, and other listed species like whales, sharks, and rays that are adversely affected by the proposed action, and the inherent challenges with creating population models to predict extinction risks of these species, we are not inclined to add more uncertainty into our assessment by creating climate models with little data to parameterize such models. Since trying to apply a climate based model in 2012 to the SSLL biological opinion (NMFS 2012b), we have learned a few key important lessons: the climate based model incorporating fixed age (lag) is unrealistic given variability ages at sexual maturity for loggerhead and leatherback sea turtles, and fails to consider variation in age of the nesting cohort; studies have shown juvenile loggerhead sea turtles are distributing more widely than thought, and thus are likely impacted in ways not considered under the previous model.

A new dispersion model on leatherback sea turtles suggest they too may be dispersing more broadly, and affected differently than previously considered; the model did not account for impacts to more than two life-stages; and arguably, most importantly, the models did not perform as expected because the predictions were wrong for leatherback sea turtles the majority of the time, and predictions for loggerhead sea turtles were wrong half the time (Kobayashi et al. 2008, 2011; Van Houtan 2011; Van Houtan and Halley 2011; Allen et al. 2013; Arendt et al. 2013; Briscoe 2016a, 2016b; Jones et al. 2018; see also Jones memo 2018). Instead, in this assessment we rely on systematic assessments of available and relevant information to incorporate climate change in a number of ways.

We address the effects of climate, including changes in climate, in multiple sections of this assessment: Status of Listed Resources, Environmental Baseline, and Integration and Synthesis of Effects. In the Status of Listed Resources and the Environmental Baseline we present an extensive review of the best scientific and commercial data available to describe how the listed species and its designated critical habitat is affected by climate change—the status of individuals, and its demographically independent units (subpopulations, populations), and critical habitat in the Action Area and range wide.

We do this by identifying species sensitivities to climate parameters and variability, and focusing on specific parameters that influence a species health and fitness, and the conservation value of their habitat. We examine habitat variables that are affected by climate change such as sea level rise, temperatures (water and air), and changes in weather patterns (precipitation), and we try to assess how species have coped with these stressors to date, and how they are likely to cope in a changing environment. We look for information to evaluate whether climate changes affects the species' ability to feed, reproduce, and carry out normal life functions, including movements and migrations.

We review existing studies and information on climate change and the local patterns of change to characterize the Environmental Baseline and Action Area changes to environmental conditions that would likely occur under RCP 8.5, and where available we use changing climatic parameters (magnitude, distribution, and rate of changes) information to inform our assessment. In our exposure analyses, we try to examine whether changes in climate related phenomena will alter the timing, location, or intensity of exposure to the action. In our response analyses we ask, whether and to what degree a species' responses to anthropogenic stressors would change as they are forced to cope with higher background levels of stress cause by climate-related phenomena.

1.5.4 Statistical Intervals and Anticipated Future Captures Used in this Assessment

Throughout our assessment, we calculate a variety of metrics to evaluate the status of the species and the effects of the action. Where we have data to develop statistical inferences, we relied on the credible interval or the confidence interval to estimate the probability and the uncertainty of the estimate. Confidence intervals were used to capture the level of confidence that the true value (i.e., take level) is within a specified range of values (confidence interval) and, more specifically, to describe the uncertainty with the statistic. Bayesian predictions were used to estimate the credible interval, or the probability that the true value (i.e., interaction level) is within a specified range of values (credible interval).

Section 7(a)(2) of the ESA requires federal agencies to insure that any action they authorize, fund, or carry out is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of critical habitat of such. As part of this process, agencies are required to base their conclusion on the best scientific and commercial data available (16 U.S.C. 1539(a)(2)).

For the purpose of insuring that the action satisfies the requirements of section 7(a)(2) of the ESA, our analysis focuses on a range of likely outcomes, including the mean, the maximum 5 year running average, and the 95% percentile as described below, to account for our confidence in the underlying data and the inherent uncertainty in our projections. Our assessment relies on the mean and 95th percentile which corresponds to the upper end of the 90% credible interval, or the range between the 5th and 95th percentiles of the distribution around the mean to estimate the anticipated future interactions as well as the maximum 5-year running average of interactions. The mean values represent the number of interactions that we expect per year on average over time, and provides the most reliable estimate of the expected number of interactions between the fishery and the listed species over an extended time frame. However, in any one year, higher or lower numbers of interactions are likely to occur and we would expect the mean to be exceeded approximately 50% of the time.

The maximum 5-year running average captures these higher years by considering average interactions over shorter timeframes and identifying the timeframe with the highest running average. This has the effect of buffering the mean to a slightly higher number in most circumstances. Focusing our risk analysis on the maximum 5-year running average rather than the mean ensures we have assessed the potential impact to the species for changes in environmental conditions, location of fishery effort, and magnitude of fishery effort that may temporarily increase incidental captures of the species considered in this biological opinion. The upper 95th percentile values represents our upper estimate of interactions in any given year. The 95th percentile conveys that, based on the data, there is a 95% probability that the value will be lower in any year. In terms of the number of interactions, this means that there is a 95% probability that the true number of animals incidentally captured or killed is within the credible interval. While we base our analyses on these three metrics, we present subsets of key results that best describe the level of impact over short- and long-term timeframes.

To calculate the maximum 5-year running average, we first calculate 5-year running averages from 2004 to 2019 on the time series of estimated annual incidental captures from McCracken (2019a) and McCracken and Cooper (2020a, 2020b, 2021a, 2021b, 2022a, 2022b). In other words, our first 5-year running average is the average of the estimated incidental captures from 2004 to 2008, and our last value is the average of incidental captures from 2016 to 2021. From that resulting time series of running averages, we find the maximum value and that is the maximum 5-year running average. Multiplying the maximum 5-year running average by five gives the maximum 5-year running sum. We used both of these metrics, maximum 5-year running average and sum, along with the mean and 95th percentile values from McCracken (2019a) in our assessment of the impact of the fishery on the species considered in this biological opinion.

Given the year-to-year variability in the number of captured ESA-listed species, a single year estimate of captures that is not routinely exceeded would need to be a large value, such as the 95th percentile. However, using the 95th percentile to describe fishery impacts to ESA-listed species would lead to estimates of impacts over longer time periods that are exaggerated and not reasonably likely to occur. Because of this limitation, recent fisheries biological opinions have relied on rolling 5-year estimates, such as the maximum 5-year running sum described above, to describe the likely impact of fishery captures on ESA-listed species (see NMFS 2020c, 2021d, 2021e as examples).

Our assessment does not focus on the lower limits of the 95% confidence intervals because they represent lesser risk to endangered and threatened species; nevertheless, values between the mean and the lower limit 95% intervals are also compatible with the data and are not ignored in our analyses. However, for risk analyses over longer time intervals (for example, 40 years) we rely on largely on the mean estimates and 5-year maximum running averages because long time series are most likely to include values above and below the mean.

As noted earlier, we also understand that the fishery has been increasing the number of hooks in the water overtime and this effort is expected to continue into the future (NMFS 2018b). The modeling done by McCracken (2019a) to estimate anticipated annual incidental captures based on historic observed incidental captures does not explicitly account for effort. This is because the distribution of fishing operations change seasonally and from year to year and therefore the number of hooks deployed in a given year does not necessarily increase a species' chance of bycatch (McCracken 2019a). Therefore the estimated number of incidental captures based both on McCracken (2019a) and on the maximum 5-year running average described above inherently assume that the characteristics of the fishery will not change in the future compared to the historical period of 2004 to 2022.

As we cannot assume that the anticipated increase in hooks per year will not increase the likelihood of a species' capture, we compared hooks per year and observed incidental captures for each species as part of our analysis to assess if there is a statistically significant relationship between these two metrics. We did not detect a meaningful relationship between fishing effort (e.g., number of hooks in the water) and captures of oceanic whitetip shark, Indo-West Pacific scalloped hammerhead shark, giant manta ray, leatherback sea turtles, and loggerhead sea turtles. Captures of sperm whales and MHI IFKW are so rare (1 and 0 observed captures from 2004 to 2022 respectively) that we did not analyze these species. We did find statistically significant relationships between annual number of hooks and annual number of captures for green and olive ridley sea turtles. The lack of a relationship for most species was not particularly surprising because our time series is rather short (17 samples) and we lacked a number of relevant covariates (variables other than number of hooks). In addition, the lack of statistical significance should not be interpreted to mean that an increasing number of hooks in the water is not meaningful either, in fact as hooks are increased it would seem reasonable that the probability of capture should go up. However, the relationship is multifaceted and we lack information to incorporate future growth in the fishery into our analysis. Hence, there may be a more significant relationship between hooks and captures than we are currently not able to detect for most species and we assert that this will need to be carefully monitored as the fishery effort continues to increase into the future.

For the species where we did detect a significant correlation, we stress that this is a simple correlation that does not necessarily imply causation. Hooks per year has a strong trend and correlation with year, and, considering the increasing trends for East Pacific and Central North Pacific green sea turtles as well as olive ridley sea turtles, we may simply be seeing increased densities of these species leading to more captures, or other changes in their distribution due to environmental correlates that are leading to higher captures rates. Regardless, we address the potential impacts of these correlations in the species-specific exposure sections.

Finally, constraints in the available data limit our long-term predictive ability. Most importantly, random unpredictable events (positive or negative) may result in relatively large population changes. Furthermore, over the next couple decades, many population conditions and habitat parameters will change, making long-term projections of abundance and fishery effects unreliable. For these reasons, we focus our analysis over the next 40 years, except for the West Pacific population of leatherback sea turtles for which we use 20 years. We are not reasonably certain we can reliably predict the effects of the fishery's continued operations beyond that period of time. We have, however, carried the effects of the fishery during the analytical period into the future, accounting for the future effects of animals killed during the analytical period. We also anticipate the fishery to continue into the future for at least 40 years. We consider this reasonable as the fishery has been operating as the DSLL for 18 years and for decades prior to the deep-set portion being designated as a separate fishery.

1.5.5 Evidence Available for this Consultation

Section 7(a)(2) of the ESA and its implementing regulations require NMFS to use the best scientific and commercial data available during consultations. The most credible and relevant data available for our exposure and response analyses are (1) data NMFS' observer program collects on interactions between the DSLL fishery and endangered or threatened species collected as bycatch between 2004 and 2021 and (2) the Pacific Islands Fisheries Science Center's analyses of these data (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2021b, 2022a, 2022b; Martin et al. 2020a, 2020b). We supplemented the data from these two sources with information contained in SFD's 2018 Biological Evaluation on the Hawaii

Deep-set Pelagic Longline Fishery (NMFS 2018b), and the Western Pacific Regional Fishery Management Council's Annual Stock Assessment and Fishery Evaluation Reports (WPRFMC 2019, 2020, 2021, 2022). These data represent credible and relevant commercial data. As a result, these represent the best scientific and commercial data available at the time of consultation.

To support our status assessments, assessments of the expected impacts of the environmental baseline on endangered and threatened species and designated critical habitat, response analyses, and risk assessments, we relied on NMFS' marine mammal stock assessment reports, the 2016 Report of the Rare Events Bycatch Workshop Series (WPRFMC 2016), the Bycatch Management Information System (BMIS), and recovery plans that have been developed for the endangered or threatened species that may be affected by the DSLL fishery. We supplemented these sources with electronic searches of literature published in English or with English abstracts to cross search multiple databases for relevant scientific journals, open access resources, proceedings, web sites, doctoral dissertations and master's theses. Particular databases we searched for this consultation included Google Scholar, Bielefeld Academic Search Engine (BASE), CORE, Bing, Microsoft Academic, Science Direct, Web of Science, Science.gov, and JStor (to identify older studies) with targeted searches of websites for the journals Copeia, Marine Biology, Marine Ecology Progress Series, Marine Pollution Bulletin, Public Library of Science - Biology (PLoS Biology), and Public Library of Science - One (PLoS One).

We conducted literature searches to collect general information we needed to support the analyses that we present in the Status of the Species, Environmental Baseline, Effects of the Action, and Cumulative Effects sections of this biological opinion. We also conducted multiple literature searches to address a set of specific questions:

1. What is the population structure of the endangered and threatened species this consultation considered? Specifically, we targeted information that would allow us to identify the number of populations and sub-populations that comprise the endangered and threatened species this consultation considers.
2. What effects have been reported for the endangered and threatened species this consultation considers after interactions with longline gear? We were specifically interested in effects reported from the Pacific, but we also collected any at-vessel, post release, and overall or total mortality rates we could identify that may be pertinent to the species under consideration. We considered surrogate species as well when species-specific information was not available.
3. What, if any, patterns are available in the literature for the endangered and threatened species considered in this consultation after an interaction with different types of longline gear? We were specifically interested in differences between gear types and operational characteristics of fisheries that use either monofilament and/or wire leaders that may inform our effects analysis. We considered related surrogate species when species-specific information was not available.
4. What post-release changes in reproductive variables have been reported for the endangered and threatened species this consultation considers after interactions with longline gear? We were specifically interested in data on the effects of capture myopathy and stress pathology on the interval between reproductive events and natality (live births) in any of the endangered and threatened species this consultation considers.

For our literature searches, we used paired combinations of the keywords: “sea turtles,” “Pacific,” “*Dermochelys*,” “*Caretta*,” “*Chelonia,” “Eretmochelys,” Lepidochelys,” “Manta,”* “*Manta birostris,*” “*Mobula*,” “*Mobula birostris,*” “*Carcharhinus longimanus*,” “*Sphyrna lewini*,” “oceanic whitetip,” “scalloped hammerhead,” “elasmobranchs,” “shark,” “life history,” “population structure,” “population trend,” “demography,” “vital rates,” “bycatch,” “longline,” “at-vessel mortality,” “post-release mortality,” “haulback,” “fishery impacts,” and “climate change.” These keyword pairs captured the majority of relevant hits; however, to identify additional sources of relevant data and other information, we also included the following keywords in additional searches “cryptic mortality,” “unaccounted mortality,” “unobservable mortality,” “survivorship,” “precatch loss,” “slipped catch,” “circle hooks,” “bait,” “hot spots,” “shark finning,” “marine debris,” “micro plastic,” “plastic ingestion,” “debris entanglement,” “climate change,” “increased sand temperature and sex determination,” “habitat loss,” “prey availability,” “age to maturity,” “tag retention,” “egg mortality.” For giant manta rays and leatherback sea turtles, we conducted separate searches for data on environmental variables that explain or are correlated with their pelagic distribution, using the keywords “environmental correlates” and “manta ray” or “*mobulid*” or “leatherback” or “sea turtle” and “distribution.” We recognize this is not an exhaustive list of all resources that were referenced.

Electronic searches have important limitations. First, often they only contain articles from a limited time span (e.g., First Search only provides access to master’s theses and doctoral dissertations completed since 1980 and Aquatic Sciences and Fisheries Abstracts only provide access to articles published since 1964). Second, electronic databases commonly do not include articles published in small or obscure journals or magazines that contain credible and relevant scientific and commercial data. Third, electronic databases do not include unpublished reports from government agencies, consulting firms, and non-governmental organizations that also contain credible and relevant scientific and commercial data. To overcome these limitations, we supplemented our electronic searches by searching the literature cited sections and bibliographies of references we retrieved to identify additional papers that had not been captured in our electronic searches. We acquired references that, based on a reading of their titles and abstracts, appeared to comply with our keywords. If a references’ title and abstract did not allow us to eliminate it as irrelevant to this inquiry, we acquired the reference.

To supplement our searches, we examined the literature that was cited in documents and any articles we collected through our electronic searches. If, based on a reading of the title or abstract of a reference, the reference appeared to comply with the keywords presented in the preceding paragraph, we acquired the reference. If a reference’s title did not allow us to eliminate it as irrelevant to this inquiry, we acquired it. We continued this process until we identified all of the relevant references cited by the introduction and discussion sections of the relevant papers, articles, books, modeling results, and, reports and all of the references cited in the materials and methods, and results sections of those documents. We did not conduct hand searches of published journals for this consultation.

These procedures allowed us to identify relevant data and other information that was available for our analyses. In many cases, the data available were limited to a small number of datasets that either did not overlap or did not conflict. In those cases, none of these sources were “better” than the alternatives and we used all of these data. However, when data and other information supported different conclusions, we used study design, sample size, level of scrutiny prior to and during publication (which included peer review) to determine which data sets were “best.” For example, we ranked carefully designed field experiments (for example, experiments that control variables, such as other sources of sound in an area, which might produce the same behavioral responses) higher than field experiments that were not designed to control such variables. We ranked carefully designed field experiments higher than computer simulations. Studies that were based on large sample sizes with small variances were generally ranked higher than studies with small sample sizes or large variances.

Finally, we also relied on data from the observer program for past interactions between the United States DSLL fishery and sea turtles, marine mammals, and elasmobranchs, recorded as bycatch between 2004 and 2022. These data represent credible and relevant commercial data. As a result, these represent the best scientific and commercial data available at the time of consultation.

Importantly, observer data is raw data. To be useful for consultation and decision-making in general, raw data requires processing. Once processed, the data becomes useful for decision-making. When a federal action agency has observational data collected from their specific action, as is the case for the United States DSLL fishery that raw data is usually going to be the best scientific and commercial data available for consultation on that action. Through collection, sorting, analyses and interpretation that raw best scientific and commercial data available is transformed into the best scientific and commercial data available for consultation. We would not be able to examine the past effects or predict future effects from the fishery if we did not transform the data.

2 STATUS OF LISTED RESOURCES

NMFS has determined that the action that it proposes to authorize, the Hawaii-based DSLL fishery, may affect the threatened and endangered species listed in Table 2. Some of these listed resources were not included in NMFS BE (NMFS 2018b); however, because they occur in the Action Area and may be affected by the proposed action they are included in this biological opinion. These listed resources are provided protections under the ESA.

Table 2. Listed resources in the Action Area that may be affected by the proposed action.

SPECIES COMMON NAME	**SCIENTIFIC NAME**	**LISTING STATUS**
	MARINE INVERTEBRATES	
Black Abalone	*Haliotis cracherodii*	Endangered
White Abalone	*Haliotis sorenseni*	Endangered
Corals	*Acropora globiceps*	Threatened
	Acropora jacquelineae	Threatened
	Acropora retusa	Threatened
	Acropora speciosa	Threatened
	Euphyllia paradivisa	Threatened

SPECIES COMMON NAME	**SCIENTIFIC NAME**	**LISTING STATUS**
	Isopora crateriformis	Threatened
Chambered Nautilus	*Nautilus pompilius*	Threatened
	SEA TURTLES	
Central North Pacific Green Sea Turtle	*Chelonia mydas*	Threatened
Central South Pacific Green Sea Turtle	*C. mydas*	Endangered
Central West Pacific Green Sea Turtle	*C. mydas*	Threatened
East Indian West Pacific Green Sea Turtle	*C. mydas*	Endangered
East Pacific Green Sea Turtle	*C. mydas*	Threatened
Southwest Pacific Green Sea Turtle	*C. mydas*	Threatened
Hawksbill Sea Turtle	*Eretmochelys imbricata*	Endangered
Leatherback Sea Turtle	*Dermochelys coriacea*	Endangered
Loggerhead Sea Turtle (North and South Pacific)	*Caretta caretta*	Endangered
Olive Ridley Sea Turtle (Mexican breeding population)*	*Lepidochelys olivacea*	Endangered
Olive Ridley Sea Turtle (all other populations)	*L. olivacea*	Threatened
	FISH	
California Central Valley Steelhead*	*Oncorhynchus mykiss*	Threatened
California Coast Steelhead*	*O. mykiss*	Endangered
Southern California Coast Steelhead*	*O. mykiss*	Threatened
Giant Manta Ray	*Manta birostris*	Threatened
Oceanic Whitetip Shark	*Carcharhinus longimanus*	Threatened
Scalloped Hammerhead - Eastern Pacific Shark	*Sphyrna lewini*	Endangered
Scalloped Hammerhead - Indo-West Pacific Shark	*S. lewini*	Threatened
Central Valley Spring-Run Chinook Salmon*	*Oncorhynchus tshawytscha*	Threatened

SPECIES COMMON NAME	SCIENTIFIC NAME	LISTING STATUS
Sacramento River Winter-Run Chinook Salmon*	*O. tshawytscha*	Endangered
Central California Coast Coho Salmon*	*Oncorhynchus kisutch*	Threatened
Southern North American Green Sturgeon*	*Acipenser medirostris*	Threatened
	MARINE MAMMALS	
Blue Whale	*Balaenoptera musculus*	Endangered
Fin Whale	*Balaenoptera physalus*	Endangered
Sei Whale	*Balaenoptera borealis*	Endangered
Mexico Humpback Whale	*Megaptera novaeangliae*	Threatened
Sperm Whale	*Physeter macrocephalus*	Endangered
False Killer Whale - Insular Main Hawaiian Islands	*Pseudorca crassidens*	Endangered
Southern Resident Killer Whale*	*Orcinus orca*	Endangered
Hawaiian Monk Seal	*Neomonachus schauinslandi*	Endangered
North Pacific Right Whale	*Eubalaena japonica*	Endangered
	CRITICAL HABITAT	
Black Abalone	-	Designated
Hawaiian Monk Seal	-	Designated
False Killer Whale - Main Hawaiian Islands Insular	-	Designated
Sacramento River Winter-Run Chinook Salmon	-	Designated
Central California Coast Coho Salmon	-	Designated
Steller Sea Lion	-	Designated
California Coast Steelhead	-	Designated
Southern North American Green Sturgeon	-	Designated

2.1 Listed Resources Not Likely Adversely Affected

As described in the Approach to the Assessment section of this biological opinion, NMFS uses two criteria to identify those endangered or threatened species or critical habitat that are not likely to be adversely affected by the Hawaii-based DSLL fishery. The first criterion is exposure or some reasonable expectation of a co-occurrence between one or more potential stressor associated with the Hawaii-based DSLL fishery and a particular listed species or designated critical habitat. If we conclude that a listed species or designated critical habitat extremely unlikely to be exposed to the Hawaii-based DSLL fishery, we must also conclude that the species and critical habitat is not likely to be adversely affected by those activities. The second criterion is the severity of a response given exposure, which considers susceptibility: for example, species that may be exposed to vessel noise from fishing vessels operating near them but are not likely to respond to that noise (at noise levels they are likely exposed to) and are also not likely to be adversely affected by the Hawaii-based DSLL fishery.

Based on the general exposure profiles that we developed during the course of this consultation, the following threatened and endangered species are not likely to be adversely affected by the DSLL fishery: black abalone, white abalone, corals, Southern Resident killer whale, Central California Coast coho salmon, Central Valley spring-run chinook salmon, Sacramento River winter-run Chinook salmon, Hawaiian monk seal, South Pacific loggerhead, hawksbill sea turtle, eastern Pacific scalloped hammerhead shark, California Coast steelhead, California Central Valley steelhead, southern California coast steelhead, southern North American green sturgeon, blue whale, fin whale, Mexico humpback whale, North Pacific right whale, and sei whale. Additionally, based on our evaluation of the proposed action, critical habitats designated for the following species are not likely to be adversely affected by the proposed action: black abalone, Sacramento River Winter-Run Chinook Salmon, Central California Coast Coho Salmon, Steller Sea Lion, California Coast Steelhead, Southern North American Green Sturgeon, leatherback sea turtle, Hawaiian monk seal, and MHI IFKW. We discuss the basis of these determinations in Appendix A.

2.2 Status of Listed Species That Are Likely to be Affected

The rest of this section of our biological opinion consists of narratives for each of the threatened and endangered species that occur in the Action Area and that may be adversely affected by the Hawaii-based DSLL fishery. These status assessments provide the point of reference for our analyses of whether or not the action's direct and indirect effects are likely to appreciably reduce a species' probability of surviving and recovering in the wild. To fulfill that purpose, each species' narrative presents a summary of (1) the species' distribution and population structure (which are relevant to the distribution criterion of the jeopardy standard); (2) the status and trend of the abundance of those different populations (which are relevant to the numbers criterion of the jeopardy standard); (3) information on the dynamics of those populations where it is available (which is a representation of the reproduction criterion of the jeopardy standard); and (4) natural and anthropogenic threats to the species, which helps explain our assessment of a species' likelihood of surviving and recovering in the wild. This information is integrated and synthesized in a summary of the status of the different species.

Following the narratives that summarize information on these three topics, each species' narrative provides information on the diving and social behavior of the affected species because that behavior helps assess a species' probability of being captured by longline fishing gear. Anyone interested in more detailed background information on the general biology and ecology of these species can be found in status reviews and recovery plans for the various species as well as the public scientific literature.

2.2.1 Giant Manta Ray

Distribution and Population Structure

The giant manta ray occurs across the globe in tropical and warm temperate bodies of water from 36°S to 40°N (Mourier 2012). The documented range for this species within the Northern hemisphere includes: Mutsu Bay, Aomori, Japan; the Sinai Peninsula and Arabian Sea, Egypt; the Azores Islands, Portugal; and as far north as southern California (west coast) and New Jersey (east coast), United States (Kashiwagi et al. 2010; Moore 2012; Convention on International Trade in Endangered Species [CITES] 2013). In the southern hemisphere, the giant manta has been documented as far south as Peru, Uruguay, South Africa, French Polynesia, New Zealand, and most recently, photographed in eastern Australia off Montague Island and Tasmania at 40° S (Mourier 2012; CITES 2013; Couturier et al. 2015). Couturier et al. (2015) documented the presence of the species for the first time in waters off eastern Australia and off the northeast coast of Tasmania. In addition, the giant manta ray has been observed in a predictable seasonal pattern in estuarine waters of Florida, Uruguay, and Brazil suggesting that they may use estuaries as nursery areas during summer months (Adams and Amesbury 1998; Milessi and Oddone 2003; Medeiros et al. 2015).

Previously considered to be monospecific, Marshall et al. (2009) presented new data to support the splitting of the *Manta* genus into two species: giant manta ray (*M. birostris*) and reef manta ray (*M. alfredi*). Prior to 2009, all *Manta* species were categorized as giant manta ray (*M. birostris*). The reef manta ray inhabits tropical coastal areas while the giant manta ray's habitat is more offshore and extends to sub-tropical regions; however, there is overlap in the habitats of the two species. Furthermore, while there are distinct morphological differences between the two species, they can be difficult to distinguish without adequate training and identification keys (Stevens et al. 2018). Therefore, correct identification to the species level is likely an issue in fisheries observer data, especially when the distinguishing morphological features cannot be seen by the observer.

Area of occupancy for giant manta rays was estimated from observations and expert opinion by Lawson et al. (2017; Figure 8). The environmental variables that drive giant manta ray habitat use in the ocean are largely unknown although temperature is a clear correlate (Jaine et al. 2014). Giant manta rays are found offshore in oceanic waters near productive coastlines, continental shelves, offshore pinnacles, seamounts, and oceanic islands. In a satellite tracking study off of Mexico, Graham et al. (2012) found that 95% of locations occurred in waters warmer than 21.6° C and that most locations were correlated with high surface chlorophyll concentrations.

Stewart et al. (2016b) also reported that giant manta ray off Mexico tend to occur near the upper limit of the pelagic thermocline where zooplankton aggregate. Burgess (2017) suggested that giant manta ray specifically feed on mesopelagic plankton, which would place them at depths as deep as 1,000 meters (also see Marshall et al. 2018). Giant manta ray are also observed at cleaning sites at offshore reefs where they are cleaned of parasites by smaller organisms.

The population structure of giant manta rays the number of populations and sub-populations that comprise the species, whether they are linked by immigration and emigration, and the strength of those links is largely unknown. At a minimum, the evidence suggests that giant manta rays in the Atlantic and giant manta rays in the Indo-Pacific represent separate populations because this species does not appear to migrate to the Pacific through Drake Passage (or vice versa) and they do not appear to migrate around the Cape of Good Hope to the Indian Ocean (Lawson et al. 2017; Marshall et al. 2018; Figure 8).

Several authors have reported that giant manta ray likely occur in small regional subpopulations (Lewis et al. 2015; Stewart et al. 2016a; Marshall et al. 2018; Beale et al. 2019) and may have distinct home ranges (Stewart et al. 2016a). The degree to which subpopulations are connected by migration is unclear but is assumed to be low (Stewart et al. 2016a; Marshall et al. 2018) so regional or local populations are not likely to be connected through immigration and emigration (Marshall et al. 2018), making them effectively demographically independent. While NMFS' concluded that the species is likely to become endangered within the foreseeable future throughout a significant portion of its range (the Indo Pacific and eastern Pacific), NMFS did not find the species met the criteria to list as a DPS (83 FR 2916 and 82 FR 3694). This decision is unique to the listing process, and does not mean that NMFS should not or would not consider the potential role that populations play in evaluating whether a proposed action is likely to result in appreciable reduction in numbers, distribution or reproduction, or whether such reductions may affect the viability of the putative populations that comprise the listed species.

The preponderance of current evidence, combined with expert opinion suggest the species likely has a complex population structure, and while it may occasionally be observed making long distance movements, it likely occurs in small spatially separated subpopulations, though to be viable the abundance of each subpopulation likely needs to be at least 1,000 individuals (Frankham et al. 2014). This structure is further supported by studies described by Beale et al. (2019) that have documented fisheries-induced declines in several isolated subpopulations (Lewis et al. 2015; Stewart et al. 2016a; Moazzam 2018).

Several studies have tracked individual giant manta rays and provide information on the spatial extent of giant manta ray subpopulations. Stewart et al. (2016a) studied four subpopulations of giant manta ray using genetics, stable isotopes, and satellite tags. They found that these subpopulations appeared to be discrete with no evidence of movement between them. The home ranges for three of these subpopulations (all of which are outside of the Action Area), defined as the areas where tagged animals were expected to spend 95% of their time encompassed areas of 79,293 km^2 (Raja Ampat, Indonesia), 70,926 km^2 (Revillagigedo Islands, Mexico), and 66,680 km^2 (Bahia de Banderas, Mexico). They suggest that their findings indicate that giant manta rays form discrete subpopulations that exhibit a high degree of residency. Stewart et al. (2016a) state that this does not preclude occasional long-distance migrations, but that these migrations are likely rare and do not generate substantial gene flow or immigration of individuals into these subpopulations.

The Status Review (Miller and Klimovich 2017), notes only four instances of individual tagged giant manta rays making long-distance migrations. Of those, one animal was noted to travel a maximum distance of 1,151 km but that was a cumulative distance made up of shorter movements within a core area (Graham et al. 2012). No giant manta ray in that study moved further than 116 km from its tagging location and the results of Graham et al. (2012) support site fidelity leading to subpopulation structure. The remaining references to long distance migrations include Mozambique to South Africa (1,100 km), Ecuador to Peru (190 km), and the Yucatan into the Gulf of Mexico (448 km). The last two distances are well within core areas of subpopulation habitat use as specified in Stewart et al. (2016a) and may only represent movements between coastal aggregation sites and offshore habitats as discussed in Stewart et al.

(2016a). An additional instance of a long-distance migration is from Hearn et al. (2014) who tracked nine giant manta rays at Isla de la Plata, Ecuador. Eight of the nine tagged giant manta rays remained in an area of 162,500 km^2, while the ninth traveled a straight-line distance of 1,500 km to the Galapagos Islands, however, Stewart and Hearn later believed it may have been from a floating tag and not the result of a long distance migration (J. Stewart pers. comm. to J. Rudolph, October 7, 2020).

In contrast with these few individuals making long-distance movements, most tracked individuals (Hearn et al. 2014 [8 out of 9 individuals]) or all tracked individuals (Graham et al. 2012 [6 individuals]; Stewart et al. 2016a [18 individuals]) from other studies remained within defined core areas, supporting subpopulation structure. Marshall et al. (2018) summarizes that current satellite tracking studies and international photo-identification matching projects suggest a low degree of interchange between subpopulations.

To date there have been limited genetics studies on giant manta ray; however, Stewart et al. (2016a) found genetic discreteness between giant manta ray populations in Mexico suggesting isolated subpopulations with distinct home ranges within 500 km of each other. In addition to genetics, differentiation was discovered through isotope analysis between those two Mexican populations (nearshore and offshore) and between two others (Indonesia and Sri Lanka). Using satellite tagging, stable isotopes and genetics, Stewart et al. (2016a) concluded that, in combination, the data strongly suggest that giant manta rays in these regions are well-structured subpopulations that exhibit a high degree of residency. In the Gulf of Mexico, Hinojosa-Alvarez et al. (2016) propose a genetically distinct diverged group that may be a separate species and tentatively termed *M. cf. birostris*.

A vulnerability analysis conducted by Dulvy et al. (2014) indicates that mobulid populations can only tolerate very low levels of fishing mortality and have a limited capacity to recover once their numbers have been depleted (Couturier et al. 2012; Lewis et al. 2015). Furthermore, Lewis et al. (2015) suggests local populations in multiple areas in Indonesia have been extirpated due to fishing pressure noting that *M. birostri*s was the most common species previously caught in these areas. Additionally, White et al. (2015) documented an 89% decline in the observed *M. birostris* population in Cocos Island National Park over a 20-year period and is believed to be from overfishing outside of the park. Note that these declines are from directed fishing and not bycatch.

A population structure described by small, isolated subpopulations does not conflict with seasonal sightings of giant manta ray as described for a number of the subpopulations studies with photo-identification or acoustic arrays (in contrast with those using satellite tagging; Dewar et al. 2008; Marshall et al. 2009; Rohner et al. 2013). Stewart et al. (2016a) suggest that habitats used by giant manta rays include both nearshore and offshore locations, and that the core spatial distribution of giant manta ray subpopulations encompass both types of habitats, leading to seasonal observations of giant manta rays in the nearshore habitats in many areas. Water temperature and productivity may dictate giant manta ray movements (Freedman and Roy 2012; Beale et al. 2019). In a subpopulation off the coast of North Carolina (United States), Freedman and Roy (2012) found that in the cooler winter months, giant manta ray distribution was extremely limited with a tight clustering in an area associated with the Gulf Stream and warmer waters, while in summer giant manta ray were distributed across a larger area, and individuals were more spread out, yet still in a discrete area.

Not all giant manta ray subpopulations are defined by seasonal sightings. Studied subpopulations that have more regular sightings include the Similan Islands (Thailand); Raja Ampat (Indonesia); northeast North Island (New Zealand); Kona, Hawaii (USA); Laje de Santos Marine Park (Brazil); Isla de la Plata (Ecuador); Ogasawara Islands (Japan); Isla Margarita and Puerto la Cruz (Venezuela); Isla Holbox, Revillagigedo Islands, and Bahia de Banderas, Mexico (Notarbartolo-di-Sciara and Hillyer 1989; Homma et al. 1999; Duffy and Abbott 2003; Luiz et al. 2009; Clark 2010; Kashiwagi et al. 2010; Marshall et al. 2011; Stewart et al. 2016a).

Given the current understanding of giant manta ray population structure, for the remainder of this biological opinion, we will use the terms 'giant manta ray' or 'species' to refer to the giant manta ray as they were listed, the term 'population' to refer to the Indo-Pacific population as a whole, and 'subpopulation' to refer to independent subunits considered in this biological opinion. We note that for some of the study areas identified in Table 3 where only small numbers of individuals have been identified, these may not represent regionally defined subpopulations and we consider them aggregations until further data can be collected.

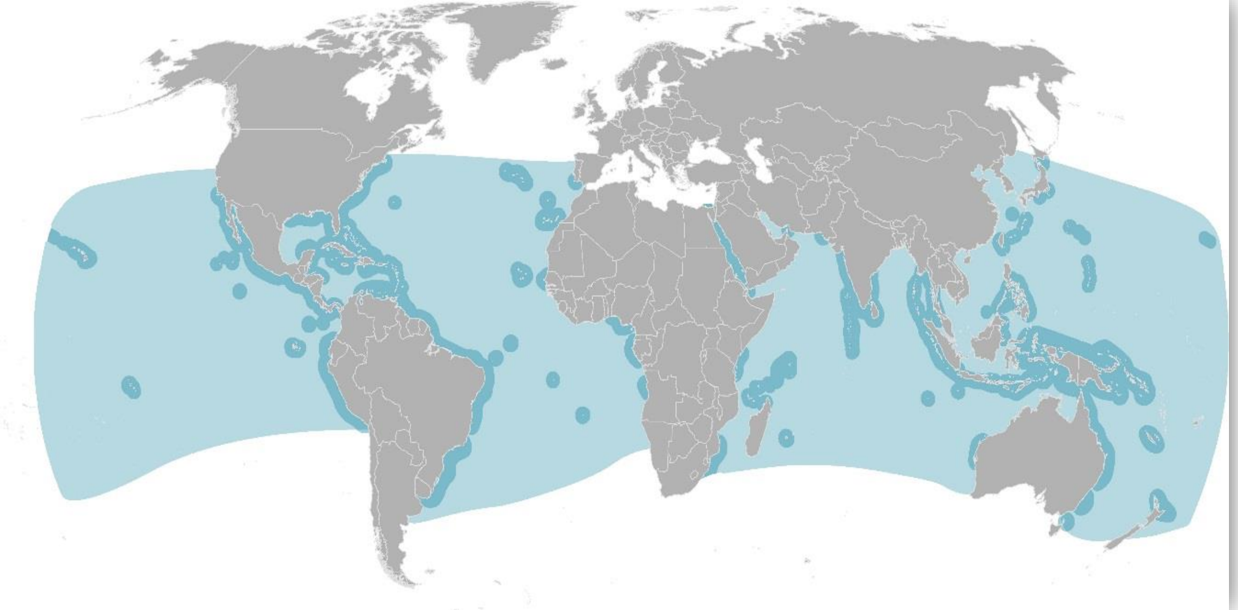

Figure 8. Distribution map for the giant manta ray. Extent of occurrence is depicted by light blue and the area of occupancy is noted in darker blue. (Figure 3 from Lawson et al. 2017).

Status and Trends

NMFS listed giant manta rays globally as threatened in 2018. The International Union for Conservation of Nature (IUCN) lists them as vulnerable (the category that immediately precedes endangered in the IUCN classification system), with a decreasing population trend. The number of regional subpopulations is unknown, and abundance estimates are lacking for most. Abundances for regional subpopulations with estimates range from 600 to 25,250 (CITES 2013; Marshall et al. 2018; Beale et al. 2019; Table 3). The Convention on International Trade in Endangered Species (CITES 2013) highlights three giant manta ray subpopulations that have been studied and population estimates provided, and counts for more than ten aggregations

(Table 3). CITES (2013) also discusses an additional approximately 25 aggregations where species-level information (i.e., *M. birostris* vs *M. alfredi*) does not exist and, while actual abundance estimates are not available, it is assumed they consist of very small number of individuals. This information was compiled from O'Malley et al. (2013), Heinrichs et al. (2011), Lewis et al. (2015), and Fernando and Stevens (2011). The most comprehensive of these is O'Malley et al. (2013) that presents an overview of the economic value of manta ray watching tourism. They highlight 23 sites globally, although none are within the Action Area. Overall, giant manta ray subpopulations appear to be regionally distinct (Lewis et al. 2015; Stewart et al. 2016a; Moazzam 2018; Beale et al. 2019) and may have distinct home ranges (Stewart et al. 2016a). We note that the reef manta ray is not listed under the ESA.

Most documented giant manta ray subpopulations appear to be composed of relatively small population sizes. Photo-identification studies for giant manta ray subpopulations include southern Mozambique (*n*= 180-254; Marshall et al. 2009); southern Brazil (*n*= 60; Luiz et al. 2009); Revillagigedo Islands, Mexico (*n*= 916; J. Stewart pers. comm. to A. Garrett citing pers. comm to R. Rubin and K. Kumli [2021])); the Ogasawara Islands, Japan (*n*= 42; Kashiwagi et al. 2010); the Maldives (*n*= 716; J. Stewart pers. comm. to A. Garrett citing S. Hilbourne pers. comm. 2021)); Isla Holbox, Mexico (*n*= 200; S. Hinojosa-Alvarez unpubl. data 2010 cited in Marshall et al. 2018); with many of these studies having been conducted for the last 10–20 years (Table 3).

A study of Japan-wide photographic records confirmed that the known main aggregation in Ogasawara Islands (42 known individuals during 1995–1998 study) represents a part of a fairly isolated population (Kashiwagi et al. 2010). A mark-recapture population study in southern Mozambique over five years from 2003 to 2008 estimated the local population during that time to be 600 individuals (Marshall et al. 2009). Flight surveys and re-sightings data of individuals at Isla Holbox, Mexico have estimated that roughly 100 manta rays use this area during every season (S. Hinojosa-Alvarez unpubl. data 2010 cited in Marshall et al. 2018). However, 'recorded individuals' as identified in Table 3 may not be indicative of population size.

The number of individually identified giant manta ray for each studied aggregation ranges from less than 50 in regions with low survey effort or infrequent sightings to more than 1,000 in some regions with targeted, long-term studies. However, ongoing research including mark-recapture analyses suggests that typical subpopulation abundances are more likely in the low thousands (e.g., Beale et al. 2019) and in rare cases may exceed 10,000 in areas with extremely high productivity (pers. comm. Joshua Stewart, Manta Trust to A. Garrett 2021).

Of the 12 studied subpopulations identified in Table 3, statistical analyses of sightings/photo-identification data to estimate total population size has only been conducted for three of them. For Raja Ampat, CITES (2013) indicated that there were 72 identified individuals. After additional research and an analysis of resighting data, Beale et al. (2019) estimated the total population size to be approximately 1,875 individuals. Isla de la Plata, Ecuador had approximately 650 identified individuals reported in CITES (2013), in this case, Burgess (2017) conducted further analyses and estimates the total population size to be 2,464 individuals. Similarly, for the Republic of Maldives, as of 2013, 63 individuals had been identified (CITES 2013), Nicholson-Jack (2020) reported 378, and further study indicates a more than 10-fold increase over the initial number of identified individuals (n = 716; J. Stewart pers. comm. to A. Garrett citing S. Hilbourne pers. comm. 2021; Table 3). Thus, while some subpopulations may have been reduced to very small population sizes due to fisheries (direct harvest or bycatch), in general, stable giant manta ray subpopulations are likely to be larger, potentially greater than 1,000 individuals, which would be in keeping with the literature that suggests subpopulations are isolated with limited movement. The current understanding of effective population sizes[4] necessary for the genetic diversity needed to maintain evolutionary fitness in isolated populations is greater than 1,000 (Frankham et al. 2014).

More importantly, the size of some of these subpopulations has declined significantly in regions subject to fishing (Marshall et al. 2018). Fisheries catch and bycatch have caused giant manta rays to decline by at least 30% globally and by up to 80% in significant portions of its range (i.e., Indonesia, Philippines, Sri Lanka, Thailand, Madagascar; Marshall et al. 2018). Lewis et al. 2015 collected data on daily landings of *Manta* and *Mobula* species from 2002 to 2014 for eight locations in Indonesia. For Manta species, *M. birostris* was the primary target of these fisheries. Total annual landings were estimated by multiplying the number of recorded or observed daily landings by the number of fishing days per year. For the three locations with the most complete data, landings of *Manta* species declined by 71% to 95%. Reports from fishermen suggest that these data are representative of declines in abundance rather than shifts in effort.

Within the Action Area, Tremblay-Boyer and Brouwer (2016) present catch per unit effort (CPUE) data for giant manta ray observed captures in the WCPO longline and purse seine fisheries. Giant manta ray were not reliably identified to species by observers in the WCPO purse seine fishery until about 2011 (NMFS 2021a). In their analysis, Tremblay-Boyer and Brouwer (2016) found increasing trends in CPUE from 2005 to 2016 for giant manta rays but they caution that these trends represent increases in compliance with reporting the species and does not represent an index of abundance. CPUE trends in the longline fisheries indicate that giant manta rays are observed less frequently in recent years compared to 2000-2005, suggesting a decline in abundance (Tremblay-Boyer and Brouwer 2016).

Table 3. Numbers of recorded individuals and subpopulation estimates of giant manta ray at identified locations adapted from CITES (2013) and updated with supplementary references as specified.

Location	Recorded Individuals	Subpopulation Estimate	Reference
Mozambique	180 - 254	600	Marshall et al. (2009) and pers. comm. cited in CITES (2013); MantaMatcher (2016)
Egypt	60	-	Marine Megafauna (2011) as cited in CITES (2013)

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[4] The effective population size is considered the number of individuals in a population who contribute offspring to the next generation (i.e. breeding adults; Ridley 2003).

Republic of Maldives	716	-	J. Stewart pers. comm. to A. A. Garrett citing S. Hilbourne pers. comm. (2021)
Republic of Maldives	378	-	Nicholson-Jack (2020)
Kona, Hawaii (United States)	29	-	Clark (2010)
Thailand	365	-	J. Stewart pers. comm. to A. Garrett citing Manta Trust data (2021)
Raja Ampat, Indonesia	588	1,875	Beale et al. (2019)
Isla de la Plata, Ecuador	~650	1,500	M. Harding, pers. comm. cited in CITES (2013); Sanchez (2016)
Isla de la Plata, Ecuador	2,464	25,250	MantaMatcher (2016); Burgess (2017); Marshall and Holmberg 2011 as cited in Burgess (2017); Subpopulation estimate from J. Stewart pers. comm. to A. Garrett (2021)
Brazil	60	-	Laje Viva Institute unpubl. cited in CITES (2013); Luiz et al. (2009)
Mexico (Revillagigedos Is.)	916	-	J. Stewart pers. comm. to A. Garrett citing pers. comm to R. Rubin and K. Kumli (2021)
Mexico (Isla Holbox)	> 200	-	R. Graham, pers. comm. cited in CITES (2013)
Jupiter, Florida (United States)	59	-	Pate and Marshall (2020)

Flower Garden Banks (United States EEZ)	>70	-	Graham and Witt (2008) cited in CITES (2013)
Flower Garden Banks (United States EEZ)	95 (52 proposed *M. cf. birostris*)	-	Stewart et al. (2018)
Japan (Ogasawara Islands)	42	-	Kashiwagi et al. (2010)
Azores, Portugal	31	-	J. Stewart pers. comm. to A. Garrett citing A. Sobral pers. comm. (2021).
Myanmar	201	-	J. Stewart pers. comm. to A. Garrett citing Manta Trust data (2021)
Costa Rica	52	-	J. Stewart pers. comm. to A. Garrett citing Manta Trust data (2021)

Population Dynamics

Giant manta rays are a long-lived, late maturing species with productivity that is among the lowest of all elasmobranchs. Rambahiniarison et al. (2018) estimated that giant manta ray off the Philippine Islands matured at about 9 years and had their first pregnancy at about 13 years of age. Overall, age at maturity estimates range from three to more than 15 years. Giant manta rays typically give birth to only one pup every two to three years, but this can range from annual to 5 years (Notarbartolo-Di-Sciara 1988; Marshall and Bennett 2010; Dulvy et al. 2014; Rambahiniarison et al. 2018). Rambahiniarison et al. (2018) reported that the proportion of pregnant females in subpopulations of giant manta ray in the Philippine Islands averaged about 9 out of every 100 females (9%), but they suggested this might depend on the length of the inter-pregnancy period which could depend on the availability of resources. Additionally, sex ratios may differ between populations. Beale et al. (2019) noted a statistically significant female-biased sex ratio of 2.62(f):1 in Raja Ampat. However, Pate and Marshall (2020) did not find a statistical difference in Florida with a sex ratio of 1:1 and Stewart et al. (2018) noted a ratio of 1.3(f):1 in the Flower Garden Banks of the Gulf of Mexico. Differences between locations may be due to unique threats to each population.

Gestation is thought to last around a year. Although manta rays have been reported to live at least 40 years (Dulvy et al. 2014), not much is known about their growth, development, and population dynamics, although generation time is estimated at 25 years. Nevertheless, the combination of long-lives, late-maturation, and low productivity would make this species particularly vulnerable to harvests that target adults (Dulvy et al. 2014; Croll et al. 2016; Miller and Klimovich 2017), which would limit their ability to recover from over-exploitation (Crouse 1999). To illustrate this point, Rambahiniarison et al. (2018) estimated that giant manta ray subpopulations would require about 36.5 to 86.6 years to double in size (the former based on estimated age to maturity; the latter based on estimated age of first pregnancy). A population that requires about 4 to almost 9 decades to double in size has limited ability to recover from exploitation and disturbance, particularly when the exploitation is constant.

In order to determine how changes in survival may affect populations, Smallegange et al. (2016) modeled the demographics of reef manta rays (*M. alfredi*), which have similar life history characteristics to giant manta rays, therefore we chose this species as a proxy and assume their results are relevant to giant manta rays. In their own observations of the population off the southern coast of Mozambique, the authors estimated an adult survival rate of 0.67 (± 0.16 SE). Results from the population modeling showed that, at this adult survival rate and yearling survival rates greater than 0.75, population growth rate was most sensitive to changes in juvenile survival, while if yearling survival rates were less than 0.75, population growth rates were most sensitive to adult survival rates. They contrasted these results to a population model based on an estimated survival rate of 0.95 for a stable reef manta ray population in Japan (Kashiwagi 2014).

Based on the elasticity analysis, population growth rate was most sensitive to changes in the survival rate of adults regardless of yearling and juvenile survival rates (Smallegange et al. 2016). In other words, in order to prevent populations from declining further, Smallegange et al. (2016) found that increases in adult survival rates would have the greatest impact, such as through protection of adult aggregation sites or a reduction in fishing of adult manta rays (Smallegange et al. 2016). However, their results also show that low yearling and juvenile survival can result in declining populations even if adult survival remains high, indicating that increased mortality of those life stages are also important to population dynamics.

Diving and Social Behavior

Although giant manta rays are considered more oceanic and solitary than the reef manta, they have been observed congregating at cleaning sites at offshore reefs and feeding in shallow waters during the day at depths <10 m (O'Shea et al. 2010; Marshall et al. 2011; Rohner et al. 2013). Unlike the reef manta ray, the giant manta ray does not appear in large schools (<30 individuals; Marshall et al. 2018) and despite having a larger distribution when compared to the reef manta, they are encountered with far less frequency.

Giant manta rays appear to exhibit a high degree of plasticity in terms of their use of depths within their habitat. Tagging studies have shown that the species conducts night descents to 200-450 m depths (Rubin et al. 2008 as cited in Miller and Klimovich 2017; Stewart et al. 2016b) but is capable of diving to depths exceeding 1,000 m (A. Marshall et al. unpubl. data 2011 cited in Marshall et al. 2011).

Threats to the Species

Giant manta rays are reportedly targeted in fisheries in Indonesia, Philippines, India, Thailand, Mozambique, Tonga, Micronesia, Peru, Ghana, and previously in Mexico and possibly the Republic of Maldives. Indonesia is reported to be one of the top countries that catch mobulid rays (Heinrichs et al. 2011). Manta and devil ray fisheries span the majority of the Indonesian archipelago, with most landing sites along the Indian Ocean coast of East and West Nusa Tenggara and Java (Lewis et al. 2015). Although fishing for manta rays was banned within the

Indonesian exclusive economic zone (EEZ) in February 2014, in May 2014, manta rays were still being caught and processed at Lamakera, with the giant manta the most commonly targeted species through 2015 (Marshall and Conradie 2014; Booth et al. 2021). A subsequent integrated intervention program that involved community outreach, incentives and targeted enforcement reduced manta ray landings beginning in 2016, with devil rays becoming the most commonly landed species (Booth et al. 2021).

Giant manta rays are also frequently caught as bycatch in a number of commercial and artisanal fisheries worldwide, particularly purse-seine and gillnet fisheries and to a lesser extent commercial longline and trawl fisheries off Europe, western Africa, the Atlantic coast of the United States, Australia, and the Pacific and Indian Oceans.

In regions outside of the Action Area considered in this biological opinion (captures in fisheries that overlap the Action Area are considered in the Environmental Baseline section), giant manta rays are caught in the United States WCPO purse seine fishery and the ASLL fishery. The United States WCPO purse seine fishery incidentally captured 1,523 giant manta rays from 2010-2018 and an estimated 3,676 (95% CI: [3,119, 4,467]) interactions accounting for unidentified *Manta* species and unavailable observer data (NMFS 2021a). However, it is also considered highly likely that a large portion (~75%) of those individuals identified as giant manta ray were misidentified by observers primarily based on the estimated weights of captured individuals (see discussion in NMFS 2021a). In contrast the ASLL fishery captured 12 giant manta rays from 2010-2017 (based on 19 - 25% observer coverage), resulting in an estimated 122 interactions accounting for unobserved sets and individuals not identified to species (McCracken 2019b; NMFS 2022c).

Conservation

Domestic fishery regulations prohibit the retention of manta rays by persons under United States jurisdiction. Additionally, as noted in the final status review report (Miller and Klimovich 2017), established Marine Protected Areas (MPAs) that limit or prohibit fishing also exist that cover areas with observed giant manta ray presence, including the waters off Guam (Tumon Bay Marine Preserve), within the Gulf of Mexico (Flower Garden Banks National Marine Sanctuary), and in the Central Pacific Ocean (Pacific Remote Islands Marine National Monument).

Internationally, the giant manta ray is protected in the Maldives, Philippines, Mexico, Brazil, Ecuador, Yap, Indonesia, Western Australia, and New Zealand (Miller and Klimovich 2017). These protections range from restrictions on knowingly capturing or killing rays, to bans on exportation of ray species and their body parts from established Marine Protection Areas of known giant manta ray aggregations. However, many of these restrictions are difficult and rarely enforced; in Indonesia, restrictions have driven the price of manta ray products up (Marshall and Conradie 2014), which has likely increased demand and had the opposite effect intended.

Manta rays were included on Appendix II of CITES at the 16 Conference of the CITES Parties in March 2013. Export of manta rays and manta ray products, such as gill plates, require CITES permits that ensure the products were legally acquired and that the Scientific Authority of the State of export has advised that such export will not be detrimental to the survival of that species (after taking into account factors such as its population status and trends, distribution, harvest, and other biological and ecological elements). Although this CITES protection was not considered to be an action that decreased the current listing status of the threatened giant manta ray, it may help address the threat of foreign overutilization for the gill plate trade by ensuring that international trade of this threatened species is sustainable (Miller and Klimovich 2017).

In November 2014, the Convention on the Conservation of Migratory Species of Wild Animals listed the giant manta ray on Appendix I and II of the Convention (CMS 2014). Under this designation, Conservation of Migratory Species Parties strive to protect these animals, conserve and restore habitat, mitigate obstacles to migration and engage in international and regional agreements.

There are many conservation efforts presently ongoing to collect research on manta ray life history, ecology, and biology, and to raise awareness of threats to manta rays. Some of these efforts are spearheaded by non-profit organizations specifically dedicated to manta ray conservation, such as the Manta Trust (Stevens et al. 2018), the Marine Megafauna Foundation, the Manta Pacific Research Foundation and MantaWatch. Others are driven by the countries whose economies largely depend on manta ray tourism such as in Raja Ampat Indonesia (Erdmann 2014; Beale et al. 2019).

In addition, guidelines for best practices for the safe release of manta rays caught in purse seine and longline fisheries have been developed (Hutchinson et al. 2017) and, as discussed in the Description of the Proposed Action section, went into effect as a West Central Pacific Fisheries Convention Implementation Act in January 2021. CMM 2019-05 (effective January 1, 2021) prohibits all fishing vessels operating in the high seas and/or exclusive economic zones of the Convention area and flagged to Members, Cooperating Non-Members and Participating Territories of the WCPFC from targeted fishing or intentional setting on mobulid rays; from retaining on board, transshipping, or landing any part or whole carcass of mobulid rays; fishing vessels must promptly release animals alive and unharmed that will result in the least possible harm to the individuals captured. The U.S. issued a rule to put the handling practices in CMM 2019-05 into regulation for U.S. fisheries (88 FR 30671).

Summary of the Status

In this section of this biological opinion, we explained that the giant manta ray is highly fragmented and, while they occur across the Pacific, their distribution is sporadic, which contributes to the lack of information on this species. It is one of the least understood of the marine mega vertebrates. Many of the studied giant manta ray populations' have declined significantly in areas subject to fishing (Marshall et al. 2018). Fisheries catch and bycatch have caused giant manta rays to decline by at least 30% globally and by up to 80% in significant portions of its range (i.e., Indonesia, Philippines, Sri Lanka, Thailand, Madagascar; Marshall et al. 2018). In Indonesia, manta ray landings are estimated to have declined by 71% to 95%, with potential extirpations noted in certain areas (Lewis et al. 2015).

Targeted capture and bycatch in fisheries is arguably the most significant threat to the giant manta ray (Croll et al. 2016). Due to their particular life-history characteristics (e.g., slow growth, late maturity, and low fecundity), elasmobranchs, and specifically, the giant manta ray, are vulnerable to high and sustained levels of fishing exploitation (Hoenig and Gruber 1990; Stevens et al. 2000; Couturier et al. 2012; Dulvy et al. 2014). Despite conservation efforts of protections and conservation measures, the overall trend of the giant manta ray continues to decline.

2.2.2 Indo-West Pacific Scalloped Hammerhead Shark

Distribution and Population Structure

In 2014, the scalloped hammerhead shark was determined to consist of six DPS' and of those, four were listed as either threatened or endangered including the Indo-West Pacific scalloped hammerhead shark (79 FR 38213; Figure 9). The majority of the Action Area overlaps with the range of the Central Pacific scalloped hammerhead shark which is not listed under the ESA. While most observed scalloped hammerhead shark captures have occurred within the range of the Central Pacific scalloped hammerhead shark, there have been a smaller number of captures overlapping with the range of the Indo-West Pacific scalloped hammerhead shark. Our assessment is limited to analyzing the effect of the Hawaii DSLL fishery on threatened Indo-West Pacific scalloped hammerhead sharks.

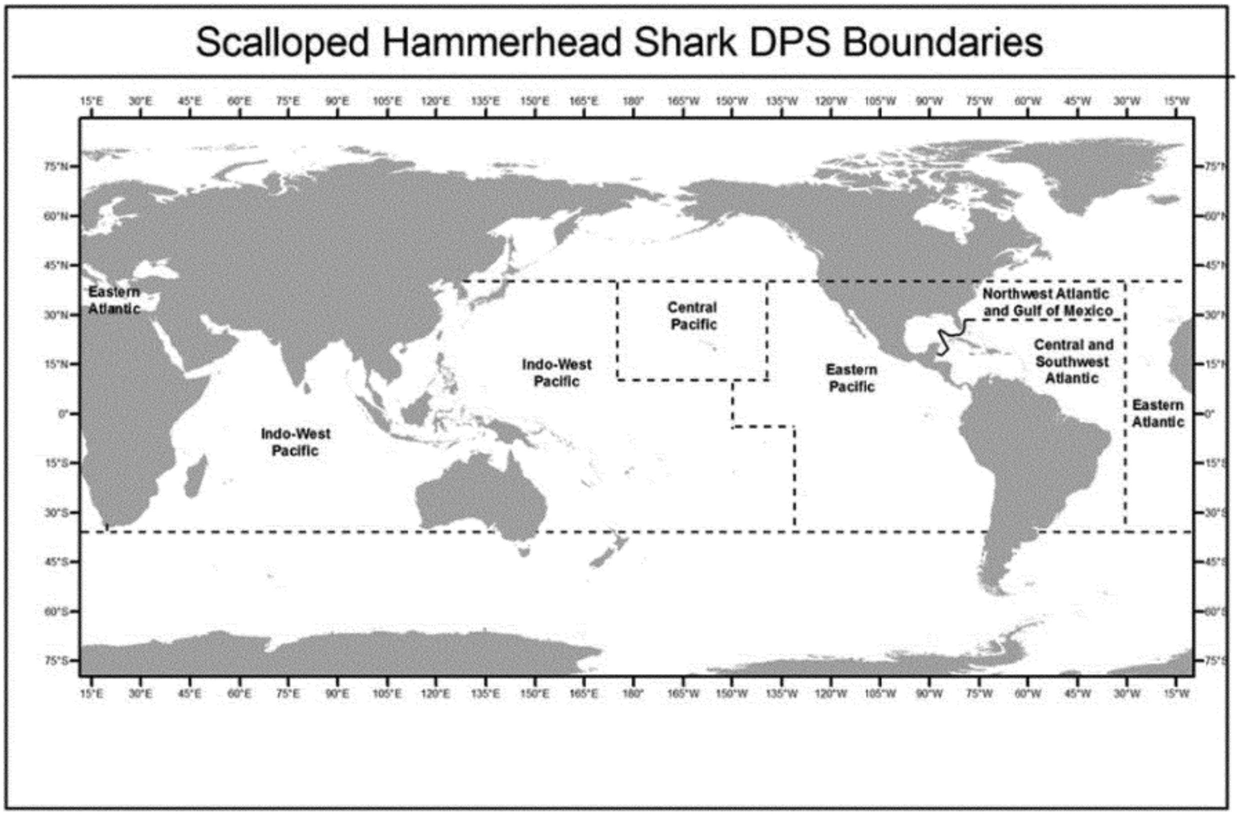

Figure 9. DPS boundaries of the scalloped hammerhead shark (79 FR 38213).

Scalloped hammerhead sharks (*S. lewini*) can be found in coastal warm temperate and tropical seas worldwide. Indo-west Pacific scalloped hammerhead sharks can be found throughout the entire Indian Ocean and in the western Pacific from Japan and China to New Caledonia, including throughout the Philippines, Indonesia, and off Australia. The scalloped hammerhead shark occurs over continental and insular shelves, as well as adjacent deep waters, but is seldom found in waters cooler than 22°C (Compagno 1984).

These sharks have been observed making migrations along continental margins as well as between oceanic islands in tropical waters (Kohler and Turner 2001; Duncan and Holland 2006;

Bessudo et al. 2011; Diemer et al. 2011). Tagging studies reveal the tendency for scalloped hammerhead sharks to aggregate around and travel to and from core areas or "hot spots" within locations (Holland et al. 1993; Duncan and Holland 2006; Hearn et al. 2010; Bessudo et al. 2011), however they are also capable of traveling long distances (1671 km, Kohler and Turner 2001; 1941 km, Bessudo et al. 2011; 629 km, Diemer et al. 2011).

Status and Trends

Indo-west Pacific scalloped hammerhead sharks are listed as threatened because of overharvesting. Although range-wide trends in the abundance of this species are unknown, CPUE data suggest that local abundance throughout the range of the species have declined significantly (Miller et al. 2014). For example, the hammerheads in Australia's northwest marine region have been estimated to have declined between 58-76% between 1996 and 2005 (Miller et al. 2014). Similarly, catch rates of *S. lewini* in beach mesh programs in South Africa have declined by 99%, 86%, and 64% from 1952-1972, 1961-1972, and 1978-2003, respectively (Dudley and Simpfendorfer 2006; Ferretti et al. 2010). Estimates of the decline in Australian hammerhead abundance range from 58-85% (Heupel and McAuley 2007; CITES 2010). Data from protective shark meshing programs off beaches in New South Wales (NSW) and Queensland also suggest significant declines in hammerhead abundance off the east coast of Australia. From 1973 to 2008, the number of hammerheads caught per year in NSW beach nets decreased by more than 90% from over 300 individuals to fewer than 30 (Reid and Krogh 1992; Williamson 2011; Miller et al. 2014). Similarly, data from the Queensland shark control program indicate declines of around 79% in hammerhead shark abundance between the years of 1986 and 2010, with *S. lewini* abundance fluctuating over the years but showing a recent decline of 63% between 2005 and 2010 (QLD DEEDI 2011 as cited in Miller et al. 2014).

Estimates of current effective population size are available for the scalloped hammerhead shark, but are considered qualitative indicators rather than precise estimates given their reliance on mutation rates and generation times (Duncan et al. 2006). Using two generation times (5.7 and 16.7 years), Duncan et al. (2006) calculated the effective female population (N_f) size of *S. lewini* for the major ocean basins. Based on a 1:1 sex-ratio (Clarke 1971; Chen et al. 1988; Stevens and Lyle 1989; Ulrich et al. 2007; White et al. 2008; Noriega et al. 2011), these calculations have been converted into total (both females and males) effective population size (N_e) by using the formula $N_e = 2(N_f)$. Results of N_e greatly varied within and between ocean basins, with the global N_e estimated at 280,000 using a generation time of 5.7 years, and 94,000 using a generation time of 16.7 years (Miller et al. 2014). There are no estimates of abundance for the Indo-West Pacific scalloped hammerhead sharks but we can assume it is less than the global abundance of 280,000.

Pacoureau et al. (2021) indicates a 67% global decline from 1970 to 2018 equating to a 2.31% decline per year. This decline is based on trends from the North Atlantic, South Pacific and Indian Oceans. South Pacific scalloped hammerhead sharks are estimated to have declined by 94% from 1970 to 1998, equating to a decline of 9.8% per year. But from 1998 to 2006, the population increased at a rate of 2.8%. Indian Ocean scalloped hammerhead sharks declined by 66% from 1978 to 2000, equation to a decline of 4.9% per year. But from 2000 to 2003, the population increased at a rate of 2.8% per year. Given that these data represent limited geographic locations and there are no new data since 2006, we cannot be certain that the slight positive growth indicated at the end of the time series are indicative of overall positive trends for the Indo-West Pacific scalloped hammerhead shark. To account for this uncertainty, we use the mean of the population growth rates (-9.8%, -4.8%, 2.8%, 2.8%) or -2.28% per year as the current trend in our analysis.

Population Dynamics

Like the other elasmobranchs included in this biological opinion, scalloped hammerhead shark are long lived, late maturing, and with low productivity (Branstetter 1990). Although their age at maturity varies geographically, scalloped hammerhead sharks are generally considered mature about 200-250 cm total length (females) while males reach maturity at smaller sizes (range 128 – 200 cm). These lengths correspond to ages from 3.8 to 15.2 years. They are estimated to live for at least 20 to 30 years, have gestation periods of 9 to 12 months (Branstetter 1987; Stevens and Lyle 1989), give birth to live young, and females may rest for about 12 months between births (Liu and Chen 1999).

Diving and Social Behavior

Both juvenile and adult scalloped hammerhead sharks occur as solitary individuals, pairs, or in schools. The schooling behavior has been documented during summer migrations off the coast of South Africa as well as in areas of permanent residents, like those in the East China Sea (Compagno 1984). Adult aggregations are most common offshore over seamounts and near islands, especially near the Galapagos, Malpelo, Cocos and Revillagigedo Islands, and within the Gulf of California (Compagno 1984; CITES 2010; Hearn et al. 2010; Bessudo et al. 2011). Neonate and juvenile aggregations are more common in nearshore nursery habitats, such as Kaneohe Bay in Oahu, Hawaii, and Guam's inner Apra Harbor (Duncan and Holland 2006; Bejarano-Alvarez et al. 2011). It has been suggested that juveniles inhabit these nursery areas for up to or more than a year, as they provide valuable refuges from predation (Duncan and Holland 2006).

Threats to the Species

Overharvest in commercial and artisanal fisheries and illegal fishing are the most serious threats to Indo-west Pacific scalloped hammerhead sharks. Scalloped hammerhead sharks in general are captured in targeted fisheries and captured as bycatch in pelagic longline fisheries and purse seine fisheries. Miller et al. (2014) noted that significant catches of scalloped hammerheads have and continue to go unrecorded or underreported in many countries outside the United States. Furthermore, Miller et al. (2014), discussed that data on catches of scalloped hammerheads are suspected to underestimate the true catch because many records do not account for discards (example: where the fins are kept but the carcass is discarded) or reflect dressed weights instead of live weights. In addition, many catch records do not differentiate between hammerhead species, or sharks in general, and thus species-specific population trends for scalloped hammerheads are not readily available (Miller et al. 2014).

Contributing to the scalloped hammerhead shark’s biological vulnerability is the fact that these sharks are obligate ram ventilators and suffer very high at-vessel fishing mortality from fisheries where they are not able to continually swim forward (Morgan and Burgess 2007; Macbeth et al. 2009; Miller et al. 2014; Dapp et al. 2016). For example, between 92 to 94% of the hammerhead sharks captured in bottom longline fisheries die at vessel and this does not include post release mortality (Morgan and Burgress 2007). Considering purse seine fisheries, while Hutchinson’s

(2015) study focused on silky sharks, the study showed that sharks confined in the sack portion of the net just prior to loading suffered much higher mortality with only a 6.67% chance of survival after brailing. This highlights the consequences of restricting the movement of hammerhead shark species given their respiratory mode (i.e., obligate ram ventilation). Compared to other chondrichthyans, scalloped hammerhead sharks appear to sustain a higher level of fishing mortality (Miller et al. 2014). Miller et al. (2014) further ranked high at-vessel mortality as the most serious threat to the species.

Catches of Indo-west Pacific scalloped hammerhead sharks from foreign fisheries have decreased since reaching a maximum of 798 t in 2002 (see Figure 2 in Miller et al. 2014). According to shark fin traders, hammerheads are one of the sources for the best quality fin needles for consumption and fetch a high commercial value in the Asian shark fin trade (Abercrombie et al. 2005). In Hong Kong, the world's largest fin trade market, scalloped hammerhead, and smooth hammerhead sharks are found under the "Chun chi" market category, the second most traded fin category in the market (Clarke et al. 2006a). Applying a Bayesian statistical method to the Hong Kong shark fin trade data, Clarke et al. (2006b) estimated that between 1 and 3 million hammerhead sharks, with an equivalent biomass of 60 – 70 thousand metric tonnes, are traded per year.

United States fisheries appear to have less influence on this species status when compared to foreign fisheries. United States fisheries in Alaska and California, and the Hawaii SSLL do not overlap with the species range. Thus these fisheries do not interact with Indo-West Pacific scalloped hammerhead sharks. However, other than the DSLL, the United States WCPO purse seine and ASLL fisheries interact with the Indo-West Pacific scalloped hammerhead.

A total of 14 Indo-West Pacific scalloped hammerhead sharks were caught and positively identified in the United States WCPO purse seine fishery between 2008 and 2018. However, NMFS estimates a total of 41 (95% CI: [31, 51]) Indo-West Pacific scalloped hammerhead sharks were captured between 2008 and 2018 using the Bayesian model approach and is expected to interact with 5 individuals a year with 100% mortality (NMFS 2021a).

Lastly, the ASLL fishery is expected to have interacted with approximately 60 Indo-west Pacific scalloped hammerhead sharks over a 9-year period from 2010 to 2019 (2nd quarter; McCracken 2019b). Most confirmed Indo-west Pacific hammerhead sharks were released alive (73%) and no sharks were recorded as retained. Average at-vessel mortality of Indo-west Pacific hammerhead sharks is 27% in the ASLL fishery. However, the publicly available data compiled by Dapp et al. (2016), estimate 37.6% at-vessel mortality based on the gear type (longline) and the respiratory mode of the animals (i.e., obligate ram-ventilation). Thus the greatest influence on the decline of this species is from foreign fisheries throughout the species range in the western Pacific.

Conservation

Within the WCPO, finning bans have been implemented by Australia, Cook Islands, Micronesia New Zealand, Palau, Republic of the Marshall Islands and Tokelau, as well as by the IATTC and the WCPFC. These finning bans range from requiring fins remain attached to the body to allowing fishermen to remove shark fins provided that the weight of the fins does not exceed 5% of the total weight of shark carcasses landed or found onboard. The WCPFC has implemented several conservation and management measures for sharks with the following objectives (Clarke 2013): (1) promote full utilization and reduce waste of sharks by controlling finning (perhaps as a means to indirectly reduce fishing mortality for sharks); (2) increase the number of sharks that are released alive (in order to reduce shark mortality); and (3) increase the amount of scientific data that is collected for use in shark stock assessments.

Also of relevance is the FAO International Plan of Action for the Conservation and Management of Sharks which recommends that RFMOs carry out regular shark population assessments and that member States cooperate on joint and regional shark management plans.

Summary of the Status

In this section of this biological opinion, we explained that the Indo-West Pacific scalloped hammerhead shark is threatened, and that the species' has suffered substantial historical declines. The species' current trends are unclear but may have stabilized at depressed levels.

Based on the best scientific and commercial data available the Indo-West Pacific scalloped hammerhead shark appears to have undergone substantial declines although recent trends suggest populations may have stabilized (Pacoureau et al. 2021). Evidence of heavy fishing pressure by industrial/commercial and artisanal fisheries, and reports of significant IUU fishing, especially off the coast of Australia, have likely led to overutilization coupled with inadequate regulatory mechanisms are the most concerning threats that may contribute to the extinction risk of the species.

2.2.3 Oceanic Whitetip Shark

Distribution and Population Structure

Oceanic whitetip sharks are distributed in circumtropical and subtropical regions across the world, primarily between 30° North and 35° South latitude (Compagno 1984; Baum et al. 2015; Young et al. 2017), although, the species has been reported as far as 45°N and 40°S in the Western Atlantic (Lessa et al. 1999b). These sharks occur throughout the Central and Western Pacific, including Australia (southern Australian coast), China, New Caledonia, the Philippines, Taiwan, and the Hawaiian Islands south to Samoa Islands, Tahiti and Tuamotu Archipelago, and west to the Galapagos Islands. In the Eastern Pacific, they occur from southern California to Peru, including the Gulf of California and Clipperton Island (Compagno 1984). In the western Atlantic, oceanic whitetips occur from Maine to Argentina, including the Caribbean and Gulf of Mexico. In the central and eastern Atlantic, the species occurs from Madeira, Portugal south to the Gulf of Guinea, and possibly in the Mediterranean Sea. In the western Indian Ocean, the species occurs in waters of South Africa, Madagascar, Mozambique, Mauritius, Seychelles, India, and within the Red Sea.

The geographic distribution of oceanic whitetip shark occurs in a 10° band centered on the equator (Figure 10); their abundance decreases with increasing distance from the equator and increasing proximity to continental shelves (Backus et al. 1956; Strasburg 1958; Compagno 1984; Nakano et al. 1997; Bonfil et al. 2008; Clarke et al. 2011a; Hall and Roman 2013; Tolotti et al. 2013; Young et al. 2017).

Tagging studies have provided information on potential population structure (reviewed in Young and Carlson 2020). Two studies have found evidence of site fidelity in the Atlantic Ocean (Howey-Jordon et al. 2013; Tolotti et al. 2015). Howey-Jordon et al. (2013) found that oceanic whitetip sharks tagged in the Bahamas (1 male and 10 females tagged but the tag on the male shark failed) stayed within 500 km of their tagging site for at least 30 days, at which point they dispersed in different directions across a wide area with some sharks travelling more than 1,500 km from their tagging site. The six tagged sharks that retained their tags for longer than 150 days (*n* = 6) were all located within 500 km of their tagging site when their tags popped off. Similarly, Tolotti et al. (2015) tagged eight oceanic whitetip sharks (sex of sharks was not reported) and found that the tagging and pop-up locations were relatively close to each other, but some individuals traveled long distances (up to 2,500 km) in between these events. Together, these studies suggest that oceanic whitetip sharks can display a high degree of philopatry to certain sites and may not mix with other regional population (Howey-Jordon et al. 2013; Tolotti et al. 2015; Young and Carlson 2020).

Few studies have been conducted on the global genetics and population structure of the oceanic whitetip shark. But those few suggest there may be some genetic differentiation between various ocean basins such as the Indo-Pacific and the Atlantic but limited structuring between adjacent ocean basins such as the East Atlantic and the Indian Ocean (Camargo et al. 2016; Ruck 2016; Sreelekshmi et al. 2020). Camargo et al. (2016) compared the mitochondrial control region in 215 individuals from the Atlantic and Indian Oceans. They found evidence of moderate levels of population structure resulting from restricted gene flow between the western and eastern Atlantic Ocean, they also found evidence of connectivity between the eastern Atlantic Ocean and the Indian Ocean (although the sample size from the Indian Ocean was only nine individuals). This study only used mitochondrial markers, meaning male-mediated gene flow is not reflected in these relationships (Young et al. 2017) although other species in the *Carcharhinus* genus are known to exhibit male-mediated gene flow between populations (Portnoy et al. 2010).

Ruck (2016) compared samples of 171 individual sharks from the western Atlantic, Indian, and Pacific Oceans specifically looking at the mitochondrial control region, a protein-coding mitochondrial region, and nine nuclear microsatellite loci and found no fine-scale matrilineal structure within ocean basins. Ruck (2016) did detect weak but significant differentiation between the Atlantic and Indo-Pacific Ocean populations. An additional analysis of the samples from both studies (Camargo et al. 2016; Ruck 2016) did detect matrilineal population structure within the Atlantic Ocean basin with three lineages, the Northwest Atlantic, the rest of the Western Atlantic, and the Eastern Atlantic Ocean (C. Ruck, personal communication, 2016 as cited in Young et al. 2017).

Sreelekshmi et al. (2020) looked at the genetic diversity of oceanic whitetip sharks along the coast of India and found no significant genetic differentiation, with evidence of substantial gene flow and connectivity. They further indicate that comparing their data with those of Camargo et al. (2016) and Ruck (2016) indicate significant connectivity and gene flow between the Indian Ocean and the East Atlantic. Thus we are unclear of the population structure of oceanic whitetip sharks in the Pacific Ocean, and specifically if there is gene flow between the West and East Pacific Ocean.

While much more work is needed to fully understand the species population structure, Young et al. (2017) concluded the studies up to that point did not provide "unequivocal evidence for genetic discontinuity or marked separation between Atlantic and Indo-Pacific subpopulations." The more recent work by Sreelekshmi et al. (2020) is similar. While there is not unequivocal evidence, we assume the weak differentiation found by Ruck (2016) indicates oceanic whitetip sharks in the Pacific Ocean may be their own population. Frequently distinctions are made between the oceanic whitetip sharks in the East Pacific and the West Pacific; however, this distinction appears to be one of convenience based on fishery management areas and may be biologically arbitrary. However, there is currently no scientific evidence indicating a lack of connectivity across the Pacific Ocean.

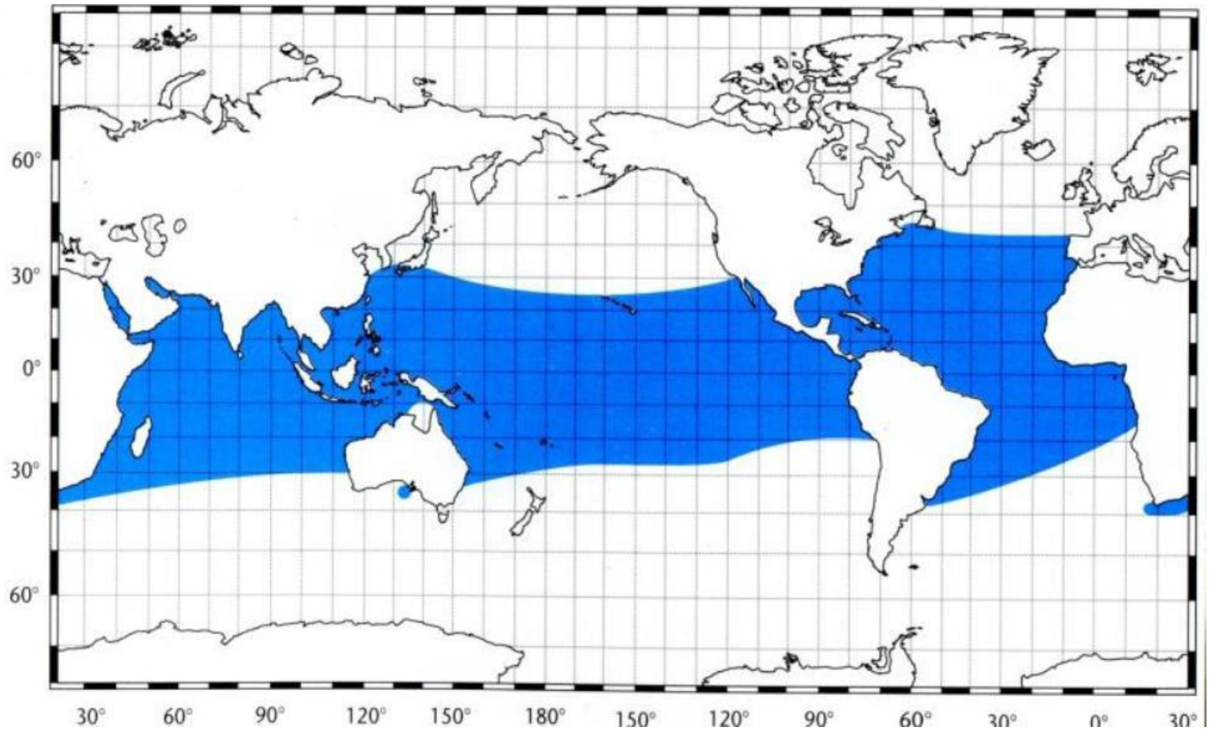

Figure 10. Geographical distribution of the oceanic whitetip shark (Last and Stevens 2009).

Status and Trends

Oceanic whitetip sharks were listed globally as threatened in 2018. Historically, oceanic whitetip sharks were described as one of the most abundant species of shark found in warm tropical and sub-tropical waters of the world (Backus et al. 1956; Strasburg 1958). Oceanic whitetip sharks occur throughout their range with no evidence of range contraction or range erosion (gaps within the species' range that form when populations become extinct locally or regionally; Lomolino and Channell 1995, 1998; Collen et al. 2011). However, recent estimates of their abundance suggest the species has experienced significant historical declines throughout its range. Declines in abundance range from 80-96% across the Pacific Ocean since the late 1990s (Clarke et al. 2012; Rice and Harley 2012; Brodziak et al. 2013; Hall and Roman 2013; Rice et al. 2015; Tremblay-Boyer et al. 2019), 50-88% across the Atlantic Ocean (Baum and Meyers 2004; Santana et al. 2004; Cortes et al. 2007; Driggers et al. 2011); and have been variable across the Indian Ocean, ranging from 25-40% (Anderson et al. 2011; IOTC 2011, 2015; Ramos-Cartelle et al. 2012; Yokawa and Semba 2012).

The only formal stock assessments for the Pacific represent a portion of the total Pacific Ocean population–the West Pacific portion of the population's range (aka. the West Pacific stock). Unfortunately, it remains unclear how much of the total Pacific Ocean oceanic whitetip population this one population assessment covers. As noted above, oceanic whitetip sharks occur primarily between 30° North and 35° South latitude. We used ArcGIS to estimate the area of the

Pacific Ocean between these latitudes, as well as, the area of the WCPO between these latitudes. From this assessment, we estimate that the area of oceanic whitetip shark habitat in the WCPO represents about 60% of the total habitat within the Pacific Ocean.

Two stock assessments have been conducted for the oceanic whitetip shark in the WCPO to date and the conclusions have been reinforced by additional studies (Clarke et al. 2011b; Brodziak et al. 2013; Rice et al. 2015; Tremblay-Boyer et al. 2019). Most recently, Tremblay-Boyer et al. (2019) utilized the Stock Synthesis modeling framework (Methot Jr and Wetzel 2013), which is an integrated age-structured population model. The population dynamics model was informed by three sources of data: historical catches, time series of CPUE and length frequencies. The longline fishery was split into bycatch and target fleets, and the purse-seine fishery into fleets of associated and unassociated sets. This assessment also included scenarios of discard mortality assuming 25%, 43.75% and 100% mortality on discards. The stock of oceanic whitetip shark was found to be overfished and undergoing overfishing based on SB/SBMSY and F/FMSY reference points. The current spawning stock biomass (232–-507 metric tonnes) is predicted to be below 5% of the unfished spawning biomass and the population could go extinct over the long-term based on current levels of fishing mortality (Tremblay-Boyer et al. 2019). The most recent assessment concluded that total biomass in 2010 was 19,740 metric tons and that biomass declined to 9,641 metric tons by 2016.

In previous biological opinions, NMFS has estimated that the biomass translates to 200,000 sharks (NMFS 2019a) and 264,318 sharks (NMFS 2021a), following an analysis in FAO (2012). The stock assessment conducted by Tremblay-Boyer et al. (2019) included 648 model runs accounting for assumptions about life-history parameters and impact of fishing underpinning the assessment. Using the underlying data from these 648 models in their structural uncertainty grid in Tremblay-Boyer et al. (2019), the authors subsequently estimated the median value of the current total number of individuals in the WCPO (*n*= 775,214) (see NMFS 2020b). We consider this estimate as the current best available scientific information and use it as our best estimate of the size of the WCPO portion of the Pacific Ocean population of oceanic whitetip sharks and is in alignment with previous supplements (see NMFS 2022b, 2022c).

Assuming a similar density of oceanic whitetip shark in the East Pacific to that of the WCPO, and using the proportion described above that the area of the WCPO between the latitudes where oceanic whitetip sharks are found represents 60% of habitat in the entire Pacific Ocean, we estimate a total population size of 1,292,023 ([775,214/60] x100) oceanic whitetip sharks in the Pacific Ocean. However, given that this estimate requires an assumption regarding the density of oceanic whitetip sharks in the East Pacific, in our analysis we consider both 774,214 as a minimum population estimate and 1,292,023 as an upper estimate of the population size assuming the densities of sharks in the East Pacific is similar to that of the WCPO.

Rice et al. (2021) estimate that WCPO oceanic whitetip sharks will decline by an additional 13.3% (mean; 14.6% median) over 10 years which equates to an annual decrease of 1.4% (mean; 1.6% median) assuming incidental captures and mortalities remain the same as 2016. If longline fishery mortalities are decreased by 10% across the WCPO, Rice et al. (2021) estimate that the WCPO population will only decline by an additional 0.4% (mean; 1.2% median) which equates to annual declines of 0.04% (mean; 0.13% median). If longline fishery mortalities are decreased further, by 20% across the WCPO, Rice et al. (2021) estimate that the WCPO population will increase by 4.2% (mean; 3.3% median) over the next 10 years, which equates to an annual increase of 0.46% (mean; 0.36% median). Rice et al. (2021) indicate that recent catch is likely bounded by the latter two scenarios, or reductions of between 10% and 20% due to adoptions of CMMs and slight decreases in the amount of longline fishing effort.

More recently, Bigelow et al. (2022) updated the projections of Rice et al. (2021) with contemporary estimates of at-vessel and post-release mortality rates, and catch reductions facilitated by switching to monofilament leaders. Their results are summarized by projections of the ratio of spawning biomass (projected to 2031) to the equilibrium unfished spawning biomass (i.e. the biomass of an unfished population). This provides a relative measure of the size of the spawning biomass of a population whereby increasing ratios indicate higher biomass. The mean values of these ratios increase from 0.039 estimated for 2016 to 0.118 with updated assumptions regarding at-vessel and post-release mortality reductions and prohibition of wire leaders and shark lines (Figure 11; see Table 3 of Bigelow et al. 2022). These results are based on optimistic post-interaction mortality rates of 3.4 to 8.1% with an at-vessel mortality rate of 19.2% (see Table 1 of Bigelow et al. 2022). The implementation of CMM-2022-04 is anticipated to improve the survival of released sharks throughout the WCPO by eliminating wire leaders and shark lines.

We believe this new information provided by Bigelow et al. (2022) constitutes the best available. However, Bigelow et al. (2022) do not provide specific population trends, only indicating that the trends in spawning biomass ratios are anticipated to be positive (Figure 11). Additional years of data are needed before we can calculate an estimated population trend. Given the uncertainty in the applicability of the assumption made by Bigelow et al. (2022) to the broader WCPO fisheries, we consider it reasonable to assess the range of population trends presented in Rice et al. (2021) for reductions in fishery mortality between 10 and 20%. Therefore, we focus our analysis on the scenarios presented by Rice et al. (2021) whereby the actual population trend is between a declining rate of 0.13% per year (median value for 10% reduction in fishery mortalities) and an increase rate of 0.36% per year (median value for 20% reduction in fishery mortalities). These numbers include the loss of individuals from the DSLL as currently operated.

Historic declines in abundance of WCPO oceanic whitetip sharks are attributable to impacts from pelagic fisheries, both longline and purse seine fisheries as well as smaller fisheries such as troll, handline and shortline fisheries. As noted above in the Distribution and Population Structure section, it is possible that oceanic whitetip sharks are philopatric; therefore, the declines in abundance may have resulted in localized depletions resulting in a loss of genetic diversity, and changes in distribution.

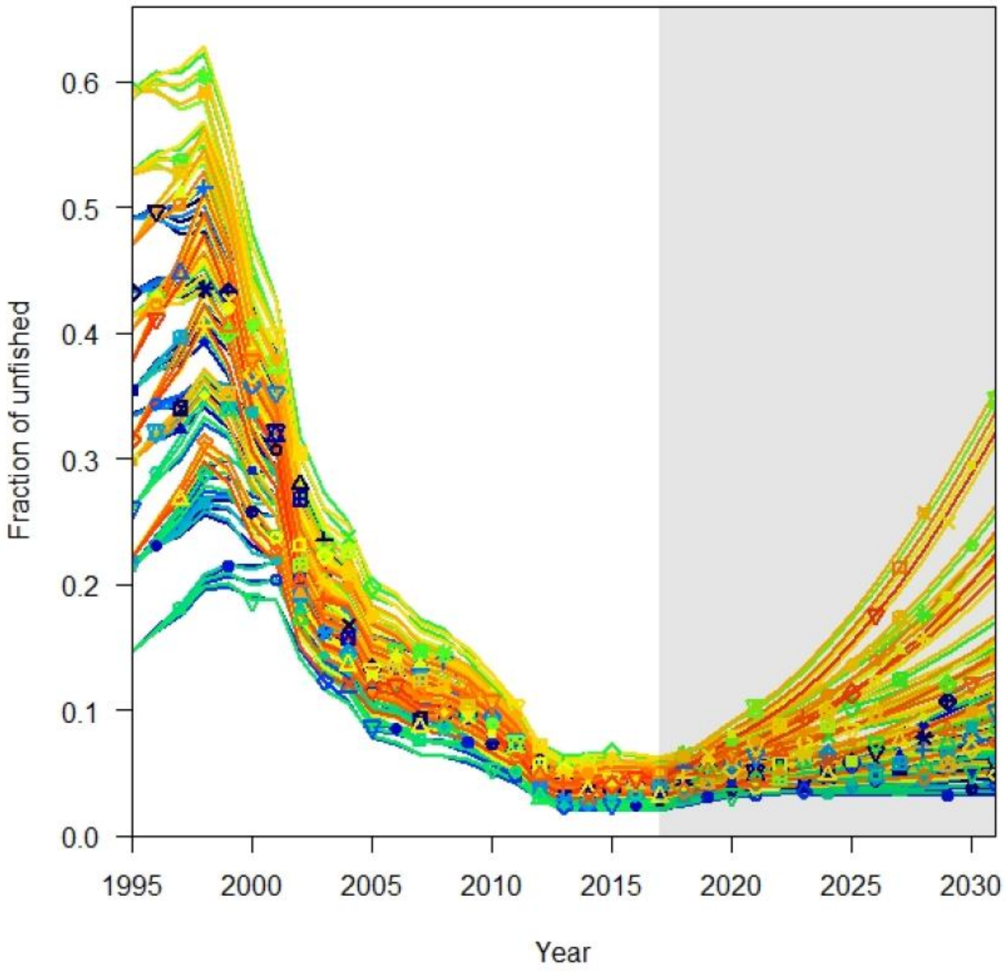

Figure 11. Projected ratios of spawning biomass (projected to 2031) to the equilibrium unfished spawning biomass for WCPO oceanic whitetip sharks with updated at-vessel and post-release mortality rates and the prohibition of wire branchlines and shark line (Figure 7 in Bigelow et al. 2022).

Population Dynamics

Oceanic whitetip sharks are a relatively long-lived, late maturing species with low-to-moderate productivity. These sharks are estimated to live up to 19 years (Seki et al. 1998; Lessa et al. 1999a; Joung et al. 2016), although their theoretical maximum age has been estimated to be approximately 36 years. Female oceanic whitetip sharks reach maturity between 6 and 9 years of age, although this varies with geography (Seki et al. 1998; Lessa et al. 1999a; Joung et al. 2016) and give birth to live young after a very lengthy gestation period of 9 to 12 months (Bonfil et al. 2008; Coelho et al. 2009). The reproductive cycle is thought to be biennial, with sharks giving birth every one or two years in the Pacific Ocean (Seki et al. 1998; Chen 2006 as cited in Liu and Tsai 2011) and alternate years in other ocean basins. Litters range from 1 to 14 pups with an average of 6 (Seki et al. 1998; Lessa et al. 1999a; Juong et al. 2016). Their generation time has been estimated to range between 7 and 11 years (Cortes 2002; Smith et al. 2008).

Diving and Social Behavior

Oceanic whitetip sharks generally prefer mixed surface layers where temperatures typically remain greater than 20°C to 150 m in depth, with brief deep dives into deeper waters (Howey-Jordan et al. 2013; Howey et al. 2016; Tolotti et al. 2017; Young et al. 2017). The maximum recorded dive of the species was to a depth of 1,082 m (Howey-Jordan et al. 2013). Aggregations of oceanic whitetip sharks have been observed in the Bahamas (Madigan et al. 2015; Young et al. 2017), but there is no evidence of social interactions between individuals or groups of individuals.

Threats to the Species

The primary threat to oceanic whitetip sharks worldwide is intentional targeting and incidental bycatch in commercial fisheries (Young et al. 2017; Young and Carlson 2020). Because of their preferred distribution in warm, tropical waters, and their tendency to remain at the surface, oceanic whitetip sharks have high encounter and mortality rates in fisheries throughout their range. They are frequently caught as bycatch in many global fisheries, including pelagic longline fisheries targeting tuna and swordfish, purse seine, gillnet, and artisanal fisheries. They are also a preferred species for the international fin trade, discussed in more detail below. Impacts to the species from fisheries (U.S. and foreign) that overlap the Action Area will be discussed in the Environmental Baseline, as appropriate.

Bycatch-related mortality in longline fisheries are considered the primary drivers for these declines (Clarke et al. 2011b; Rice and Harley 2012; Young et al. 2017), with purse seine (11,139 observed captures from 1995 to 2015; Tremblay-Boyer and Brouwer 2016) and artisanal fisheries being additional sources of mortality. In addition to bycatch-related mortality, the oceanic whitetip shark is a preferred species for opportunistic retention because its large fins obtain a high price in the Asian fin market, and comprises approximately 2% of the global fin trade (Clarke et al. 2006a). Despite finning bans and retention prohibitions, this high value and demand for oceanic whitetip fins incentivizes the opportunistic retention and subsequent illegal finning of oceanic whitetip sharks when caught, and thus represents the main economic driver of mortality of this species in commercial fisheries throughout its global range. We note that retention/finning is not practiced in U.S. fisheries.

As a result, oceanic whitetip biomass has declined by 88% since 1995 (Tremblay-Boyer et al. 2019). Currently, the population is overfished and overfishing is still occurring throughout much of the species' range (Rice and Harley 2012; Tremblay-Boyer et al. 2019; 83 CFR 46588). As a result, catch trends of oceanic whitetip shark in both longline and purse seine fisheries have, until recently (i.e. Bigelow et al. 2022), significantly declined, with declining trends also detected in some biological indicators, such as biomass and size indices (Clarke et al. 2011b; Young et al. 2017).

U.S. fisheries in the Pacific that capture oceanic whitetip sharks include the SSLL, DSLL and ASLL fisheries, as well as the U.S. purse seine fisheries. The SSLL is estimated to interact with up to 102 oceanic whitetip sharks a year (95th percentile; NMFS 2019a). The DSLL is estimated to interact with a mean of 1,708 (95th percentile: 3,185) oceanic whitetip sharks annually (McCracken 2019a; NMFS 2018b), though see the discussion in the Effects of the Action section regarding the effect of the fishery switching to monofilament leaders. The ASLL will be discussed in the Environmental Baseline, as that fishery overlaps the Action Area. No interactions have been noted with oceanic whitetip sharks in any West Coast Highly Migratory Species fishery to date (C. Villafana and C. Fahy pers. comm. to J. Rudolph; March 7, 2019).

Overall, the species has experienced significant historical abundance declines in all three ocean basins (Atlantic, Pacific, and Indian Oceans) due to overutilization from fishing pressure and inadequate regulatory mechanisms to protect the species (Hazin et al. 2007; Lawson 2011; Clarke et al. 2012; Hasarangi et al. 2012; Hall and Roman 2013; Young et al. 2017; Tremblay-Boyer et al. 2019). Their population dynamics –long-lived and late maturing with low-to-moderate productivity– makes this species particularly vulnerable to harvests that target adults and limits their ability to recover from over-exploitation.

Conservation

Due to reported population declines driven by the trade of oceanic whitetip shark fins, the oceanic whitetip shark was listed under Appendix II of CITES in 2013. This listing went into effect as of September 2014.

Within the WCPO, finning bans have been implemented by the United States, Australia, Cook Islands, Micronesia New Zealand, Palau, Republic of the Marshall Islands and Tokelau, as well as by the IATTC and the WCPFC. These finning bans range from requiring fins remain attached to the body to allowing fishermen to remove shark fins provided that the weight of the fins does not exceed 5% of the total weight of shark carcasses landed or found onboard. The WCPFC has implemented several conservation and management measures for sharks with the following objectives (Clarke 2013): (1) promote full utilization and reduce waste of sharks by controlling finning (perhaps as a means to indirectly reduce fishing mortality for sharks); (2) increase the number of sharks that are released alive (in order to reduce shark mortality); and (3) increase the amount of scientific data that is collected for use in shark stock assessments.

Also, specific to oceanic whitetip sharks, CMM 2011-04 prohibits WCPFC vessels from retaining onboard, transshipping, storing on a fishing vessel, or landing any oceanic whitetip shark, in whole or in part, in the fisheries covered by the Convention. This CMM was later replaced in 2019 by CMM-2019-04 which was in-turn was replaced in 2022 by CMM-2022-04 for all sharks. The measure retains the retention prohibition for oceanic whitetip sharks, and includes additional measures on minimizing bycatch (including some gear restrictions), implementing safe release practices, and prohibiting wire leaders and shark lines for longline fishing.

Summary of the Status

In this section of this biological opinion, we explained that the oceanic whitetip shark is globally threatened, and that the species' population has suffered substantial historic declines, however recent data suggests evidence of a reversal of this decline (Bigelow et al. 2022) for the WCPO. We used our knowledge of the species' demography and population ecology to capture the primary factors that appear to determine the oceanic whitetip shark population dynamics. Primary threats that have contributed to the species' decline and listing include overutilization due to fisheries bycatch and opportunistic trade of the species' fins, as well as inadequate regulatory mechanisms related to commercial fisheries management and the international shark fin trade (Young et al. 2017).

As a result of fishing mortality, oceanic whitetip biomass has declined by 86% in the Western and Central Pacific Ocean, with an estimated decline of 1.6% per year (Young et al. 2017; Rice et al. 2020). The stock is overfished, and overfishing may still be occurring (Rice and Harley 2012; Trembolay-Boyer et al. 2019; Bigelow et al. 2022; 83 CFR 46588). In a recent assessment, Bigelow et al. (2022) suggest the recent initiatives that prohibit retention, improve handling and release conditions, and shifts to monofilament leaders are likely to result in increasing trends for WCPO oceanic whitetip sharks. Historically, catch trends of oceanic whitetip shark in both longline and purse seine fisheries have significantly declined, with declining trends also detected in some biological indicators, such as biomass and size indices (Clarke et al. 2011a; Young et al. 2017). Similar results between analyses of The Pacific Community observer data from the larger Western and Central Pacific and the observer data from the Hawaii-based pelagic longline fishery suggest that the stock decline of oceanic whitetip sharks in this portion of its range is not just a localized trend, but rather a Pacific-wide phenomenon (Brodziak et al. 2013). Based on Bigelow et al. (2022), these trends may turn around, however fishery bycatch, direct harvest and finning continue to be the primary threats to oceanic whitetip sharks.

2.2.4 Green Sea Turtle

In 1978, green sea turtles were listed as threatened (43 FR 32800, July 28, 1978), except for breeding populations that occur in Florida and the Pacific coast of Mexico, which were listed as endangered. In 2016, NMFS and the FWS replaced the global green sea turtle listing with 11 distinct population segments (DPSs) (Figure 12). These 11 DPSs are demographically, spatially, and genetically independent. As a result, they have been listed as separate "species" for the purposes of the ESA. Eight of these DPSs are listed as threatened (light- colored polygons in Figure 12) and three are listed as endangered (dark-colored polygons in Figure 12). The DPSs of green sea turtles whose individuals are most likely to occur in the range of the Hawaii-based DSLL fishery are limited to those that occur in the Pacific Ocean. Specifically, based on genetic analyses of tissue samples collected from green turtles captured in the DSLL fishery (Dutton, pers. comm. 2018), the green turtles captured in the fishery can represent up to 6 of the 11 DPSs of green turtle that have been listed as endangered or threatened: (1) Central North Pacific; (2) Central West Pacific; (3) Central South Pacific; (4) East Pacific; (5) East Indian-West Pacific; (6) Southwest Pacific green turtles (see Section 4.3.4 for details on this assessment).

However, because of similarities in their population dynamics, diving and social behavior, and vocalizations and hearing, we present this information in this introductory section to avoid having to repeat it for each of the six DPS of green turtle that may be affected by the proposed action.

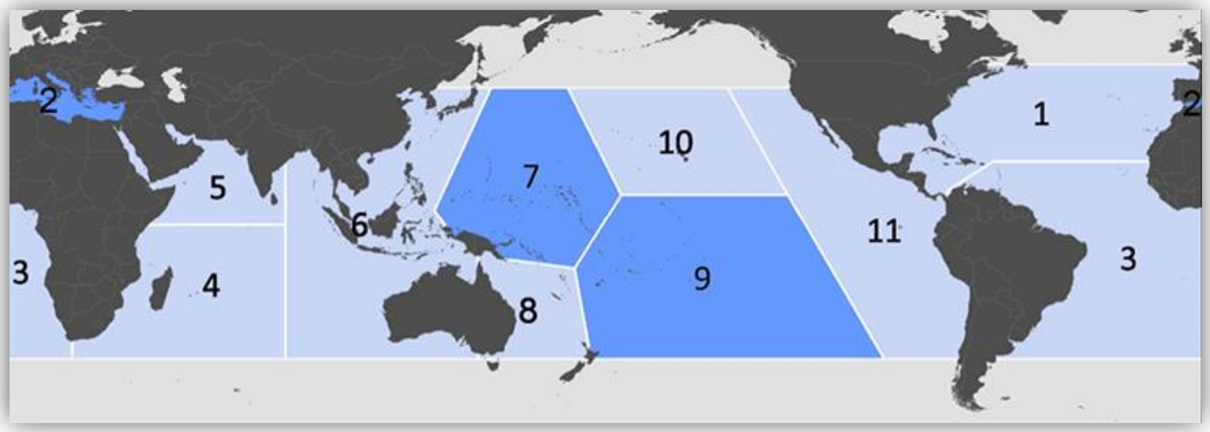

Figure 12. Overview of listed green sea turtle DPSs; (1) North Atlantic; (2) Mediterranean; (3) South Atlantic; (4) Southwest Indian; (5) North Indian; (6) East Indian-West Pacific; (7) Central West Pacific; (8) Southwest Pacific; (9) Central South Pacific; (10) Central North Pacific; and (11) East Pacific. Light blue indicates threatened populations and dark blue indicates endangered populations.

During their pelagic phase, juvenile green turtles have a varied diet that includes planktonic material including crustaceans, jellyfish, and ctenophores. Sub-adults and adult green turtles are attracted to baited hooks in tuna longline fisheries (Beverly and Chapman 2007). The deepest dives recorded for green turtles are from adults migrating from the MHI to the Northwest Hawaiian Islands. Several of these turtles dove to greater than 100 m depth in pelagic areas, where they may have been feeding on plankton, resting, or avoiding predators (Rice and Balazs 2008). Hatase et al. (2006) observed nighttime dives >20 m and indicated that green turtles were not only resting, but feeding on macroplankton that exhibit diel migrations. Neritic green turtles typically forage in shallow coastal areas, primarily on algae and seagrass.

General Population Dynamics

The population dynamics of green sea turtles, like all of the other sea turtles this biological opinion considers, are usually based on adult females, rather than their male counterparts. The spatial structure of male sea turtles and their fidelity to specific coastal areas is unknown; however, we discuss sea turtle populations based on the nesting beaches that female sea turtles return to when they mature. Because the patterns of increase or decrease in the abundance of sea turtle nests over time are determined by internal dynamics rather than external dynamics, we make inferences about the growth or decline of sea turtle populations based on the status and trend of their nests.

The life cycle of green sea turtles encompasses several stages: eggs, hatchlings, juveniles, sub-adults, and adults. The dynamics are determined by the number of eggs that survive to adults and the number of adults that produce those eggs. Adult females can take 20 to 40 years to become sexually mature. Once mature, they generally reproduce every 2-4 years, lay 2-5 clutches of eggs per season, with 80-120 eggs per clutch. Once mature, they are expected to remain reproductively active for 17 to 23 years (Carr et al. 1978; Fitzsimmons et al. 1996). Therefore, it might take 20 to 40 years for the effects of an action that affected the abundance of eggs to affect the abundance of adults in this species. Further, because most of the data we collect on this species consists of counts of adult females on nesting beaches, we have limited understanding of the true dynamics of populations of these species.

As with other sea turtle species, population censuses are primarily focused on adult females as they can be monitored on nesting beaches. Seminoff et al. (2015) present the best available information for adult female population sizes for green sea turtle species. We parameterized a post-breeding, age-based Leslie matrix model using the fecundity values in Balazs et al. (2015) and the survival rates and age at maturity of Piacenza et al. (2016) and conducted 10,000 simulations drawing from triangular distributions for the survival rates and age to maturity ranges specified in Table 4. All other parameters were held stable as specified in Table 4 across the simulations. The right eigenvector of a Leslie matrix model provides the stable age distribution of the population from which we can derive the percent of the population that are adult females. We then use the adult female population sizes from Seminoff et al. (2015) to estimate the total population size of individuals greater than one year old for each DPS. The results of this analysis are presented in the DPS-specific sections below.

Table 4. Parameters used to estimate total population size for green sea turtle DPSs.

Parameter	Mean	Range	Source
Remigration Interval	4	NA	Balazs et al. (2015)
Nests per Year	4	NA	Balazs et al. (2015)
Eggs per Nest	104	NA	Balazs et al. (2015)
Nest Survival Rate	0.54	NA	Balazs et al. (2015)
Sex Ratio (proportion female)	0.516	NA	Balazs et al. (2015)
First Year Survival	0.35	NA	Piacenza et al. (2016)
Pelagic Juvenile Survival	0.80	NA	Piacenza et al. (2016)
Neritic Juvenile Survival	0.824	0.799 – 0.967	Piacenza et al. (2016)
Subadult Survival	0.876	0.799 – 0.98	Piacenza et al. (2016)
Adult Survival	0.929	0.924 – 0.933	Piacenza et al. (2016)
Time to Maturity	31	17 - 41	Piacenza et al. (2016)

General information on effects of climate change for all green sea turtles in the Pacific

Effects of climate change include, among other things, sea surface temperature increases, the alteration of thermal sand characteristics of beaches (from warming temperatures), which could result in the reduction or cessation of male hatchling production (Hawkes et al. 2009; Poloczanska et al. 2009), and a significant rise in sea level, which could significantly restrict green sea turtle nesting habitat. While sea turtles have survived past eras that have included significant temperature fluctuations, future climate change is expected to happen at unprecedented rates, and if sea turtles cannot adapt quickly, they may face local to widespread extirpations (Hawkes et al. 2009). Impacts from global climate change induced by human activities are likely to become more apparent in future years (IPCC 2018, 2022). However, in some areas like the primary nesting beach in Michoacan, Mexico (Colola), the beach slope aspect is extremely steep and the dune surface at which the vast majority of nests are laid is well elevated. This site is likely buffered against short-term sea level rise as a result of climate change. In addition, many nesting sites are along protected beach faces, out of tidal surge pathways. For example, multiple nesting sites in Costa Rica and in the Galapagos Islands are on beaches that are protected from major swells.

Diving and Social Behavior

Based on the behavior of post-hatchlings and juvenile green turtles raised in captivity, it is presumed that those in pelagic habitats live and feed at or near the ocean surface, and that their dives do not normally exceed several meters in depth (NMFS and FWS 1998). The maximum recorded dive depth for an adult green turtle was 138 meters (Rice and Balazs 2008; Hochscheid 2014), while subadults routinely dive 20 meters for 9-23 minutes, with a maximum recorded dive of 66 minutes (Brill et al. 1995 in Lutcavage and Lutz 1997).

2.2.5 Central North Pacific Green Sea Turtle

Distribution and Population Structure

The distribution of Central North Pacific green turtles encompasses the Hawaiian Archipelago and Johnston Atoll. It is bounded by a four-sided polygon with open ocean extents reaching to 41˚N,169˚E in the northwest corner, 41˚N, 143˚W in the northeast, 9˚N, 125˚W in southeast, and 9˚N, 175˚W in the southwest (Figure 13). About 96% of the nesting occurs in French Frigate Shoals and half of the nesting in French Frigate Shoals occurred on East Island. Based on spatially concentrated (limited) distributions of nesting and lack of evidence of genetic substructuring, Seminoff et al. (2015) concluded that the Central North Pacific green sea turtle consists of a single population.

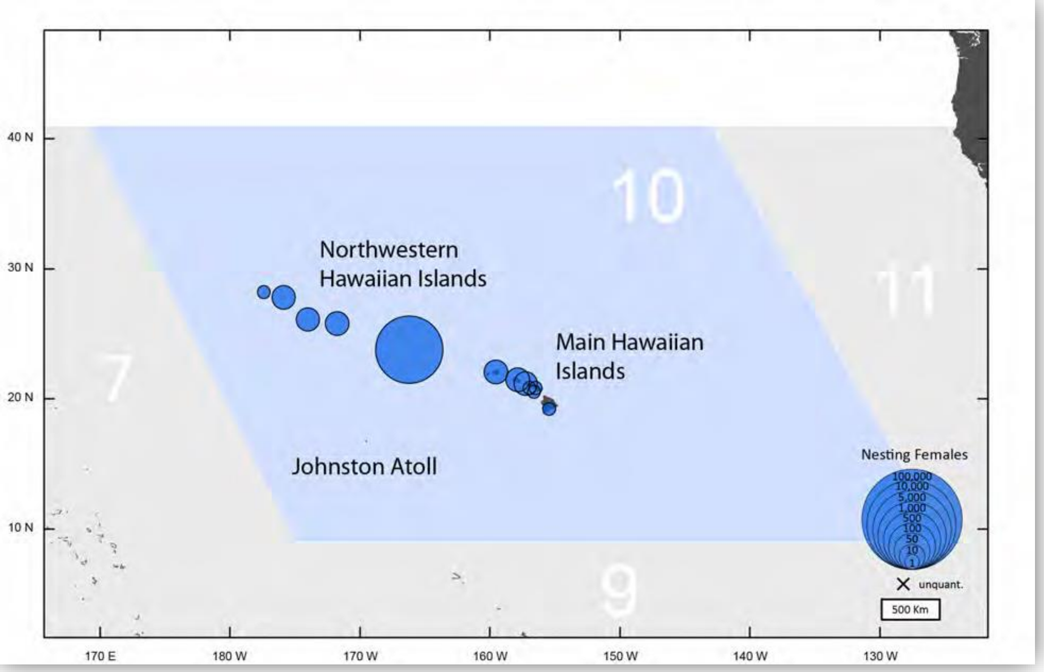

Figure 13. Nesting distribution of Central North Pacific green sea turtles. Size of circles indicates nesting estimated nester abundance. The geographic range of this DPS encompasses the entire Hawaiian archipelago and Johnston Atoll.

Based on genetic samples from 14 green sea turtles that interacted with the DSLL fishery between 2005 and 2017, NMFS estimates that between 4% and 43% of the turtles caught in the fishery may have been Central North Pacific green sea turtles (P. Dutton pers. comm. June 29, 2018).

Status and Trends

The Central North Pacific green sea turtle is listed as threatened. The IUCN Redlist categorizes this DPS as “near threatened” based on current population estimates (Chaloupka and Balazs 2007; Chaloupka and Pilcher 2019). During the most recent status review the abundance of nesting females in this DPS was estimated at 3,846 over 13 nesting sites (Seminoff et al. 2015). This is a substantial increase since nesting surveys at the French Frigate Shoals index beach began in 1973 (Balazs and Chaloupka 2004); however, recent changes in the availability and distribution of suitable nesting beaches puts continued increases in doubt (see discussion below).

Counts of nesting female green turtles have increased over the past 40 years by an average of 5.4% per year (Balazs et al. 2015). Estimates of the in-water abundance of green turtles have increased in a pattern that is similar to that of nesting trends (Balazs and Chaloupka 2004; Chaloupka et al. 2008a). In addition, there has been a dramatic increase in the number of basking turtles in the MHI and the Northwest Hawaiian Islands (Whittow and Balazs 1982; Parker and Balazs 2010). These increases have been attributed to increased survivorship (since harvesting of turtles in foraging areas was prohibited in the mid-1970s) and cessation of habitat damage at the French Frigate Shoals rookery since the early 1950s (Balazs and Chaloupka 2004).

Although the trajectory of the nesting population has been increasing in recent years, the distribution of this green turtle has declined substantially: as much as 80% of historically major nesting sites have been extirpated or the abundance of nesting females at these sites has declined substantially between 1875 and 2012 (Kittinger et al. 2013). Nesting that once occurred across a wide geographic area has been largely concentrated in a small geographic area: more than 96% of nesting occurs at French Frigate Shoals in the Northwest Hawaiian Islands, which is a low-lying coral atoll that is susceptible to erosion, geomorphological changes, and sea level rise (Seminoff et al. 2015).

A range contraction of this magnitude usually places vulnerable species at greater risk of extinction because it makes them more susceptible to being destroyed by random events such as severe storms (Udvardy 1969; Towns and Daugherty 1994; Lomolino and Channel 1995, 1998; Gaston et al. 2000; O'Grady et al. 2004; Reed 2004; Collen et al. 2011; Seminoff et al. 2015). Indeed, in 2018, Hurricane Walaka destroyed and submerged East Island which had supported the largest rookery in French Frigate Shoals and supported about half of the nesting of Central North Pacific green turtle nesting.

The consequences of this event are uncertain, but the fate of this DPS of green turtle depends on (1) whether the female green turtles that once nested on East Island relocate their nesting to other beaches, successfully nests at those beaches, and produce a net increase in the number of live hatchlings those nests produce; (2) how many years pass before this occurs (if it occurs at all); and (3) where this new nesting occurs (if it occurs). Since 2018, Central North Pacific green sea turtles have been observed nesting at Tern Island in French Frigate Shoals. In 2019, 263 females were observed nesting and in 2021, 645 females were observed nesting (NMFS unpublished data; no monitoring occurred in 2020 due to COVID 19 restrictions on field camps). These numbers, while representing only two years of data, suggest that this DPS is successfully using alternate nesting habitat following the loss of East Island.

The collapse of their geographic range, the concentration of nesting to a small geographic area, and the destruction of the most important nesting area for the DPS has placed Central North Pacific green turtles in a natural experiment whose outcome remains uncertain, although preliminary data suggest a positive outcome. Tiwari et al. (2010) suggest that other nesting beaches that could support substantially more nesting activity are available to Central North Pacific green turtles within their current distribution. At the moment, we can only continue to watch to see how the turtles that once nested on East Island respond.

We estimate total population size by using the age-based Leslie matrix model described above and the adult female population estimate of 3,846 (Seminoff et al. 2015). As we are reasonably certain the trend for this population is increasing, we restricted the 10,000 population simulations to those demonstrating an increasing population growth rate to estimate a total mean population size of 682,296 (range 60,965 to 1,145,988) individuals greater than one year old.

Threats to the Species

Phenomena related to climate change, including changing storm dynamics and intensity, and loss of nesting habitat, are emerging concerns for habitat throughout the Hawaiian Archipelago (Baker et al. 2006; Keller et al. 2009). Weather events, such as storms and seasonal changes in current patterns, can also reduce or eliminate sandy beaches, degrade turtle nesting habitat, and cause barriers to adult and hatchling turtle movements on affected beaches.

As described in Seminoff et al. (2015), Baker et al. (2006) examined the potential effects of sea level rise in the Northwest Hawaiian Islands and found that the primary nesting area for the Central North Pacific population is threatened by sea level rise through possible loss of nesting habitat. For example, Whale-Skate Island at French Frigate Shoals was formerly a primary nesting site for these green turtles, but the island has subsided and is no longer available for nesting (Kittinger et al. 2013). Trig, Gin, and Little Gin could lose large portions of their area (Baker et al. 2006). Additionally, habitat degradation resulting from the release of contaminants contained in landfills and other areas of the Northwest Hawaiian Islands could occur as the islands erode or are flooded from sea level rise (Keller et al. 2009).

Historically, these green turtles were subjected to intense harvesting pressure first by indigenous Polynesians, then by Europeans as they explored the region, then again during World War II (Kittinger et al. 2011, 2013; Van Houtan et al. 2012). By 1950, nesting was essentially extirpated everywhere except on a single remote atoll. Between 1948 and 1974, the cumulative harvest of green turtles was estimated at between 5,000 and 6,000 turtles or 180 to 230 turtles per year. Before the population collapsed, these annual totals would have been collected in a single day (Balazs 1976; Clapp and Wirtz 1975; Kittinger et al. 2011; Van Houtan et al. 2012). Since they were listed pursuant to the ESA in 1978, the harvest of green turtles has been illegal, although anecdotal reports suggest that some harvesting continues (Seminoff et al. 2015).

More recently, these green turtles are captured in pelagic longline fisheries, coastal gill net fisheries, and hook-and-line fishing. Additional green turtles are captured in gill nets, with some of them dying as a result of their capture. Chaloupka et al. (2008b) report that between 1982 and 2002 approximately 7% of strandings were attributed to hook-and-line fishing gear induced trauma, and 5% for gill net fishing gear-induced trauma. Nearshore fishery interactions have increased over time with over 60 turtles stranded in 2011 as a result of hook and line interactions, and 46 turtles in 2012 (Francke et al. 2013; Ikonomopoulou et al. 2013).

Many United States fisheries capture green sea turtles in the Pacific. However, only four - the MHI bottomfish, the West Coast Region DSLL, the Hawaii DSLL (this analysis), and the SSLL fisheries are expected to interact with this DPS. As all of these fisheries occur in portions of the Action Area, they will be discussed further in the Environmental Baseline although we provide an overview of the information relevant to the DPS' status here as well.

NMFS concluded that the MHI bottomfish fishery is not likely to adversely affect the DPS (NMFS 2013) and the West Coast Region DSLL is expected to capture and kill up to 1 Central North Pacific green sea turtle over 10 years although it is unclear whether the interaction would be attributed to the East Pacific or Central North Pacific green sea turtle (NMFS 2016a). Therefore, NMFS assessed effects from that fishery to both DPSs of green sea turtles (see NMFS 2016a). Furthermore, the SSLL is expected to capture 5 green sea turtles (all DPSs considered (*n*= 6), including the Central North Pacific) annually with one of those turtles expected to die (NMFS 2019a). Between 2004 and 2018, the SSLL fishery captured 10 green sea turtles and we expect between 20-64% of those turtles caught could be from the Central North Pacific (NMFS 2019a). All ten green sea turtles caught in the Hawaii SSLL fishery between 2004 and 2018 were released alive with no gear attached.

Historically, the Hawaii DSLL fishery has observed 25 green sea turtle interactions between 2004 and 2022 with an estimated 128 green sea turtle interactions with this fishery during this time frame (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a). NMFS estimates about 95% (95% CI = 77 to 99%) of green turtles captured in the DSLL fishery have been dead when they were hauled back to the fishing vessel based on the number of observed mortalities (this analysis).

While other United States fisheries capture green sea turtles in the Pacific, no other United States fisheries are expected to capture or kill Central North Pacific green sea turtles.

2.2.6 Central South Pacific Green Sea Turtle

Distribution and Population Structure

The Central South Pacific green sea turtle is distributed north from northern New Zealand to Fiji, Tuvalu, and Kiribati and east to include French Polynesia (Figure 14). Its open ocean polygonal boundary endpoints are (clockwise from the northwest-most extent): 9°N, 175°W to 9°N, 125°W to 40°S, 96°W to 40°S, 176°E, to 13°S, 171°E, and back to the 9°N, 175°W northwest extent. The range of this DPS includes a longitudinal expanse of 7,500 km—from Easter Island, Chile in the east to Fiji in the west, and encompasses American Samoa, French Polynesia, Cook Islands, Fiji, Kiribati, Tokelau, Tonga, and Tuvalu.

Nesting in this DPS of green turtle has been reported from 57 locations in the DPS' range (Figure 14; Seminoff et al. 2015). The genetic sampling and demographic studies of this population do not provide the data necessary to identify any subpopulations in this green turtle. Foraging areas are known to include green sea turtles from multiple nesting locations. For example, in Fiji, Piovano et al. (2019) analyzed genetic samples from 150 green sea turtles captured in two foraging habitats at Yadua and Makogai Islands, and identified 18 haplotypes with turtles in these habitats originating from American Samoa (72%), New Caledonia (17%) and French Polynesia (7%).

Central South Pacific green sea turtle nesting is geographically widespread at low levels. The most abundant nesting area is Scilly Atoll, French Polynesia, which in the early 1990s hosted 300–400 nesters annually (Balazs et al. 1995b). The current estimate is 1,050 breeding females (Seminoff et al. 2015). The most recent information is for American Samoa, with the majority of nesting at Rose Atoll and sporadic nesting on Tutuila (no recent reported nesting), Ofu and Swains Islands; sub-adult and adult turtles occur in low abundance in nearshore waters around Tutuila, Ofu, Olosega, Ta'u, and Swains islands (NMFS and FWS 1998a; Maison et al. 2010). Historically, 100–500 females nested annually at Canton Island, Kiribati (Balazs 1975).

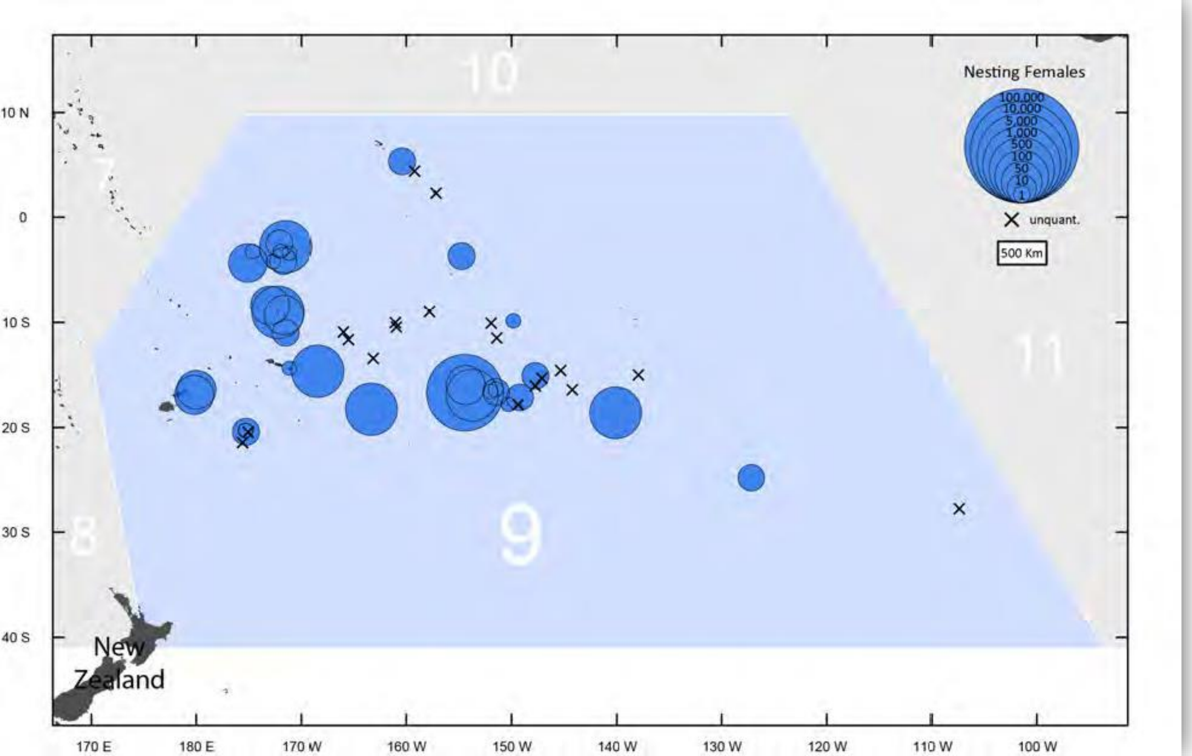

Figure 14. Nesting distribution of Central South Pacific green sea turtles. Size of circles indicates nesting estimated nester abundance. Locations marked with "X" identify nesting sites lacking abundance information.

Based on genetic samples from 14 green sea turtles that interacted with the DSLL fishery between 2005 and 2017, NMFS estimates that between 0 and 24% of the turtles caught in the fishery may have been Central South Pacific green sea turtles (P. Dutton pers. comm. June 29, 2018).

Status and Trends

The Central South Pacific green sea turtle is listed as endangered. Although population trends are not known, NMFS' Status Review Team concluded that this DPS had a 62% probability of having a greater than 1% risk of extinction over the next 100 years (Seminoff et al. 2015). The Status Review Team (SRT) estimated Central South Pacific green sea turtle nester abundance as 2,677 females (Seminoff et al. 2015) based on known nesting. They indicate that the number is likely higher due to the level of unmonitored nesting, but suggested fewer than 3,600 total nesters. We use the value of 2,677 as the minimum number of adult females, as the most reliable estimate based on data. While this is a conservative assumption, due to the lack of certainty with including unmonitored locations, we believe the value of 2,677 is the most reliable. The largest nesting site, Scilly Atoll, represents roughly one third of the DPS' nesting abundance and has reportedly significantly declined in the past 30 years as a result of commercial exploitation (Conservation International Pacific Islands Program 2013). However, this site has not been monitored since the early 1990s (Balazs et al. 1995b).

The Status Review Team concluded that chronic harvesting, the effect of coastal development, and the effect of phenomena related to climate-change (e.g., sea level rise) would be expected to place this DPS of green turtle at a relatively high risk of extinction in the foreseeable future (Seminoff et al. 2015).

We estimate total population size by using the age-based Leslie matrix model described above and the adult female population estimate of 2,677 (Seminoff et al. 2015). As the trend for this population is uncertain, we considered the mean of all 10,000 simulations to estimate a total mean population size of 631,745 (range 323,514 to 1,669,441) individuals greater than one year old.

Threats to the Species

Central South Pacific Ocean green turtles are threatened by being harvested for food and by the loss and degradation of coastal habitat. Hirth and Rohovit (1992) reports the exploitation of green turtles for eggs, meat, and parts has occurred throughout the South Pacific Region, including, but not limited to, American Samoa, Cook Islands, Fiji Islands, French Polynesia, and Kiribati. At Scilly Atoll (French Polynesia), local residents are allowed to harvest 50 adults per year from a nesting population that might be as low as 300 to 400 turtles. In Tuvalu, harvest of turtles for meat has been reported as a major threat (Alefaio et al. 2006). Elsewhere, subsistence harvests of eggs occur regularly.

In populated islands, such as Tutuila in American Samoa, human population growth and development has resulted in loss of habitat (Tuato'o-Bartley et al.1993; NMFS and FWS 1998). For example, nesting habitat has been completely removed to make way for structures such as seawalls (Saili 2005 as cited in Seminoff et al. 2015). Large areas of nesting beaches in Tonga and Tuvalu have been lost to coastal erosion (Bell et al. 2010). In addition, lighting associated with coastal development has disoriented hatchlings and resulted in them being killed by cars (A. Tagarino, American Samoa DMWR, pers. comm., 2013 reported in Seminoff et al. 2015).

In addition, Seminoff et al. (2015) reported that incidental capture of these green sea turtles in artisanal and commercial fisheries represented a major threat, particularly capture in longline, trap, or gillnet fisheries in American Samoa, Fiji, and the Cook Islands.

Many United States fisheries capture green sea turtles in the Pacific. However, only the Hawaii SSLL fishery is expected to interact with this DPS other than the DSLL within the Action Area. The effects from the SSLL will be discussed further in the Environmental Baseline. However, the SSLL is expected to capture 5 green sea turtles (all DPSs considered (n= 6), including the Central South Pacific) annually with one of those turtles expected to die (NMFS 2019a). Between 2004 and 2018, the SSLL fishery captured 10 green sea turtles and we expect less than 2% of those turtles caught could be from the Central South Pacific (NMFS 2019a). All ten green sea turtles caught in the Hawaii SSLL fishery between 2004 and 2018 were released alive with no gear attached.

Outside of the Action Area, only the following United States fisheries affect the Central West Pacific green sea turtle: the ASLL and the United States WCPO purse seine fishery. The ASLL fishery interacted with approximately 214 green turtles (with 205 estimated mortalities) between 20010 and 2019 (NMFS 2023a). Based on genetic samples from 31 green sea turtles, NMFS estimates that between 31% and 73% may have been Central South Pacific green sea turtles (P. Dutton pers. comm. July 5, 2018). This means that an estimated 66 (214 x 31/100) to 156 (214 x

73/100) Central South Pacific green sea turtles were captured by the ASLL fishery since 2010, with an estimated 63 (66 x 96/100) to 150 (156 x 96/100) of those interactions resulting in mortalities.

The United States WCPO purse seine fishery caught a total of 97 green sea turtles between 2008 and 2018. After proportioning the unidentified sea turtles and accounting for unobserved sets the estimated total number of interactions with green sea turtles from 2008-2018 is 173 (95% CI: [152, 200] (NMFS 2021a). Of those, 79% were considered by the observers to be alive and healthy at the time of release, 15% were released injured and 6% were released dying or dead. NMFS predicts that the purse seine fishery will interact with a mean of 7 (95th percentile: 11) green sea turtles per year with a maximum 5-year running average of 7.56 turtles and up to 76 turtles over 10 years. However, it is unknown what proportion of those are Central South Pacific green sea turtles (NMFS 2021a). We do not have genetic information from green sea turtles captured in the United States WCPO purse seine fishery to determine species composition although the following DPSs are expected: East Pacific green sea turtle, Central South Pacific green sea turtle, Southwest Pacific green sea turtle, Central West Pacific green sea turtle, and the East-Indian West Pacific green sea turtle (NMFS 2021a).

Historically, the Hawaii DSLL fishery has observed 25 green sea turtle interactions between 2004 and 2022 with an estimated 128 green sea turtle interactions during this time frame (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022b). NMFS estimates about 95% (95% CI = 77 to 99%) of green turtles captured in the DSLL fishery have been dead when they were hauled back to the fishing vessel based on the number of observed mortalities (this analysis).

2.2.7 Central West Pacific Green Sea Turtle

Distribution and Population Structure

The Central West Pacific green sea turtle has as their northern boundary 41°N latitude and is bounded by 41°N, 169°E in the northeast corner, going southeast to 9°N, 175°W, then southwest to 13°S, 171°E, west and slightly north to the eastern tip of Papua New Guinea, along the northern shore of the Island of New Guinea to West Papua in Indonesia, northwest to 4.5°N, 129°E then to West Papua in Indonesia, then north to 41°N, 146°E (Figure 15).

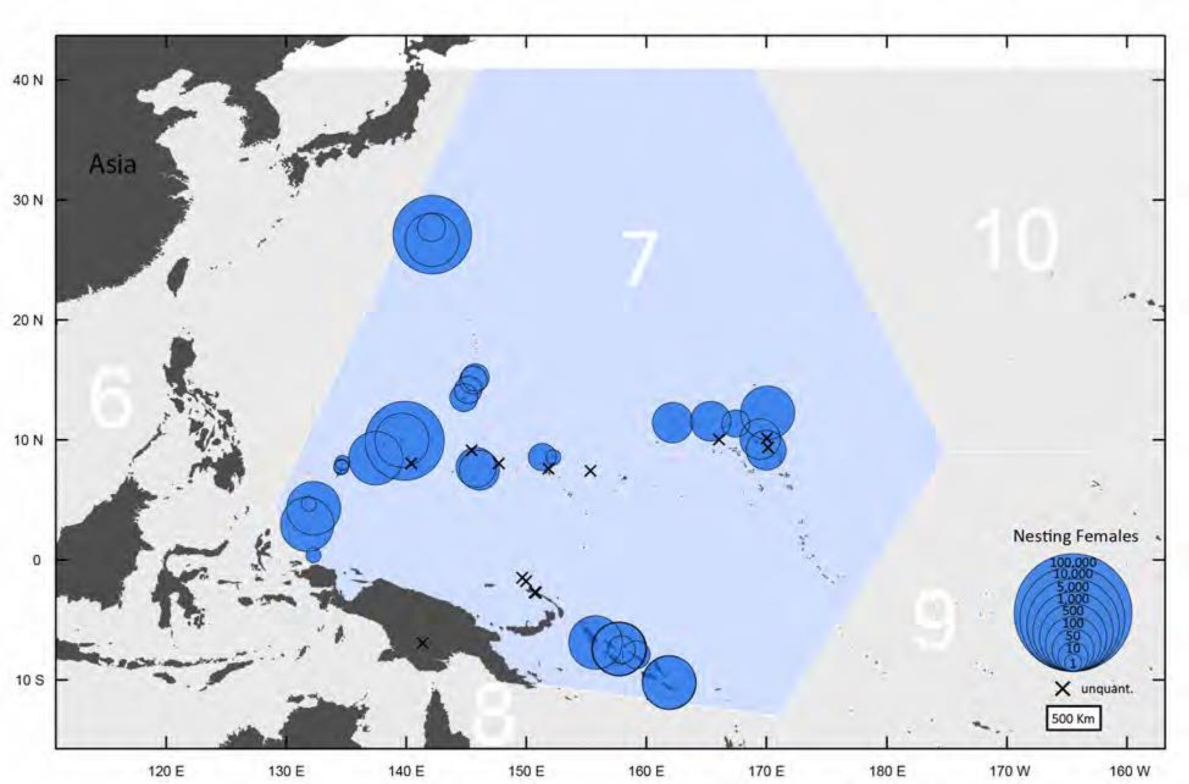

Figure 15. Nesting distribution of Central West Pacific green sea turtles (the blue-shaded area marked "76"). Size of circles indicates nesting estimated nester abundance.

The beaches with the highest abundance of nesting Central West Pacific females in this DPS are located in Gielop and Iar Island on Ulithi Atoll in Yap (abundance estimated at 1,412); Chichijima (1,301) and Hahajima (394), Ogasawara in Japan; Bikar Atoll in the Republic of the Marshall Islands (300); and Merir Island in Palau (441; NMFS and FWS 1998; Bureau of Marine Resources 2005; Barr 2006; Palau Bureau of Marine Resources 2008; Maison et al. 2010). Nesting also occurs on numerous other beaches in the Federated States of Micronesia, Solomon Islands, and Palau. A small number of green sea turtles also nest on Guam and the Commonwealth of the Northern Mariana Islands (Seminoff et al. 2015).

Central West Pacific green sea turtles travel throughout the western Pacific Ocean. Adult females from Palau have been reported from the Aru Islands in Indonesia – roughly 1,100 km away (Klain et al. 2007). Post-nesting adults from the Marshall Islands traveled to the Philippines, Kiribati, the Federated States of Micronesia, or remained in the Marshallese EEZ (Kabua et al. 2012). Turtles tagged in Yap have been recaptured in the Philippines, Marshall Islands, Papua New Guinea, Palau, and Yap (Palau BMR 2008; Cruce 2009). A turtle tagged in Yap in 1991 was recaptured in Muroto Kochi prefecture, Japan in 1999 (Miyawaki et al. 2000 as cited in Seminoff et al. 2015). A nesting female tagged in Palau was captured near the village of Yomitan Okinawa, Japan (Palau BMR 2008). Hundreds of nesting females tagged in Ogasawara Island were recaptured in the main islands of Japan, the Ryukyu Archipelago (Okinawa), Taiwan, China, and Philippines (H. Suganuma, Everlasting Nature of Asia, pers. comm., 2012;

Ogasawara Marine Station, Everlasting Nature of Asia. unpublished data as cited in Seminoff et al. 2015). A turtle tagged in Japan was reported to have nested in Yap (Cruce 2009).

The Central West Pacific green sea turtle is characterized by a small nesting population spread across a large geographic area, and is dominated by insular nesting. It is unclear how many populations comprise this DPS, but genetic analyses indicate that rookeries separated by more than 1,000 km were significantly different from each other, while rookeries within 500 km show no genetic differences (Seminoff et al. 2015). At a minimum the DPS appears to be comprised of at least seven populations based on mtDNA analyses by Dutton et al. (2014 as cited in Seminoff et al. 2015).

Based on genetic samples from 14 green sea turtles that interacted with the DSLL fishery between 2005 and 2017, NMFS estimates that between 0 and 18% of the turtles caught in the fishery may have been Central West Pacific green sea turtles (P. Dutton pers. comm. June 29, 2018).

Status and Trends

Central West Pacific green sea turtles are listed as endangered. The abundance of nesting females is relatively low with an estimated total of 6,518 females nesting at 51 documented nesting sites (Seminoff et al. 2015).

The limited available information suggests a nesting population decrease in some portions of the DPS like in the Marshall Islands, or unknown trends in other areas such as Palau, Papua New Guinea, the Marianas, Solomon Islands, or the Federated States of Micronesia (FSM)(Maison et al. 2010). There are a number of unquantified nesting sites, possibly with small numbers, although specifics regarding these sites are unknown. The largest nesting site is in the FSM, and that particular site hosts approximately 22% of the total annual nesting females (Seminoff et al. 2015). There is only one site for which 15 or more years of recent data are available for annual nester abundance (one of the standards for representing PVAs in Seminoff et al. 2015). This is at Chichijima, Japan, one of the major green sea turtle nesting concentrations in Japan (Horikoshi et al. 1994). The PVA has limitations and does not fully incorporate other key elements critical to the decision making process such as spatial structure or threats. It assumes all environmental and anthropogenic pressures will remain constant in the forecast period and it relies on nesting data alone. Chaloupka et al. (2008a) reports an estimated annual population growth rate of 6.8% per year for the Chichijima nesting site.

The PVA suggests the probability that this population will fall below the trend reference or absolute abundance reference in 100 years approaches zero (Seminoff et al. 2015). The population has increased from a mean of approximately 100 females/year in the late 1970s/early 1980s to a mean of approximately 500 per year since 2000. With the exception of the small Chichijima, Japan nesting aggregation, the overall population trend for the Central West Pacific green sea turtle is not known and given the threats to the DPS, its status as endangered under the ESA and suspected declines in certain portions we assume the DPS has an overall declining trend. We estimate total population size by using the age-based Leslie matrix model described above and the adult female population estimate of 6,518 (Seminoff et al. 2015). As the trend for this population is uncertain, we considered the mean of all 10,000 simulations to estimate a total mean population size of 1,543,625 (range 821,772 to 3,592,842) individuals greater than one year old.

Population Dynamics

The population dynamics of this DPS are similar to that of the other green turtles discussed in this opinion. However, clutch sizes appear to be more variable: clutch sizes range from 59 to 139 eggs per nest (compared with 80 to 120 eggs per nest in the general pattern; Palacios 2012a, 2012b). In addition, these green turtles appear to mature at smaller sizes. The known mean nester sizes range from 102 cm CCL in Palau to 104.5 cm curved carapace length (CCL) in Tinian, CNMI (Pultz et al. 1999). Finally, hatching success for areas where this information is available is more variable: from 44.1-73.8% (Suganuma 1985).

Threats to the Species

Predators such as ghost crabs, monitor lizards (*Varanus* sp.), wild pigs, rats, megapode birds, and iguanas have all been documented to consume large numbers of eggs in many of the nesting beaches throughout the DPS' range (Seminoff et al. 2015).

Natural environmental events, such as cyclones and hurricanes, may affect green turtles in the Central West Pacific green sea turtle. These storm events have also been shown to cause severe beach erosion and likely have negatively affected hatching success at many green turtle nesting beaches, especially in areas already prone to erosion. Shoreline erosion occurs naturally on many islands in the atolls of the Marshall Islands due to storms, sea level rise from the El Nino–Southern Oscillation, and currents (NMFS and FWS 1998a). Some erosion of nesting beaches at Oroluk was reported in 1990 after the passage of Typhoon Owen (NMFS and FWS 1998a).

Central West Pacific Ocean green turtles are threatened by being harvested for food, by the loss and degradation of coastal habitat, and by incidental capture in artisanal and commercial fisheries. Green turtle eggs are collected in the Commonwealth of the Northern Marianna Islands, Federated States of Micronesia, Guam, Kiribati, Papua, Papua New Guinea, Republic of the Marshall Islands, and Palau (Eckert 1993; Hitipeuw and Maturbongs 2002; Philip 2002; Humber et al. 2014). In addition, adult females reported to have been captured and killed for food on nesting beaches in the Commonwealth of the Northern Mariana Islands, Papua New Guinea (Limpus et al. 2002) and the Solomon Islands (Broderick 1998; Pita and Broderick 2005).

Human populations are growing rapidly in many areas of the insular Pacific and this expansion is exerting increased pressure on limited island resources. The most valuable land on most Pacific islands is often located along the coastline, particularly when it is associated with a sandy beach. Nesting habitat is being destroyed throughout the range of this DPS by coastal development and construction, placement of barriers to nesting, beachfront lighting, vehicular and pedestrian traffic, sand extraction, beach erosion, beach pollution, removal of native vegetation, and presence of non-native vegetation.

Central West Pacific Ocean green turtles are incidentally captured in artisanal and commercial fisheries throughout the region. They are captured in longline fisheries in the Federated States of Micronesia, Palau, Papua New Guinea, and the Solomon Islands. They have also been captured in purse-seine fisheries in the Republic of the Marshall Islands, Papua New Guinea, and the Solomon Islands, although all of the green turtles captured by purse seines in the Republic of the Marshall Island between 2010 and 2017 were released alive (WCPFC 2013, 2014, 2015, 2016, 2017, 2018b).

Many United States fisheries capture green sea turtles in the Pacific. However, only the Hawaii SSLL fishery is expected to interact with this DPS other than the DSLL within the Action Area. The effects from the SSLL fishery will be discussed further in the Environmental Baseline. However, the SSLL is expected to capture 5 green sea turtles annually (all DPSs considered (*n*= 6), including the Central West Pacific) with one of those turtles expected to die (NMFS 2019a). Between 2004 and 2018, the SSLL fishery captured 10 green sea turtles and we expect between 0 and 8% of those turtles caught could be from the Central West Pacific (NMFS 2019a). Therefore, the SSLL fishery may have captured as many as one Central West Pacific green sea turtle since 2004. In total, all ten green sea turtles caught in the Hawaii SSLL fishery between 2004 and 2018 were released alive with no gear attached, therefore mortality is not expected to occur from interactions in this fishery.

Outside of the Action Area, the following United States fisheries affect the Central West Pacific green sea turtle: the ASLL and the United States WCPO purse seine fishery. The ASLL fishery interacted with approximately 214 green turtles (with 205 estimated mortalities) between 2010 and 2019 (NMFS 2023a). Based on genetic samples from 31 green sea turtles, NMFS estimates that between 0 and 25% may have been Central West Pacific green sea turtles (P. Dutton pers. comm. July 5, 2018). This means that an estimated 0 to 53 Central West Pacific green sea turtles were captured by the ASLL fishery since 2010. Of those, we estimate 0 to 51 mortalities occurred from those interactions (NMFS 2023).

The United States WCPO purse seine fishery caught a total of 97 green sea turtles between 2008 and 2018. After proportioning the unidentified sea turtles and accounting for unobserved sets the estimated total number of interactions with green sea turtles from 2008-2018 is 173 (95% CI: [152, 200] (NMFS 2021a). Of those, 79% were considered by the observers to be alive and healthy at the time of release, 15% were released injured and 6% were released dying or dead. NMFS predicts that the purse seine fishery will interact with a mean of 7 (95th percentile: 11) green sea turtles per year with a maximum 5-year running average of 7.56 turtles and up to 76 turtles over 10 years. However, it is unknown what proportion of those are Central West Pacific green sea turtles (NMFS 2021a). We do not have genetic information from green sea turtles captured in the United States WCPO purse seine fishery to determine species composition although the following DPSs are expected: East Pacific green sea turtle, Central South Pacific green sea turtle, Southwest Pacific green sea turtle, Central West Pacific green sea turtle, and the East-Indian West Pacific green sea turtle (NMFS 2021a).

Historically, the Hawaii DSLL fishery has observed 25 green sea turtle interactions between 2004 and 2022 with an estimated 128 green sea turtle interactions during this time frame (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a). NMFS estimates about 95% (95% CI = 77 to 99%) of green turtles captured in the DSLL fishery have been dead when they were hauled back to the fishing vessel based on the number of observed mortalities (this analysis).

2.2.8 East Indian-West Pacific Green Sea Turtle

Distribution and Population Structure

The western boundary for the East Indian–West Pacific green sea turtle is 84°E longitude from 40°S to where it coincides with India near Odisha, northeast and into the West Pacific Ocean to include Taiwan extending east at 41°N to 146°E longitude, south west to 4.5°N, 129°E, then south and east to West Papua in Indonesia (at 135°E) and the Torres Straits in Australia (at 142°E longitude). The southern boundary is 40°S latitude, encompassing the Gulf of Carpentaria (Figure 16).

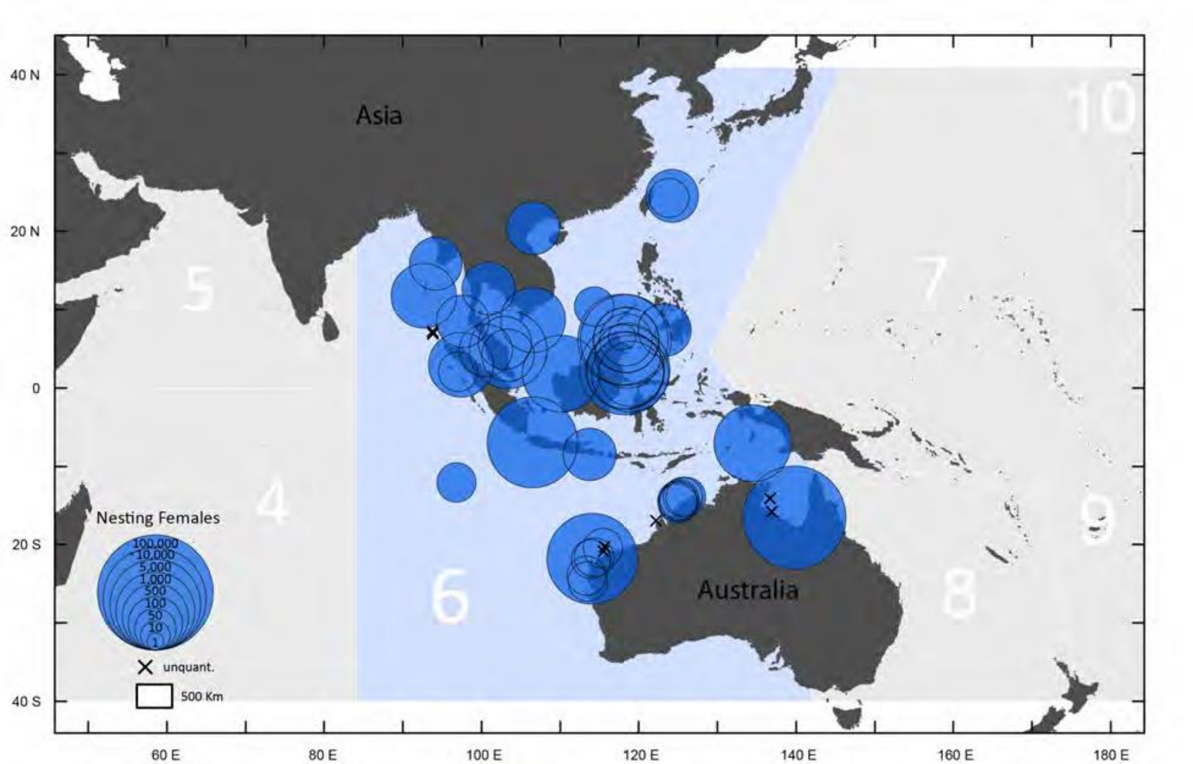

Figure 16. Nesting distribution of East Indian-West Pacific green sea turtles (the blue-shaded area marked “11”). Size of circles indicates nesting estimated nester abundance.

Genetic data suggest that the DPS is comprised of at least 16 populations:: Northwest Shelf, Scott Reef, Ashmore Reef, and the Gulf of Carpentaria (Australia); West Java, Berau Islands, and Aru (Indonesia); Peninsular Malaysia, Sarawak, Southeast Sabah (Malaysia), Sulu Sea (Malaysia/Philippines); Wan-an Island, and Lanyu Island (Taiwan); Zamami, Iriomote Island, and Ishigaki Island, Japan (Dethmers et al. 2006; Cheng et al. 2008; Hamabata et al. 2009; Nishizawa et al. 2011).

Based on genetic samples from 11 green sea turtles that interacted with the DSLL fishery between 2005 and 2017, NMFS estimates that between 0 and 28% of the turtles caught in the fishery may have been East Indian-West Pacific green sea turtles (P. Dutton pers. comm. June 29, 2018).

Status and Trends

East Indian-West Pacific green sea turtles are listed as threatened. The abundance of nesting females in this DPS is estimated at 77,009 females over 50 nesting sites (Seminoff et al. 2015). The largest nesting site is in northern Australia and supports about 25,000 nesting females however there are no recent data to support estimations of trend (Seminoff et al. 2015). The abundance of adult females at important nesting sites in Malaysia is estimated at 7,000 females and is expected to increase in the foreseeable future. However, the abundance of nesting females at several smaller sites are expected to decrease (Seminoff et al. 2015). Because of these declines, several nesting populations of this DPS have relatively high probabilities of falling below 100 nesting females per year (Seminoff et al. 2015).

Mazaris et al. (2017) conducted a meta-analysis of population trends on sea turtles nesting beaches and they summarize trends for 14 East Indian-West Pacific green sea turtle nesting beaches. Restricting these trends to those that concluded within the last 20 years (i.e. 2002) and consisting of at least 10 years of monitoring data, we estimate a mean population decline of 1.11% per year (95% CI: -1.6% to -0.6%).

We estimate total population size by using the age-based Leslie matrix model described above and the adult female population estimate of 77,009 (Seminoff et al. 2015). As the trend for this population is uncertain, we considered the mean of all 10,000 simulations to estimate a total mean population size of 18,171,565 (range 9,227,181 to 43,725,766) individuals greater than one year old.

Threats to the Species

East Indian-West Pacific Ocean green turtles are threatened by being harvested for food, by the loss and degradation of coastal habitat, and by incidental capture in artisanal and commercial fisheries. Harvesting turtle eggs is legal in several countries within the range of this DPS and continues to occur where it has been banned (Islam 2001; Seminoff et al. 2015).

Seminoff et al. (2015) identified coastal development, beachfront lighting, erosion resulting from sand mining, and sea level rise as a result of climate change, as major threats to this DPS. Driving on beaches is a threat in some areas, such as Australia. The extent of fishing practices, depleted seagrass beds, and marine pollution is broad with high levels occurring in waters where high numbers of green turtles are known to forage and migrate. All life stages of green turtles are affected by habitat destruction in the neritic/oceanic zone.

Many United States fisheries capture green sea turtles in the Pacific. However, only the Hawaii SSLL fishery is expected to interact with this DPS other than the DSLL within the Action Area. The effects from the SSLL fishery will be discussed further in the Environmental Baseline. However, the SSLL is expected to capture 5 green sea turtles annually (all species considered (n= 6), including the East Indian-West Pacific green sea turtle) with one of those turtles expected to die (NMFS 2019a). Between 2004 and 2018, the SSLL fishery captured 10 green sea turtles and we expect between 0 and 10% of those turtles caught could be from the East Indian-West Pacific green sea turtle DPS (NMFS 2019a). Therefore, the SSLL fishery may have captured one East Indian-West Pacific green sea turtle since 2004. In total, all ten green sea turtles caught in the Hawaii SSLL fishery between 2004 and 2018 were released alive with no gear attached, therefore mortality is not expected to occur from interactions in this fishery.

Outside of the Action Area, the following United States fisheries affect the East Indian-West Pacific green sea turtle: the ASLL and the United States WCPO purse seine fishery. The ASLL fishery interacted with approximately 214 green turtles (with 205 estimated mortalities) between 2010 and 2019 (NMFS 2023a). Based on genetic samples from 31 green sea turtles, NMFS estimates that between 0 and 11% may have been East Indian-West Pacific green sea turtles (P. Dutton pers. comm. July 5, 2018). This means that an estimated 0 to 24 East Indian-West Pacific green sea turtles were captured by the ASLL fishery since 2010. Of those, we estimate 0 to 23 mortalities occurred from those interactions.

The United States WCPO purse seine fishery caught a total of 97 green sea turtles between 2008 and 2018. After proportioning the unidentified sea turtles and accounting for unobserved sets the estimated total number of interactions with green sea turtles from 2008-2018 is 173 (95% CI: [152, 200] (NMFS 2021a). Of those, 79% were considered by the observers to be alive and healthy at the time of release, 15% were released injured and 6% were released dying or dead. NMFS predicts that the purse seine fishery will interact with a mean of 7 (95th percentile: 11) green sea turtles per year with a maximum 5-year running average of 7.56 turtles and up to 76 turtles over 10 years, However, it is unknown what proportion of those are East Indian-West Pacific Ocean green sea turtles (NMFS 2021a). We do not have genetic information from green sea turtles captured in the United States WCPO purse seine fishery to determine species composition although the following DPS are expected: East Pacific green sea turtle, Central South Pacific green sea turtle, Southwest Pacific green sea turtle, Central West Pacific green sea turtle, and the East Indian-West Pacific green sea turtle (NMFS 2021a).

Historically, the Hawaii DSLL fishery has observed 24 green sea turtle interactions between 2004 and 2021 with an estimated 123 green sea turtle interactions during this time frame (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a). NMFS estimates about 95% (95% CI = 77 to 99%) of green turtles captured in the DSLL fishery have been dead when they were hauled back to the fishing vessel based on the number of observed mortalities (this analysis).

2.2.9 East Pacific Green Sea Turtle

Distribution and Population Structure

The geographic range of East Pacific green sea turtles extends from the California/Oregon border (USA; 42°N) southward along the Pacific coast of the Americas to central Chile (40°S). The DPS ranges from the aforementioned locations in the United States and Chile to 143°W and 96°W, respectively (Figure 17). This DPS encompasses the Revillagigedos Archipelago (Mexico) and Galapagos Islands (Ecuador).

The two largest nesting aggregations are found in Michoacan, Mexico and in the Galapagos Islands, Ecuador (Zarate et al. 2003; Delgado-Trejo and Alvarado-Diaz 2012). Secondary nesting areas are found throughout the Pacific Coast of Costa Rica and Clarion and Socorro Islands in the Revillagigedos Archipelago, Mexico. Low-level nesting occurs in Colombia, Ecuador, Guatemala, and Peru. Scattered nesting also occurs from Mexico's Baja California Peninsula (G. Tiburcios-Pintos Minicipio de Los Cabos, pers. comm. 2012) to Peru (S. Kelez, Oceanica, pers. comm. 2012).

Based on genetic data, this DPS consists of at least five populations: two in Mexico, one in Costa Rica, one in the eastern Pacific, and one in the Galapagos Islands (Dutton unpublished data as cited in Seminoff et al. 2015). Those populations are represented by at least 39 nesting sites, with most of these sites concentrated in Mexico, Ecuador, and Costa Rica (Seminoff et al. 2015). Male-mediated gene flow between the Galapagos and Mexico populations appears to be limited (Roden et al. 2013 as cited in Seminoff et al. 2015).

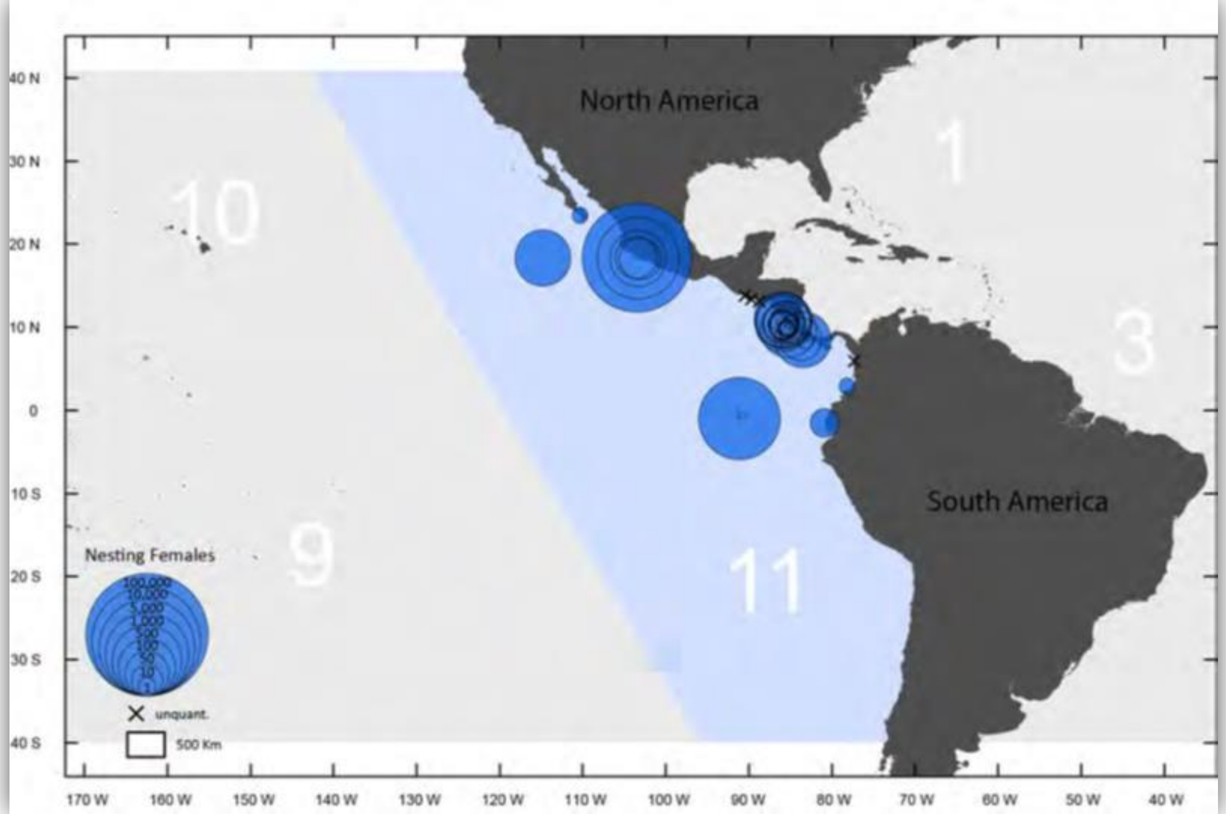

Figure 17. Nesting distribution of East Pacific green sea turtles (blue-shaded area marked with '11'). Size of circles indicates estimated nester abundance.

Based on genetic samples from 14 green sea turtles that interacted with the DSLL fishery between 2005 and 2017, NMFS estimates that between 31 and 80% of the turtles caught in the fishery may have been East Pacific green sea turtles (P. Dutton pers. comm. June 29, 2018).

Status and Trends

The East Pacific green sea turtle is listed as threatened. Seminoff et al. (2015) ranked the DPS as having a low risk of extinction based on the abundance of nesting females. The abundance of nesting females in Mexico was the largest with 13,664 nesters among seven nesting sites, followed by Ecuador with 3,603 females in the Galapagos and 15 on mainland, and Costa Rica with 2,826 females distributed among 26 nesting sites (see Table 15.1 in Seminoff et al. 2015).

Although trend information is lacking for the majority of nesting beaches, based on 25-year trend for the nesting aggregation at Colola, Mexico (which is estimated to comprise about 58% of the total adult females for the DPS), the abundance of East Pacific green turtle nesting females appears to have increased since the population's low point in the mid-1980s. Based on nesting beach data, the current adult female nester population for Colola, Michoacan is 11,588 females, which makes this the largest nesting aggregation of threatened East Pacific green sea turtle. The total for the entire East Pacific green sea turtle is estimated at 20,062 nesting females (Seminoff et al. 2015). This observed increase may have resulted from the onset of nesting beach protection in 1979, as is suggested by the similarity in timing between the onset of beach conservation and the age-to-maturity for green turtles in Pacific Mexico. Similarly, data from the Galapagos Archipelago suggest that the abundance of nesting females in that population may be increasing.

We estimate total population size by using the age-based Leslie matrix model described above and the adult female population estimate of 20,062 (Seminoff et al. 2015). As we are reasonably certain the trend for this population is increasing, we restricted the 10,000 population simulations to those demonstrating an increasing population growth rate to estimate a total mean population size of 3,580,6207 (range 2,473,546 to 5,814,512) individuals greater than one year old.

Population Dynamics

The population dynamics of this DPS are similar to that of the other green turtles, but the adult pattern begins earlier because East Pacific green turtles mature at smaller sizes (60–77 cm SCL; Seminoff et al. 2002b). Size of nesters ranges from 101.7 cm CCL (Campeche, Mexico) to 109.3 cm CCL (Isla Aguada, Mexico (Guzman-Hernandez 2001, 2006 as cited in Seminoff et al. 2015). Their age at first reproduction is younger than that of most other green turtles: it has been reported as 12 to 26 years at Tortuguero, Costal Rica (Frazer and Ladner 1986) and 12 to 20 years with an average of 16 years at Quintana Roo, Mexico (Richards et al. 2011). Seminoff et al. (2002a) reports that East Pacific green turtles reach sexual maturity between 9 and 21 years in northwest Mexico.

Remigration intervals, numbers of nests per season, and number of eggs per clutch are within the ranges reported in the General Population Dynamics narrative for green turtles (subsection 4.2.1.1). Hatching success rate was 92% in Mexico (Xavier et al. 2006 as cited in Seminoff et al. 2015), although this is probably an overestimate because the study did not include nests that show no sign of hatching in these analyses.

Survivorship tends to be lower for juveniles and subadults (0.58) than for adult green turtles (0.97) in northwest Mexico (Seminoff et al. 2003). A study in a northern foraging area in San Diego Bay indicated an average annual survival rate of 0.86 (95% CI=0.36– 0.99), which included a wide range of age groups (Eguchi et al. 2012).

Threats to the Species

East Pacific green turtles are exposed to a variety of natural threats that include hurricanes, tropical storms, storm surges, sand accretion, and rainfall associated with storms. Predation occurs at low levels in the East Pacific. In the Galapagos Islands there is depredation on eggs and hatchlings by feral pigs (*Sus* spp.) and beetles (order Coleoptera) (Zarate et al. 2013). There are also accounts of jaguars (*Panthera onca*) killing adult female green turtles (L. Fonseca, National University of Costa Rica, unpubl. data 2009 as cited in Seminoff et al. 2015) at beaches in Costa Rica.

The largest threat on East Pacific green sea turtle nesting beaches is reduced availability of habitat due to heavy armament and subsequent erosion. In addition, while nesting beaches in Costa Rica, Revillagigedos Islands, and the Galapagos Islands are less affected by coastal development than green turtle nesting beaches in other regions around the Pacific, several of the secondary green sea turtle nesting beaches in Mexico suffer from coastal development. For example, effects of coastal development are especially acute at Maruata, a site with heavy tourist activity and foot traffic during the nesting season (Seminoff 1994). Nest destruction due to human presence is also a threat to nesting beaches in the Galapagos Islands (Zarate et al. 2006). However, such threats vary by site.

In some countries and localities within the range of the East Pacific green sea turtle, harvest of turtle eggs is legal, while in others it is illegal but persistent due to lack of enforcement. The impact of egg harvest is exacerbated by the high monetary value of eggs, consistent market demand, and severe poverty in many of the countries in the eastern Pacific Region. Egg harvest is a major conservation challenge at several sites in Costa Rica, including Nombre de Jesus and Zapotillal Beaches, where 90% of the eggs were taken by egg collectors during one particular study (Blanco 2010). Egg harvest is also expected to occur at unprotected nesting sites in Mexico, Guatemala, El Salvador, and Nicaragua (NMFS and FWS 2007a). Mancini and Koch (2009) describe a black market that killed tens of thousands of green turtles each year in the eastern Pacific Region. Sea turtles were, and continue to be, harvested primarily for their meat, although other products have served important non-food uses. Sea turtle oil was used as a cold remedy and the meat, eggs and other products have been highly valued for their presumed aphrodisiacal qualities.

Effects of climate change include, among other things, sea surface temperature increases, the alteration of thermal sand characteristics of beaches (from warming temperatures), which could result in the reduction or cessation of male hatchling production (Hawkes et al. 2009; Poloczanska et al. 2009), and a significant rise in sea level, which could significantly restrict green turtle nesting habitat. While sea turtles have survived past eras that have included significant temperature fluctuations, future climate change is expected to happen at unprecedented rates, and if sea turtles cannot adapt quickly, they may face local to widespread extirpations (Hawkes et al. 2009). Impacts from global climate change induced by human activities are likely to become more apparent in future years (IPCC 2018, 2022). However, at the primary nesting beach in Michoacán, Mexico (Colola), the beach slope aspect is extremely steep and the dune surface at which the vast majority of nests are laid is well-elevated. This site is likely buffered against short-term sea level rise resulting from climate change. In addition, many nesting sites are along protected beach faces, out of tidal surge pathways. For example, multiple nesting sites in Costa Rica and in the Galapagos Islands are on beaches that are protected from major swells.

Incidental capture in artisanal and commercial fisheries is a significant threat to the survival of green turtles throughout the eastern Pacific Ocean. The primary gear types involved in these interactions include longlines, drift nets, set nets, and trawl fisheries. These are employed by both artisanal and industrial fleets, and target a wide variety of species including tunas, sharks, sardines, swordfish, and mahi mahi. In the eastern Pacific Ocean, particularly areas in the southern portion of the range of this DPS, significant bycatch has been reported in artisanal gill net and longline shark and mahi mahi fisheries operating out of Peru (Kelez et al. 2003; Alfaro-Shigueto et al. 2006) and, to a lesser extent, Chile (Donoso and Dutton 2010).

The fishing industry in Peru is the second largest economic activity in the country. Over the last few decades, evidence of marine vertebrate bycatch has been collected for a range of industrial fisheries. It has recently been acknowledged that large impacts may also result from similar interactions with small-scale fisheries (SSF) due largely to their diffuse effort and large number of vessels in operation. From 2000 to 2007, shore-based and onboard observer programs from three SSF ports in Peru were used to assess the impact on marine turtles of small-scale longline, bottom set nets and driftnet fisheries (Alfaro- Shigueto et al. 2011). From this study, a total of 807 sea turtles were captured, 91.8% of which were released alive. It was estimated that 5,900 turtles were captured annually (3,200 loggerhead turtles, 2,400 green turtles, 240 olive ridleys and 70 leatherback turtles). SSFs in Peru are widespread and numerous (>100 ports, >9,500 vessels, >37,000 fishermen), and the observed effort in this study constituted 1% of longline and net deployments. Alfaro-Shigueto et al. (2011) suggest that the number of turtles captured per year is likely to be in the tens of thousands. Thus, the impacts of Peruvian SSF have the potential to severely impact sea turtles in the Pacific especially green, loggerhead, and leatherback turtles.

In Baja California Sur, Mexico, from 2006–2009 small-scale gill-net fisheries caused massive green turtle mortality at Laguna San Ignacio, where Mancini et al. (2012) estimated that over 1,000 turtles were killed each year in nets set for guitarfish. Bycatch in coastal areas occurs principally in shrimp trawlers, gill nets and bottom longlines (Orrego and Arauz 2004). However, since 1996, all countries from Mexico to Ecuador declared the use of turtle excluder devices (TEDs) as mandatory for all industrial fleets to meet the requirements to export shrimp to the United States under the United States Magnuson-Stevens Fishery Conservation and Management Act (Helvey and Fahy 2012 as cited in Seminoff et al. 2015). Since then, bycatch has not been thoroughly evaluated but it is widely expected that most fishermen either improperly implement TEDs or remove them entirely from their trawls.

Many United States fisheries capture green sea turtles in the Pacific. However, two United States fisheries- the Hawaii SSLL and the West Coast DSLL fishery are expected to interact with this DPS other than the DSLL within the Action Area. Therefore, the effects from these fisheries will be discussed further in the Environmental Baseline. However, the SSLL is expected to capture 5 green sea turtles (all species considered (*n*= 6), including the East Pacific green sea turtle) annually with one of those turtles expected to die (NMFS 2019a). Between 2004 and 2018, the SSLL fishery captured 10 green sea turtles and we expect between 0 and 10% of those turtles caught could be from the East Pacific green sea turtle DPS (NMFS 2019a). Therefore, the SSLL fishery may have captured one East Pacific green sea turtle since 2004. In total, all ten green sea turtles caught in the Hawaii SSLL fishery between 2004 and 2018 were released alive with no gear attached, therefore mortality is not expected to occur from interactions in this fishery. The West Coast DSLL is estimated to capture and kill one adult or sub-adult green sea turtle every ten years (NMFS 2016b). However, it is unclear whether the interaction would be attributed to the East Pacific or Central North Pacific green sea turtle. Therefore, NMFS assessed effects to both DPSs of green sea turtles (see NMFS 2016b).

Outside of the Action Area, the following United States fisheries affect the East Pacific green sea turtle: the ASLL, the United States WCPO purse seine fishery, the Eastern Tropical Pacific purse seine fishery, and the California/Oregon drift gillnet fishery. The ASLL fishery interacted with approximately 214 green turtles (with 205estimated mortalities) between 2010 and 2019 (NMFS 2023a). Based on genetic samples from 31 green sea turtles, NMFS estimates that between 2 and 23% may have been East Pacific green sea turtles (P. Dutton pers. comm. July 5, 2018). This means that an estimated 4 to 50 East Pacific green sea turtles have been captured by the ASLL fishery since 2010. Of those, we estimate 4 to 48 mortalities occurred from those interactions (NMFS 2023a).

The United States WCPO purse seine fishery caught a total of 97 green sea turtles between 2008 and 2018. After proportioning the unidentified sea turtles and accounting for unobserved sets the estimated total number of interactions with green sea turtles from 2008-2018 is 173 (95% CI: [152, 200] (NMFS 2021a). Of those, 79% were considered by the observers to be alive and healthy at the time of release, 15% were released injured and 6% were released dying or dead. NMFS predicts that the purse seine fishery will interact with a mean of 7 (95th percentile: 11) green sea turtles per year with a maximum 5-year running average of 7.56 turtles and up to 76 turtles over 10 years. However, it is unknown what proportion of those are East Pacific green sea turtles (NMFS 2021a). We do not have genetic information from green sea turtles captured in the United States WCPO purse seine fishery to determine DPS composition although the following DPS are expected: East Pacific green sea turtle, Central South Pacific green sea turtle, Southwest Pacific green sea turtle, Central West Pacific green sea turtle, and the East Indian-West Pacific green sea turtle (NMFS 2021a).

The Eastern Tropical Pacific purse seine fishery is estimated to interact with 350 East Pacific green sea turtles in a 10-year period, with an associated mortality of 20 of those individuals (NMFS 1999). In the California/Oregon drift gillnet fishery, NMFS exempted take for 1 interaction and mortality over a 1-year period, or 2 interactions with 1 mortality over 5 years (NMFS 2013).

Historically, the Hawaii DSLL fishery has observed 25 green sea turtle interactions between 2004 and 2022 with an estimated 128 green sea turtle interactions during this time frame (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a). NMFS estimates about 95% (95% CI = 77 to 99%) of green turtles captured in the DSLL fishery have been dead when they were hauled back to the fishing vessel based on the number of observed mortalities (this analysis).

NMFS exempted the prohibition on take for an additional 18 East Pacific green sea turtles from non-fishery related actions in the West Coast and southwest regions through the section 7 consultation process. Only three mortalities were estimated to occur from these interactions (see NMFS 2006). Lastly, NMFS exempted the prohibition on take for 100 interactions for a 5.5 year period starting April 3, 2019 for the Seal Beach Naval Weapons Station Pier Construction Project (NMFS 2019b). No mortalities were expected to occur as a result of these interactions (NMFS 2019b).

2.2.10 Southwest Pacific Green Sea Turtle

Distribution and Population Structure

The distribution of Southwest Pacific green sea turtles extends from the western boundary of Torres Strait (at 142°E longitude), southeast to the eastern tip of Papua New Guinea and out to the offshore coordinate of 13°S, 171°E; the eastern boundary runs from this point southeast to 40°S, 176°E; the southern boundary runs along 40°S from 142°E to 176°E; and the western boundary runs from 40°S, 142ºE north to Australian coast then follows the coast northward to Torres Strait (Figure 18).

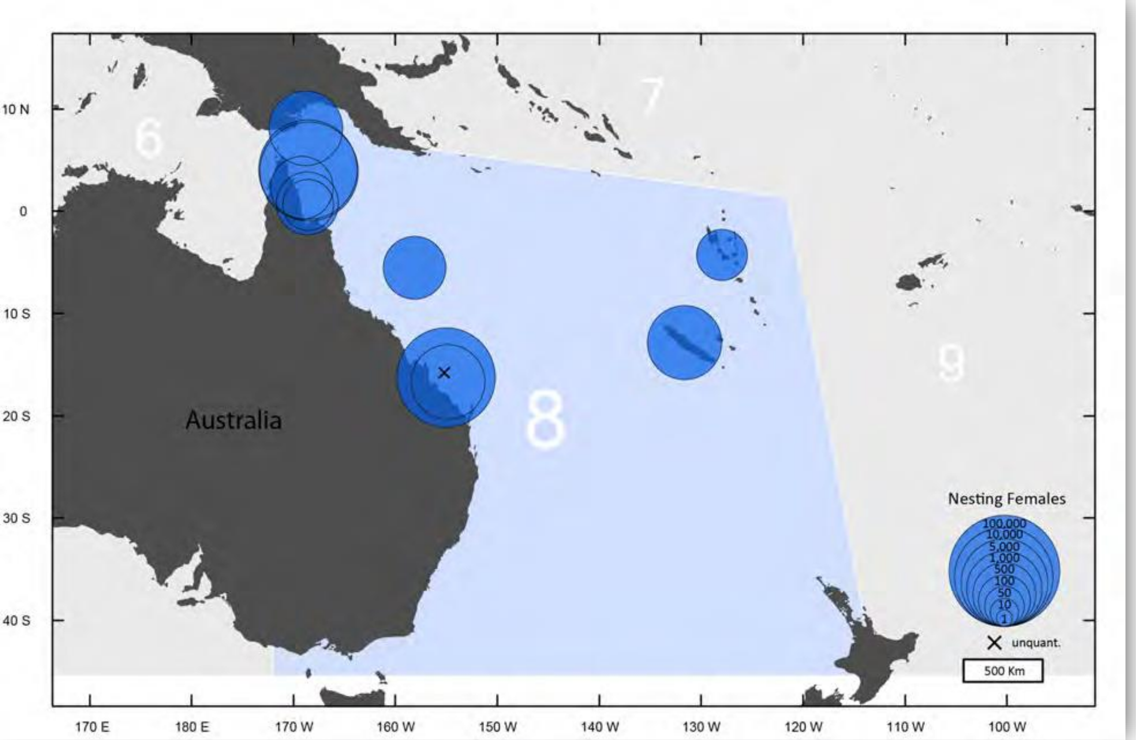

Figure 18. Nesting distribution of Southwest Pacific green sea turtles. Size of circles indicates nesting estimated nester abundance. Locations marked with '•' indicate nesting sites lacking abundance information.

Green turtle nesting is widely dispersed throughout the Southwest Pacific Ocean. Genetic data suggest the Southwest Pacific green sea turtle is comprised of at least four populations that appear to be demographically discrete: Northern Great Barrier Reef, Southern Great Barrier Reef, Coral Sea and New Caledonia (Dethmers et al. 2006; Jensen 2010; Dutton unpublished data as cited in Seminoff et al. 2015). These four populations encompass more than 100 nesting beaches; however, those beaches have historically been grouped into a smaller number of nesting aggregations (Limpus 2009). Seminoff et al. (2015) recognized 12 total nesting aggregations. Most of the nesting occur in Australia's Great Barrier Reef World Heritage Area and eastern Torres Strait. Nesting also occurs on the Coral Sea Islands, New Caledonia, and Vanuatu.

Based on genetic samples from 14 green sea turtles that interacted with the DSLL fishery between 2005 and 2017, NMFS estimates that between 0 and 25% of the turtles caught in the fishery may have been Southwest Pacific green sea turtles (P. Dutton pers. comm. June 29, 2018).

Status and Trends

Southwest Pacific green sea turtles are listed as threatened. The estimated abundance of nesting females in this population is high with 83,058 adult females nesting (Seminoff et al. 2015). The highest nesting densities for this green turtle occur in Australia, particularly in the northern Great Barrier Reef. Estimates of annual nesters at Raine Island, Australia, vary from 4,000 – 89,000

(Seminoff et al. 2004; NMFS and FWS 2007a; Chaloupka et al. 2008a; Limpus 2009). The number of turtles nesting in the Great Barrier Reef area of Australia differs widely from year to year and is well correlated with an index of the Southern Oscillation (Limpus and Nicholls 2000 as cited in Seminoff et al. 2015). Fewer turtles nest in New Caledonia and Vanuatu. In New Caledonia, Pritchard (1994 cited in Maison et al. 2010) described turtles to be abundant or near saturation levels on several islands in the archipelago. A 2006 and 2007 survey of over 6,000 km of nesting habitat identified nesting locations hosting an estimated 1,000 – 2,000 green turtle females nesting annually (Maison et al. 2010 citing Limpus et al. 2009). In Vanuatu, hundreds of nesting green turtles have been observed on Malekula Island, Southern Epi Island, Santo and Thion Islands, Tegua and Hiu Islands (Maison et al. 2010).

We estimate total population size by using the age-based Leslie matrix model described above and the adult female population estimate of 83,058 (Seminoff et al. 2015). As we are reasonably certain the trend for this population is increasing, we restricted the 10,000 population simulations to those demonstrating an increasing population growth rate to estimate a total mean population size of 14,762,154 (range 10,137,513 to 23,139,940) individuals greater than one year old.

Population Dynamics

The population dynamics of this DPS are similar to that of the other green turtles, but Southwest Pacific green sea turtles may produce higher proportions of females or females in this DPS may have survival rates that are disproportionately higher than their male counterparts. Jensen et al. (2018) collected data that linked the sex of male and female green turtles foraging in the Great Barrier Reef to their nesting beaches. These data show a moderate female sex bias (65%–69% female) in turtles originating from nesting beaches in the southern Great Barrier Reef where the temperatures of nesting beaches are cooler while turtles originating from warmer northern Great Barrier Reef nesting beaches were extremely female-biased (99.1% of juvenile, 99.8% of subadult, and 86.8% of adult-sized turtles; Jensen et al. 2018). These results suggest that the northern Great Barrier Reef green turtle rookeries have been producing mostly females for more than two decades and that the complete feminization of this DPS is possible in the foreseeable future. If this occurs, it would have severe implications for the future of this DPS.

These green turtles also appear to have higher annual survival rates when compared with other green turtles. Annual survival was 88% for juveniles, 85% for subadults, and 95% for adults (Chaloupka and Limpus 2005). These adult survival rates are high for green turtles and may be biased by the long-term decline in the size of adult nesting females and an increase in remigration interval (Limpus 2009), although other explanations are possible.

Threats to the Species

Sea level rise represents one of the primary contemporary threats to Southwest Pacific green sea turtles. Fuentes et al. (2010b), estimated that up to 38% of suitable green turtle nesting across the northern Great Barrier Reef may be inundated by sea level rise. This will exacerbate density-dependent pressures on these nesting beaches, which includes nest destruction by other turtles and which already occurs on several of these beaches (Fuentes et al. 2010b).

These green turtles are also vulnerable to harvest throughout Australia and neighboring countries such as New Caledonia, Fiji, Vanuatu, Papua New Guinea, and Indonesia (Limpus 2009). Cumulative annual harvest of green sea turtles that nest in Australia may be in the tens of thousands, and it appears likely that historic indigenous peoples harvests may have been in the same order of magnitude (Limpus 2009). Annual harvest in the southern East Coast is estimated to be 500–1000, mostly large females (Limpus 2009). Estimated annual harvest is even larger for the northeast part of the country, primarily Torres Strait which may have a large harvest of turtles (4,000) and eggs (unquantified). Annual indigenous harvest from northern and Western Australia may be several thousand turtles per year plus non-permitted egg harvest (Limpus 2009). Harvest from neighboring countries (New Caledonia, Fiji, Vanuatu, Papua New Guinea, Indonesia) may be as high as several thousand juvenile or adults per year and may threaten the sustainability of the Australian green turtle stocks (Limpus 2009).

Many United States fisheries capture green sea turtles in the Pacific. However, only the Hawaii SSLL fishery is expected to interact with this DPS other than the DSLL within the Action Area. The effects from the SSLL fishery will be discussed further in the Environmental Baseline. However, the SSLL is expected to capture 5 green sea turtles (all species considered (*n*= 6), including the Southwest Pacific green sea turtle) annually with one of those turtles expected to die (NMFS 2019a). Between 2004 and 2018, the SSLL fishery captured 10 green sea turtles and we expect between 0 and 8% of those turtles caught could be from the Southwest Pacific green sea turtle DPS (NMFS 2019a). Meaning that the SSLL fishery may have captured one Southwest Pacific green sea turtle since 2004. In total, all ten green sea turtles caught in the Hawaii SSLL fishery between 2004 and 2018 were released alive with no gear attached, therefore mortality is not expected to occur from interactions in this fishery.

Outside of the Action Area, the following United States fisheries affect the Southwest Pacific green sea turtle: the ASLL and the United States WCPO purse seine fishery. The ASLL fishery interacted with approximately 214 green turtles (with 205 estimated mortalities) between 2010 and 2018 (NMFS 2023a). Based on genetic samples from 31 green sea turtles, NMFS estimates that between 12% and 47% may have been Southwest Pacific green sea turtles (P. Dutton pers. comm. July 5, 2018). This means that an estimated 25 to 100 Southwest Pacific green sea turtles were captured by the ASLL fishery since 2010, with 24 to 96 mortalities per year.

The United States WCPO purse seine fishery caught a total of 97 green sea turtles between 2008 and 2018. After proportioning the unidentified sea turtles and accounting for unobserved sets the estimated total number of interactions with green sea turtles from 2008-2018 is 173 (95% CI: [152, 200] (NMFS 2021a). Of those, 79% were considered by the observers to be alive and healthy at the time of release, 15% were released injured and 6% were released dying or dead. NMFS predicts that the purse seine fishery will interact with a mean of 7 (95th percentile: 11) green sea turtles per year with a maximum 5-year running average of 7.56 turtles and up to 76 turtles over 10 years however it is unknown what proportion of those are East Indian-West Pacific Ocean green sea turtles (NMFS 2021a). We do not have genetic information from green sea turtles captured in the United States WCPO purse seine fishery to determine species composition although the following DPS are expected: East Pacific green sea turtle, Central South Pacific green sea turtle, Southwest Pacific green sea turtle, Central West Pacific green sea turtle, and the East Indian-West Pacific green sea turtle (NMFS 2021a).

Historically, the Hawaii DSLL fishery has observed 25 green sea turtle interactions between 2004 and 2022 with an estimated 128 green sea turtle interactions during this time frame (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a). NMFS estimates about 95% (95% CI = 77 to 99%) of green turtles captured in the DSLL fishery have been dead when they were hauled back to the fishing vessel based on the number of observed mortalities (this analysis).

Summary of the Status of Green Sea Turtles

In this section of this biological opinion, we explained that six DPSs of green sea turtles are likely to be adversely affected by the proposed action: Central North Pacific, East Pacific, Southwest Pacific, and East Indian-West Pacific green sea turtles are threatened; and Central West Pacific and Central South Pacific green sea turtles are endangered. We used our knowledge of the species' demography and population ecology to capture the primary factors that appear to determine the green sea turtle's population dynamics.

As previously mentioned in the green sea turtle status discussions, sea turtles face various threats throughout each stage of their respective life cycles. We also examined other threats that may be applicable. However, as previously noted, caveats regarding the populations' trends apply to specific green sea turtle DPSs. Some Central West Pacific green sea turtle rookeries are increasing in nester abundance while other sites have unknown trends. Not enough data are available for the Central South Pacific green sea turtle to identify an overall trend. The East Indian-West Pacific green sea turtles rookeries have both increasing and decreasing trends depending on location, yet not enough data are available at this time to determine an overall trend for this particular DPS.

While the Central North Pacific and East Pacific green sea turtle DPSs face both natural and anthropogenic threats at all life cycle stages, the number of recruits into their populations appears to balance the loss from the various stressors which illustrates the interrelated nature of the different life stages and threats; and how these successive variables tend to cause or balance out to create the positive trend apparent in both the Central North Pacific and East Pacific green sea turtles.

2.2.11 Leatherback Sea Turtle

Distribution and Population Structure

Leatherback sea turtles are listed as endangered under the ESA throughout their global range. The leatherback turtle has the most extensive global distribution of any reptile and is distributed throughout the oceans of the world (Figure 19) from the equator to subpolar regions in both hemispheres. Leatherback turtles spend the majority of their lives at sea, where they develop, forage, migrate, and mate, nesting on beaches on every continent except Europe and Antarctica, and several islands of the Caribbean and the Indo-Pacific (Eckert et al. 2012; NMFS and FWS 2020a). Seven populations are currently recognized: (1) Northwest Atlantic; (2) Southeast Atlantic; (3) Southwest Atlantic; (4) Northeast Indian; (5) Southwest Indian; (6) West Pacific; and (7) East Pacific Ocean populations (NMFS and FWS 2020a).

For purposes of this opinion, we focus on the two populations (i.e., West Pacific and East Pacific) occurring within the Pacific Ocean basin. The marine distribution for Pacific leatherback sea turtles (hereafter Pacific leatherbacks) extends north into the Sea of Japan, northeast and east across the North Pacific to the west coast of North America (predominantly off California), west to the South China Sea and Indonesian Seas, and south into the high latitude waters of the western South Pacific Ocean and Tasman Sea (Figure 20; Benson et al. 2011).

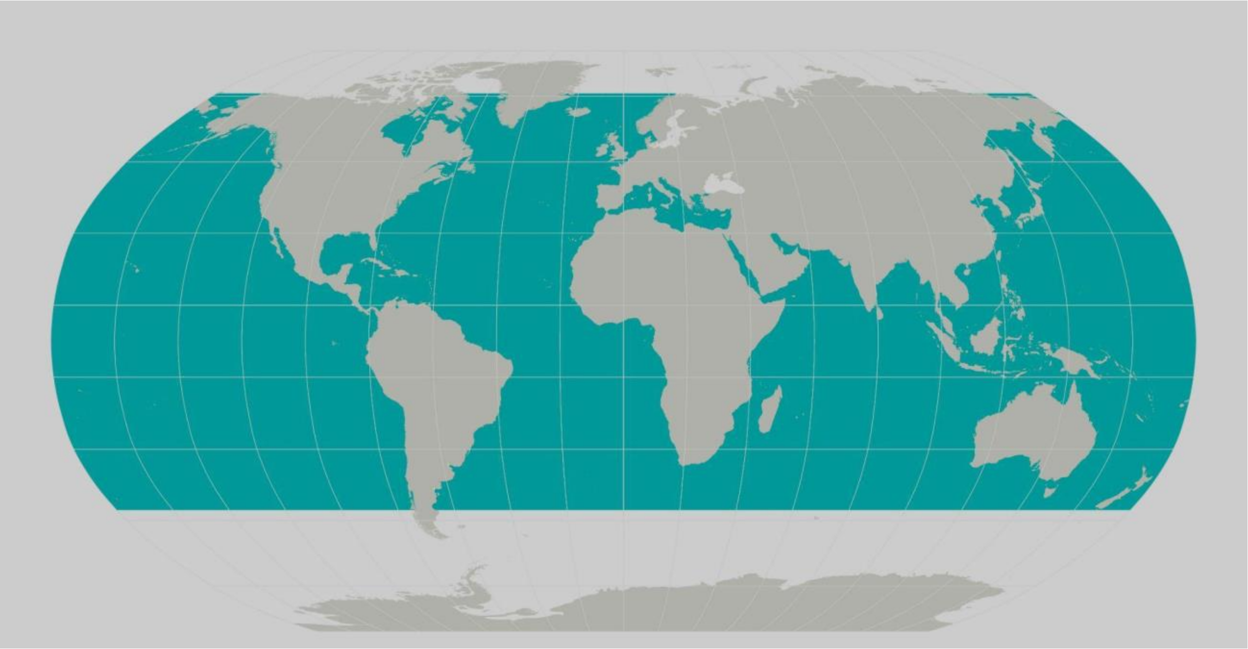

Figure 19. Map identifying the range of the endangered leatherback sea turtle. From NMFS https://www.fisheries.noaa.gov/species/leatherback-turtle#overview

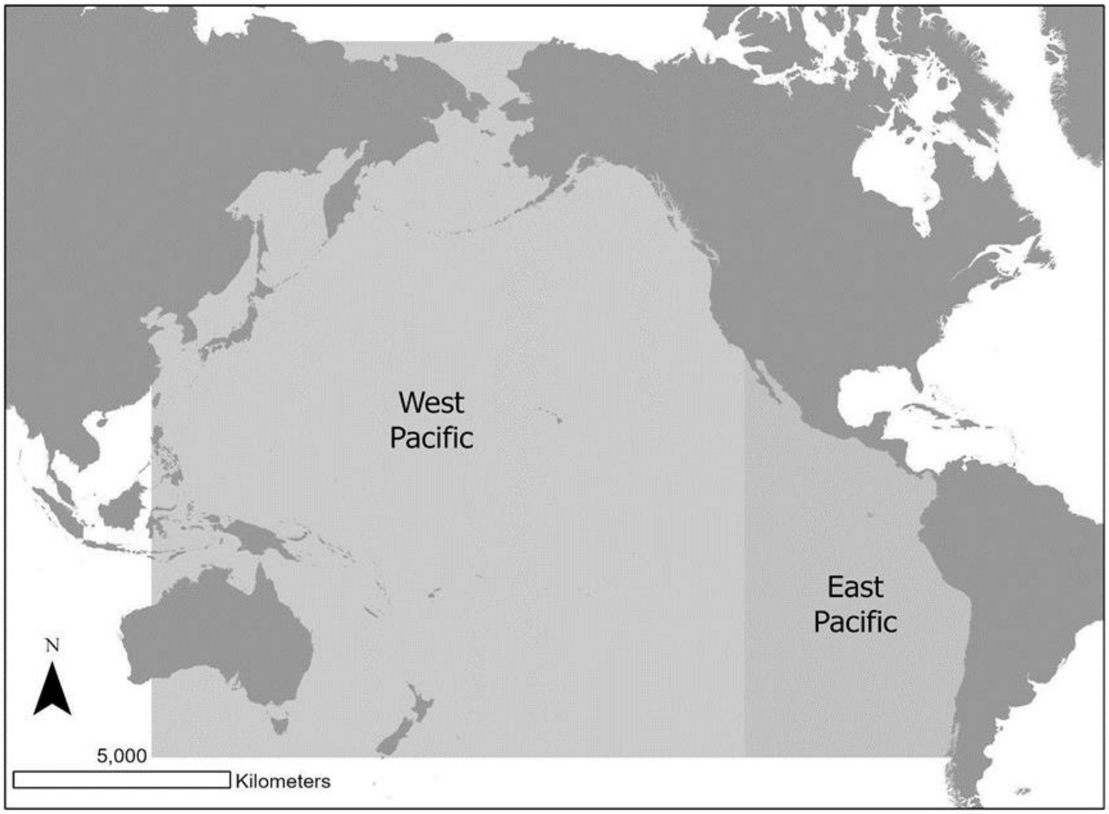

Figure 20. Map identifying the ranges of the East and West Pacific populations of leatherbacks.

Genetic samples from leatherback sea turtles interacting with the DSLL fishery indicate that most of these individuals are from the West Pacific population (94% or 15 of the 16 samples) while 6% (one of the 16 samples) are from the East Pacific population (NMFS unpublished data).

West Pacific Population

We define the West Pacific population as leatherback turtles originating from the West Pacific Ocean, with the following boundaries: south of 71° N, north of 47° S latitudes and east of 120° E, and west of 117.124° W longitudes (NMFS and FWS 2020a). Indonesia, Papua New Guinea, and Solomon Islands have been identified as the core nesting areas for this population (Figure 21; Benson et al. 2007a; Benson et al. 2007b; Benson et al. 2011; Benson 2018). Low levels of nesting are also reported in Vanuatu and the Philippines (NMFS and FWS 2020a). Recently, a new leatherback turtle nesting area was identified at Buru Island, Maluku province of Indonesia where approximately 200 nests are laid annually (NMFS and FWS 2020a; WWF 2018, 2022). However, long-term monitoring data for this population is geographically limited to the Bird's Head Peninsula in West Papua at Jamursba Medi and Wermon nesting beaches, which represent an estimated 50% to 75% of all nesting in the West Pacific (NMFS and FWS 2020a).

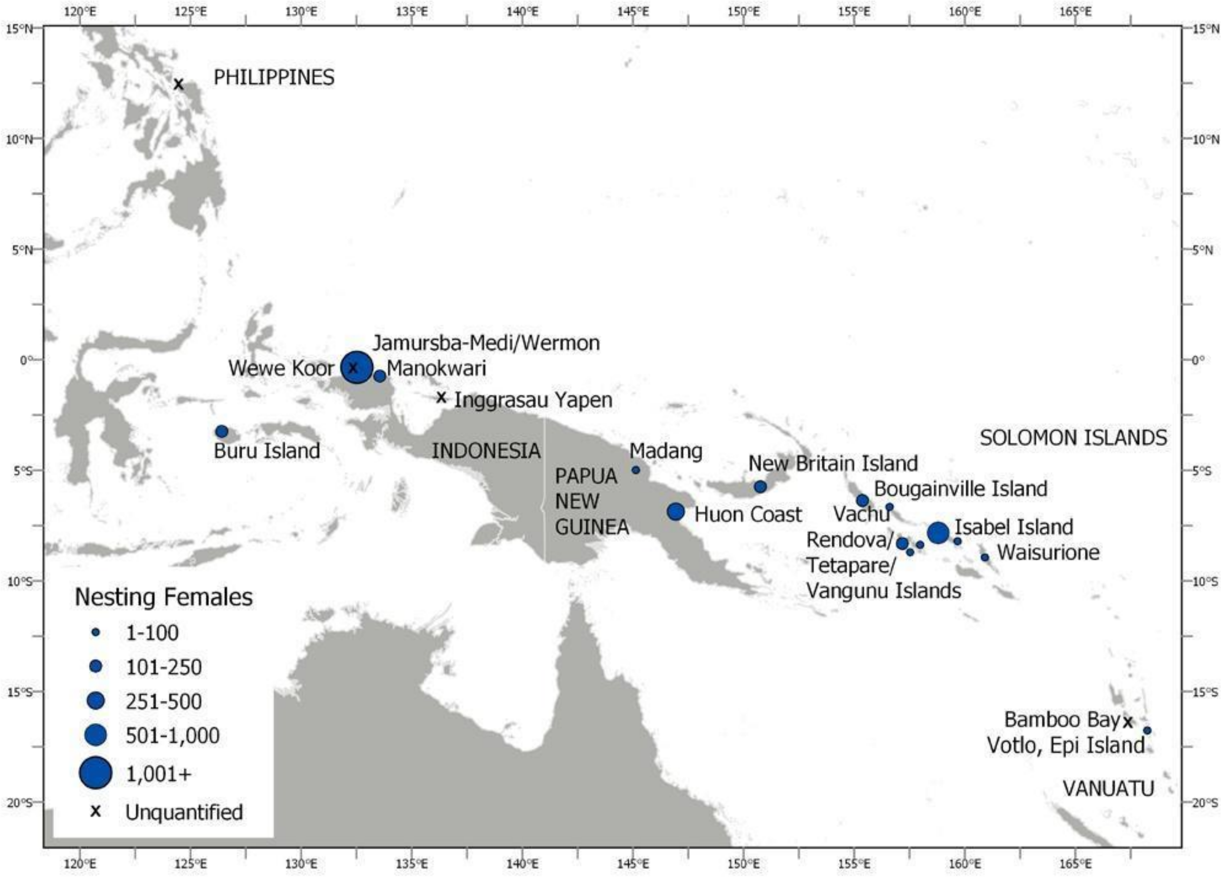

Figure 21. Nesting sites of the West Pacific leatherback population (NMFS and FWS 2020a).

The West Pacific population exhibits metapopulation dynamics and genetic population structure (NMFS and FWS 2020a). While mtDNA analyses of 106 samples from Indonesia, Papua New Guinea, and Solomon Islands did not detect genetic differentiation among nesting aggregations (Dutton et al. 2007), microsatellite DNA analyses indicate fine-scale genetic structure (Dutton et al. 2007; NMFS SWFSC unpublished data). While we often consider these different nesting aggregations separately, together they comprise the West Pacific population (NMFS and FWS 2020a).

Two life history strategies are documented in the West Pacific leatherback population: winter boreal nesters (October to March) and summer boreal nesters (April to September). The most consistent monitoring effort has been at Jamursba Medi beach, and its nesting females are primarily summer boreal nesters. Wermon beach has a stronger bimodal pattern of nesting, with summer and winter boreal nesters in roughly equal proportions. There is historical evidence to suggest a similar bimodal nesting strategy in other nesting aggregations, but data is lacking to quantify the current extent of summer nesting activity in the Solomon Islands and Papua New Guinea where the majority of nesting activity occurs during winter months (NMFS and FWS 2020a).

Migration and foraging strategies vary based on these life history strategies, likely due to prevailing offshore currents and, for hatchlings, seasonal monsoon-related effects experienced as hatchlings (Benson et al. 2011; Gaspar et al. 2012). Summer nesting females forage in Northern Hemisphere foraging habitats in Asia and the North Pacific Ocean, while winter nesting females migrate to tropical waters of the Southern Hemisphere in the South Pacific Ocean (Figure 5; Benson et al. 2011; Harrison et al. 2018). The lack of crossover among seasonal nesting populations suggests that leatherback turtles develop fidelity for specific foraging regions likely based on juvenile dispersal patterns (Benson et al. 2011; Gaspar et al. 2012; Gaspar and Lalire 2017). Stable isotopes, linked to particular foraging regions, confirm nesting season fidelity to specific foraging regions (Seminoff et al. 2012). Adult West Pacific leatherback sea turtles interacting with the DSLL fishery are most likely summer boreal nesters using the North Pacific transition zone (or Kuroshio extension), equatorial eastern Pacific, or the California Current Extension (Figure 22).

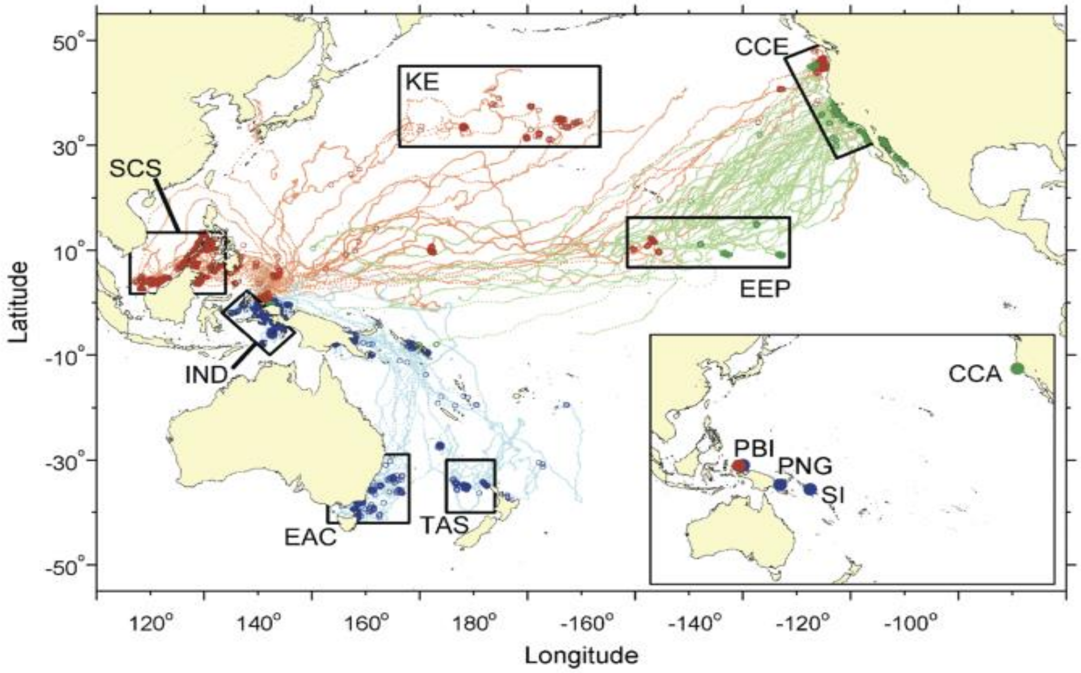

Figure 22. Satellite tracks from 126 West Pacific leatherback sea turtles. Color of track indicates deployment season: red = summer nesters, blue = winter nesters, green = deployments at central California foraging grounds. Inset shows deployment locations; PBI = Papua Barat, Indonesia, PNG = Papua New Guinea, SI = Solomon Islands, CCA = central California. Black boxes represent ecoregions for which habitat associations were quantitatively examined: SCS = South China, Sulu and Sulawesi Seas, IND = Indonesian Seas, EAC = East Australia Current Extension, TAS = Tasman Front, KE = Kuroshio Extension, EEP = equatorial eastern Pacific, and CCE = California Current Ecosystem (from Benson et al. 2011; NMFS and FWS 2020a).

East Pacific Population

We define the East Pacific population as leatherback turtles originating from the East Pacific Ocean, north of 47° S and south of 32.531° N latitudes, and east of 117.124° W, and west of the Americas. The East Pacific leatherback population is characterized by somewhat continuous and low density nesting across long stretches of beaches along the coast of Mexico and Central America (NMFS and FWS 2020a). The best available genetic data indicate a high degree of connectivity among nesting aggregations that comprise a single population without population subdivision (NMFS and FWS 2020a). This population generally occupies a marine distribution distinct from the West Pacific population, although there are some pelagic areas where East and West Pacific populations overlap. Genetic analyses of juvenile and adult leatherback sea turtles caught in fisheries off Peru and Chile indicate that a proportion (approximately 16 percent of sampled turtles) are from West Pacific rookeries (Donoso and Dutton 2010; NMFS and FWS 2013).

The foraging range of the East Pacific leatherback extends into coastal and pelagic waters of the southeastern Pacific Ocean; however, foraging is not as widely separated as compared to the West Pacific DPS (NMFS and FWS 2020a). Tagging studies have shown that East Pacific post-nesting females migrate southward to the south Pacific after nesting in Costa Rica (Figure 23; Shillinger et al. 2008) and commonly forage offshore in the South Pacific Gyre in upwelling areas of cooler, deeper water and high productivity (Shillinger et al. 2011). During the nesting season, they stay within the shallow, highly productive, continental shelf waters (Shillinger et al. 2010).

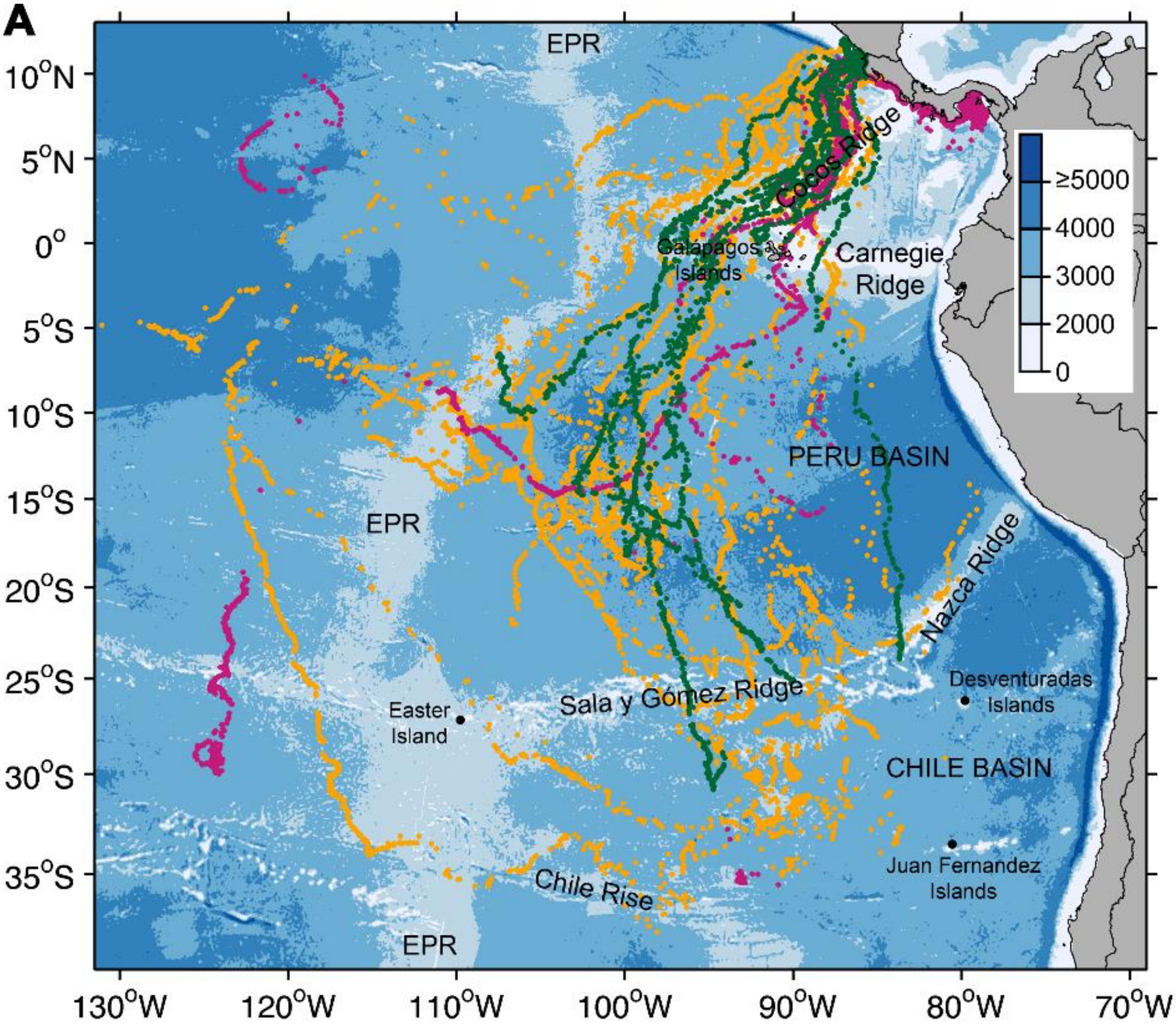

Figure 23. Satellite tracks for 46 post-nesting female leatherback sea turtles from the East Pacific population and nesting at Playa Grande, Costa Rica. Colors indicate the year of deployment: 2004 (*n* = 27, orange), 2005 (*n* = 8, purple), and 2007 (*n* = 11, green; From Shillinger et al. 2008; NMFS and FWS 2020a).

Status and Trends

Leatherback sea turtles are listed as endangered under the ESA throughout their global range. Tiwari et al. (2013) estimated that the western Pacific population had 1,438 mature individuals, which meets the IUCN threshold for endangered. However, because their estimates led them to conclude that this population had a high probability of declining by about 96% by 2040 (to about 57 nests and 260 adult females), they classified the western Pacific leatherback turtle population as critically endangered (Tiwari et al. 2013; Wallace et al. 2013b). The East Pacific population has declined by more than 97% over three generations with less than 140 nesting females per year (Wallace et al. 2013b) and are similarly classified as critically endangered by the IUCN.

Our ability to estimate leatherback population abundance is complicated by the life history of the species. Data collected at nesting beaches are often the best available but do not provide information on life stages away from the nesting beaches (i.e., immature and mature males and immature females). Additionally, standardized nesting surveys are difficult to maintain over many, consecutive years and at all nesting beaches. Here we provide data that have been consistently collected using a standardized monitoring approach over the most recent available remigration interval, providing reasonable certainty that such data are representative of recent nesting at the identified beach.

West Pacific Population

Tapilatu et al. (2013) found a 78% decline in nesting from 1984 to 2011 at Jamursba Medi and a 62.8% decline in nesting at Wermon from 2002 to 2011. Overall they estimated a 5.9% per year decline in nesting abundance for both index beaches over this time period. Since the nesting decline reported in Tapilatu et al. (2013), leatherback nest numbers have fluctuated between 477 and 1531 at Jamursba-Medi and between 602 and 1566 at Wermon in the time period from April 2012-March 2013 to April 2021-March 2022 (Lontoh et al. In Prep). While the population does not demonstrate an increasing trend, it is encouraging that the population has not declined further in the past decade (Lontoh et al. In Prep; Figure 24). At a mean of 5.5 nests per female (Tapilatu et al. 2013; Lontoh 2014), these nest numbers equate to approximately 87 to 278 annual nesting females at Jamursba Medi and 109 to 285 annual nesting females at Wermon.

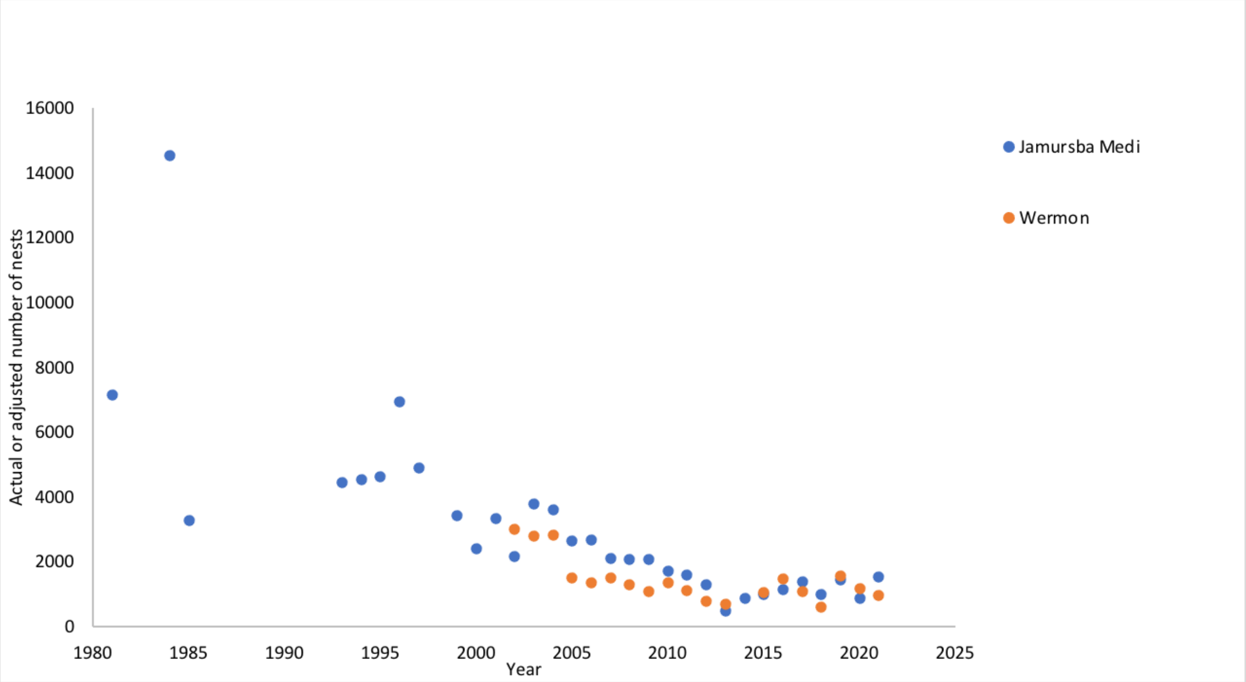

Figure 24. Actual or adjusted number of leatherback nests between 1981 and 2021 at Jamursba Medi and Wermon. Each year represents nests laid from April of one year to March of the following year (Lontoh et al. In Prep).

Using the best available data for the West Pacific leatherback population (Fitry Pakiding, University of Papua, pers. comm. 2020) and a Bayesian steady-state model, Martin et al. (2020a) provided a median estimate of the total number of nesting females (i.e., over one, 3-year, remigration interval) at Jamursba-Medi and Wermon beaches of 790 females, with a 95 % CI of 666 to 942 females, as a snapshot of abundance in 2017. As trends at these beaches between 2017 and 2022 appear to be stable (Figure 24), we consider the 2017 abundance to be the best estimate of current (2022) adult females for the index beaches.

To estimate the total number of nesting females from all nesting beaches in the West Pacific, we need to consider nesting at unmonitored or irregularly monitored beaches. Approximately 50% to 75% of West Pacific leatherback nesting occurs at Jamursba Medi and Wermon beaches (Dutton et al. 2007; NMFS and FWS 2020a). Given the lack of information regarding the extent of nesting in the rest of the West Pacific, we apply the most conservative reasonable estimate of 75% to the Martin et al. (2020a) estimate of 790 nesting females at Jamursba-Medi and Wermon beaches, the total number of nesting females in the West Pacific population would be 1,054 females with an overall 95% CI of 888 to 1,256 females. It should be noted that this estimate (i.e., 1,054) of nesting females for the West Pacific population based on more recent available information is an update of the NMFS and FWS (2020a) estimate (i.e., 1,277) which was based on a simple calculation that did not provide confidence or credible intervals.

Additional but lower levels of nesting have been documented elsewhere in Indonesia, including a new monitoring program established in 2017 on Buru Island (World Wildlife Fund [WWF] 2022), plus locations in Papua New Guinea, Solomon Islands, Vanuatu and the Philippines (NMFS and FWS 2020a). Monitoring at most of these additional sites has not been going on long enough to establish trends or abundance; therefore data from those nesting beaches cannot be used to reliably calculate those metrics at this time. An exception to this is the WWF program at Buru Island in Indonesia where data have been consistently collected since 2017 (WWF 2022). While it's only 6 years of data, this does span almost two remigration intervals (3.1 years; Martin et al. 2020a) for leatherbacks. These data indicate an increasing trend of 10.1% per year (Figure 25; 95% Confidence Interval: -26.1% to 46.3%) based on an exponential growth curve.

To encompass full boreal winter and summer nesting, the nesting data are censused from October to September, hence, for example, data from 2018 represents nesting beach data from October 2017 to September 2018. We note that the collection of data started in January of 2017 and the nest number for 2017 is missing data from October to December 2016 and therefore does not represent a full year of data. Using the same method to calculate total adult females as Martin et al. (2020a; remigration interval multiplied by the average of the last 4 years of nesters; see Equation 13 in Martin et al. 2020a), there are approximately 103 adult females nesting at Buru Island which contrasts with the modeled estimate of about 790 annual nesting females at Jamursba Medi and Wermon in 2017 (Martin et al. 2020a). Assuming a 73% female sex ratio (Benson et al. 2011) and based on NMFS' PVA results for median nest counts, the total number of adult leatherback sea turtles in the West Pacific Ocean population in 2017 would be 1,443 (([790/0.73]/0.75; 95% CI: 1,216-1,720) if the index beaches represent 75% of the population.

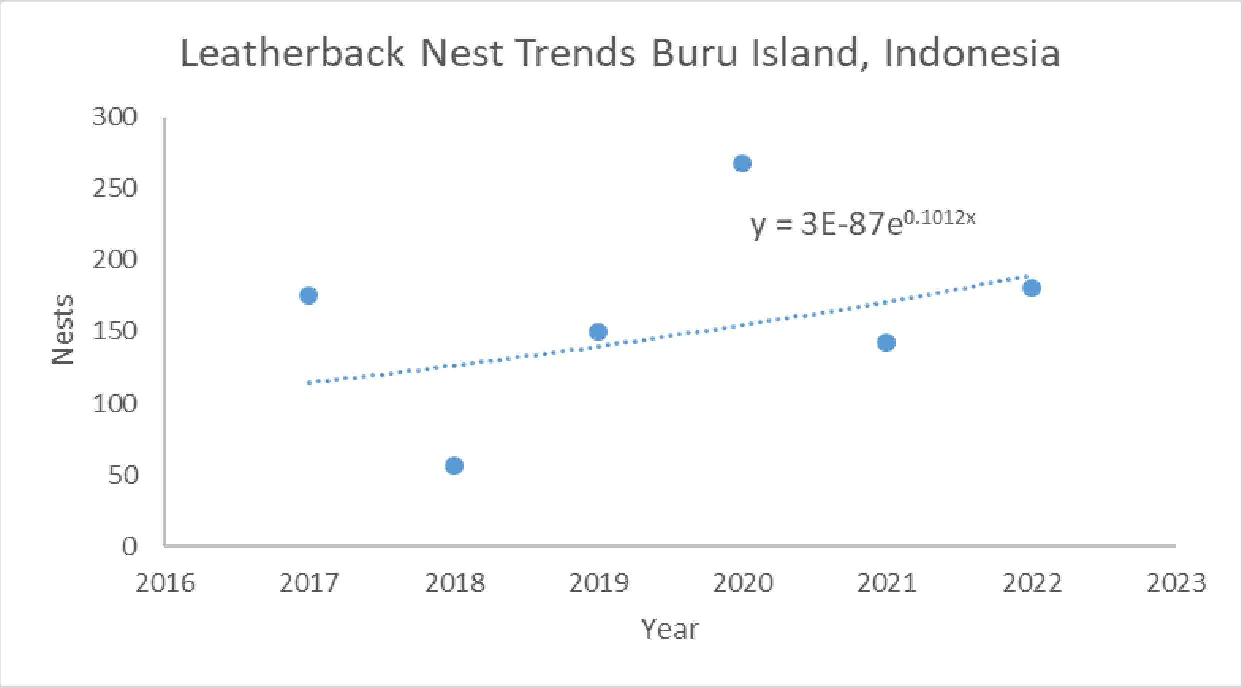

Figure 25. Leatherback nesting trend at Buru Island, data from WWF (2022).

Based on the estimates presented in Jones et al. (2012) for all Pacific populations, NMFS inferred an estimated West Pacific leatherback total population size (i.e., juveniles and adults) of 250,000 (95 percent confidence interval 97,000 to 535,000) in 2004. Based on the relative change in the estimates derived from Jones et al. (2012) and the more recent Martin et al. (2020a), NMFS estimates the current juvenile and adult population size of the West Pacific leatherback population is around 100,000 sea turtles (95 percent confidence interval 47,000 to 195,000 individuals). As nesting numbers have been stable since 2017, we assume these abundances are representative of 2022 abundances as well.

The Western Pacific population has been exhibiting low hatching success and decreasing nesting population trends due to past and current threats (NMFS and FWS 2020a). The low estimated nesting female abundance of the West Pacific population places it at elevated risk for environmental variation, genetic complications, demographic stochasticity, negative ecological feedback, and catastrophes (NMFS and FWS 2020a). These processes, working alone or in concert, place small populations at a greater extinction risk than large populations, which are better able to absorb impacts to habitat or losses in individuals (NMFS and FWS 2020a). Low site fidelity, which is characteristic of the species, results in the dispersal of nests among various beaches (NMFS and FWS 2020a). This may help to reduce population level impacts from threats which may disproportionately affect one area over another, but may also place nests in locations that are likely unmonitored and not protected from human poaching or predation, thereby increasing threats to the population. Due to its small size, this population has restricted capacity to buffer such losses (NMFS and FWS 2020a).

The median trend in annual nest counts estimated for Jamursba-Medi nesting beaches from data collected from 2001-2017 was -5.7 percent annually (95% CI: -16.2% to 5.3%; NMFS and FWS 2020a). The median trend in annual nest counts estimated for Wermon nesting beaches from data collected from 2006 to 2017 (excluding 2013–2015 due to low or insufficient effort) was -2.3 percent annually (95% CI: -19.8 to 14.9%; NMFS and FWS 2020a). Using their Bayesian steady-state model, Martin et al. (2020a) estimate the combined trends for Jamursba-Medi and Wermon to be a mean of -6.0% (95% CI: -24.1 to 12.2%) annually. We note that the data from 2018 to 2021 are preliminary and only provided to NMFS from the authors (Lontoh et al. In Prep) as the graph shown in Figure 24. Until we receive the detailed raw monthly data from the nesting beaches, the growth trend analysis of Martin et al. (2020a) cannot be updated, so we do not have any updated modeled estimates of future growth rates based on this new information and we rely on the estimates of Martin et al. (2020a) for population growth rates after 2021. In addition, given the substantial declines in the population from 1984 to 2012, the data from 2012 to 2021 are likely not of long-enough duration to definitely state that the population is now at least stable, but we cannot be certain of this until the raw data are available for analysis.

Benson et al. (2020) assessed in-water trends for leatherbacks using the California Current Ecosystem foraging area (see Figure 22). They found that abundance for the total population is declining by a mean of 5.6% per year (95% CI: -9.8% to -1.5%) over a similar timeframe (1990 to 2017). This indicates the 6.0% per year decline found by Martin et al. (2020a) on the index nesting beaches may be mirrored in the large juvenile and male portions of the mixed North Pacific foraging population. Therefore, we consider the Martin et al. (2020a) trends in annual nest counts an accurate index of the population's growth rate through 2017.

The conservation efforts at Jamursba-Medi and Wermon have continued (Pakiding et al. 2020) and from 2017 to 2019 hatchling production between April and September alone (i.e. exclusive of the Wermon boreal winter nesting season) increased to 32,000 – 50,000 hatchlings per year (Pakiding et al. 2020) in contrast with a mean annual hatchling production of 21,996 from 2005 to 2013 (Tapilatu 2014; Pakiding et al. 2020). This is in part due to increased effort to protect nests from predation, tidal inundation, erosion and high sand temperatures. Nest success rates increased from about 35% prior to 2017 to over 50% since 2017 (Pikiding et al. 2020) but also due to stable numbers of nesting females ranging from 87 to 279 annually at Jamursba Medi and 109 tp 285 annually at Wermon.

A recent report on sea turtle bycatch in New Zealand's shallow-set longline fishery indicates a marked increase in leatherback captures (Figure 26; Dunn et al. 2022). Observed captures have increased from a low of one in 2008 to a high of 50 in 2022. Accounting for effort, the resulting trend in CPUE shows a statistically significant positive trend, increasing at a rate of 19.9% per year (Figure 26; 95% Confidence Interval: 8.4% to 31.3%) based on an exponential growth curve. While fishery captures can be influenced by numerous environmental factors that can disconnect them from population trends, the strength of this trend suggests the potential for more leatherback turtles in the water in recent years, which is supportive of the information we have on nesting beach trends (WWF 2022, Lontoh et al. In Prep) and increased hatchling production (Pakiding et al. 2020) for West Pacific leatherbacks. In summary, we acknowledge that there is a great deal of uncertainty associated with the current status of West Pacific leatherback sea turtles, and assuming a long term decline of 6% per year may underestimate future projected population sizes if the recent nesting beach trends continue. However, until new nesting beach data are available in sufficient detail to reassess the population trend model, we consider the projected declines of Martin et al. (2020a) to be the best available data but acknowledge that these declines did not occur from 2017 to 2021.

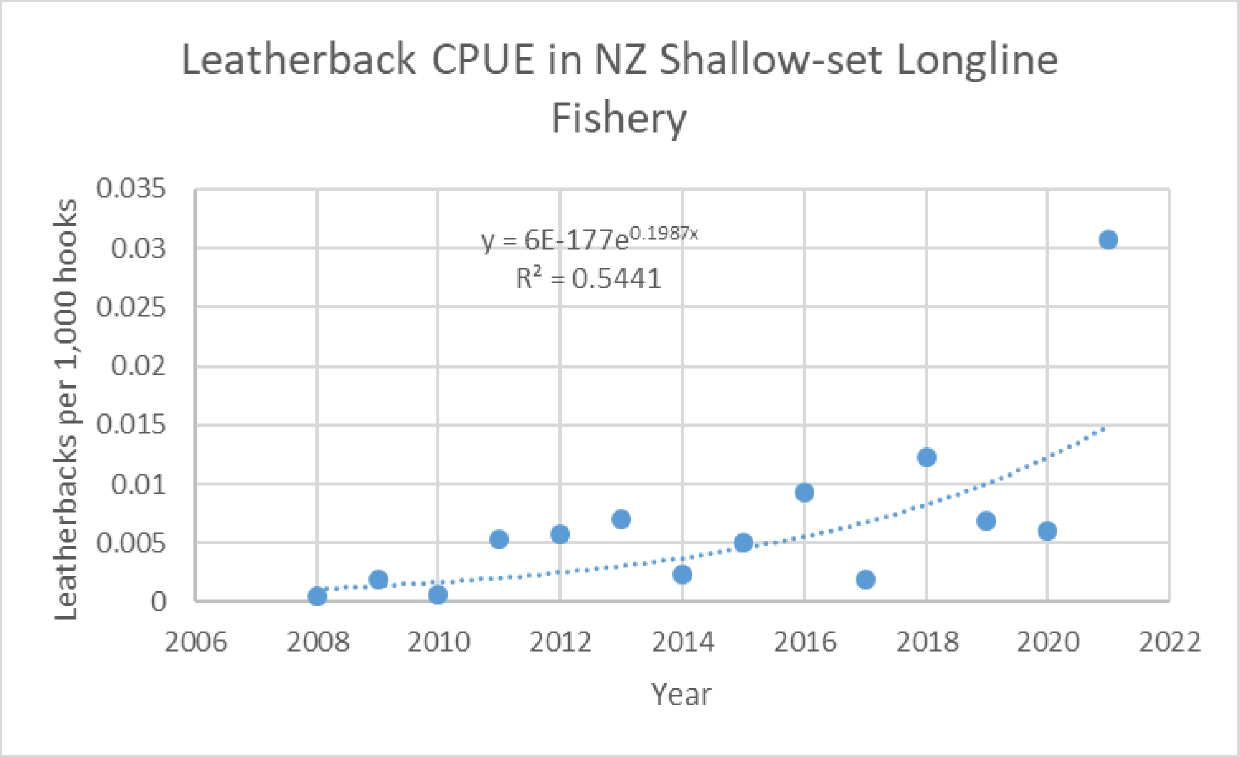

Figure 26. Trend in CPUE of leatherback sea turtles in New Zealand shallow-set longline fishery (data from Dunn et al. 2022).

Martin et al. (2020a) estimated the mean and median time until the West Pacific population declines to 50 percent, 25 percent, and 12.5 percent of its 2017 estimated abundance and Siders et al. (2023) updated these results to 2021 assuming the population declined at a rate of 6% per year from the 2017 abundance estimates. Results of this updated modeling effort indicate that the adult female portion of West Pacific leatherbacks nesting at Jamursba Medi and Wermon beaches are predicted to decline to 50 percent of their 2017 abundance in a mean of about 9 years starting in 2021 (or by about 2030; 95 % CI from 1 to 22 years) and to 25 percent of their 2017 abundance in a mean of about 20 years (or by about 2041; 95 % CI from 8 to 37 years). Again, these estimates assume a mean decline of 6% per year since 2017.

East Pacific Population

Using the best data available for the East Pacific population, NMFS and FWS (2020a) calculated the index of total nesting females to be a minimum of 755 females. We consider this the best available estimate because it is based on a complete compilation of the most recent 4-year remigration interval data for each nesting beach monitored. Model-based estimates of abundance with credible or confidence intervals are unavailable for this population. This index of nesting includes females from known nesting beaches in Costa Rica and Nicaragua, and an estimated 70 to 75 percent of total nesting in Mexico (Gaona and Barragan 2016). It does not include females nesting at inconsistently monitored beaches in Mexico, including: Agua Blanca (40 km in Baja California), Playa Ventura, Playa San Valentín, Piedra de Tlacoyunque, and La Tuza (Martínez et al. 2007). Nesting is rare in other nations (e.g., Ecuador, El Salvador, and Panama)(Sarti et al.

1999). Assuming the index beaches represent 75% of all nesting, we estimate a total of 1,007 adult females ([755/75]*100 = 1,006.7 rounded to 1,007). Assuming a sex ratio of 79% female (Santidrian Tomillo et al. 2014) suggests a total of 1,274 adults in 2020 inclusive of both males and females. We do not have data to assess the total population size; however, based on data in Table 2 of Jones et al. (2012), we expect that adults comprise a mean of 2.1% (CI: 1.3% to 3.7%) of the total population size, which would suggest a total population size of 60,611 (CI: 34,050 to 95,462) individuals in 2020.

The East Pacific leatherback population has undergone dramatic declines over the last three generations (Wallace et al. 2013b; NMFS and FWS 2020a), and to date there is no sign of recovery. In Costa Rica, a 15.5 percent annual rate of decline in nesting females has been documented at Las Baulas from 1988/1989 through 2015/2016 (Table 5; NMFS and FWS 2020a). In Mexico, a positive trend has been recorded at some nesting beaches (i.e., Barra de la Cruz/Playa Grande +9.5 percent annually) but a negative trend has been recorded in other areas (i.e., Cahuitan -4.3 percent annually over the same period; Table 5). Overall, the current and potential future trend for the population is uncertain and additional years of data are needed to ascertain if recovery is occurring in Mexico. For our analysis we use the high nest numbers and mean trends across these four index beaches to estimate a weighted average trend of -8.1% (see Table 5 for details on the calculation).

Table 5. Trends in nesting females for nesting beaches in the East Pacific leatherback population with at least 9 years of data (from NMFS and FWS 2020a).

Beach	**Years**	**High Nest Numbers**	**Mean Trend**	**95% Confidence Interval**	**Weighting Factor (High Nest Numbers * Mean Trend**
Mexico					
Tierra Colorada	1996-2016	503	0.6%	-17.1 to 18.9%	301.8
Barra Cruz/Grande	1996-2016	365	9.5%	-6.5 to 25.8%	3467.5
Cahuitan	1997-2016	75	-4.3%	-22.1 to 17.6%	-322.5
Costa Rica					
Las Baulas	1988-2015	1504	-15.5%	-23.1 to -7.8%	-23312
Weighted Average Trend = [30.18+3467.5+(-322.5)+(-23312)]/[503+365+75+1504] = -8.11%					

Population Dynamics

Avens et al. (2020) estimate a mean age at maturity of 17 years (range of 12 to 28 years) for Pacific leatherbacks (East and West populations combined). In Indonesia, the mean size of nesting females is 161 cm curved carapace length (CCL) with an observed minimum of 138 cm CCL (Hitipeuw and Maturbongs 2002; Lontoh 2014). The most comprehensive information on clutch frequencies for West Pacific leatherback nesting females is from Jamursba-Medi and Wermon beaches (Indonesia), where Tapilatu et al. (2013) estimated an average clutch frequency of 5.5 (+/- 1.6, range 3-10). For the overall West Pacific population, Lontoh (2014) found evidence that clutch frequency and remigration interval may vary with foraging life history. Mean clutch frequency varied among individuals, but was higher for the North Pacific Transition Zone foraging group (6.1) than the Northeast Pacific (5.6) and South China Sea (4.8) foraging groups and increased with turtle size (curved carapace width) (Lontoh 2014). The range of remigration intervals was 1-6 years, and the mean (± SE) remigration interval was greater for the Northeast Pacific foraging group (4 ± 0.3 years) than the North Pacific Transition Zone foraging group (2.3 ± 0.4 years) and the South China Sea foraging group (2.9 ± 0.4 years) (Lontoh 2014). Mean clutch sizes for West Pacific leatherbacks range from 76 to 94 eggs per nest (Jino et al. 2018; Pilcher 2011; Tapilatu and Tiwari 2007).

For the East Pacific population, mean sizes of nesting females range from 144 to 147 cm CCL with a minimum observed size of 125 cm CCL at Playa Langosta, Costa Rica (NMFS and FWS 2020a). Mean remigration intervals range from 3 to 3.7 years; clutch size ranges from means of 62 to 65 eggs per nest, clutch frequencies range from 3 to 12 nests per female with means of 5.5 and 7.2 for nesting beaches in Mexico (summarized in NMFS and FWS 2020a).

The mean hatchling emergence success (i.e., the number of hatchlings that successfully emerge from the nest onto the beach) is beach and seasonally specific and can range between 25 and 60 percent in the West Pacific and 35 to 52 percent in the East Pacific (summarized in NMFS and FWS 2020a).

Diving and Social Behavior

Leatherback sea turtles feed from near the surface to depths exceeding 1,000 m, including nocturnal feeding on tunicate colonies within the deep scattering layer (Spotila 2004). Although leatherback sea turtles can dive deeper than any other reptile, most dives are less than 80 m (Shillinger et al. 2011). Migrating leatherback sea turtles spend a majority of their time submerged and display a pattern of continual diving. They appear to spend almost the entire portion of each dive traveling to and from maximum depth, suggesting continual foraging along the entire depth profile (Eckert et al. 1989). Typical dive durations averaged 6.9 to 14.5 minutes per dive, with a maximum of 42 minutes (Eckert et al. 1986).

Threats to the Species

The primary ongoing threats to leatherback sea turtles worldwide are fisheries bycatch, marine debris or other entanglement, and the legal and illegal directed harvest of nesting female turtles and their eggs (NMFS and FWS 2020a). Other threats to this species include changing environmental conditions due to climate change (e.g., sand temperatures that result in egg or hatchling mortality or changes in hatchling sex ratios, erosion of nesting beaches due to rising sea levels and increased storm frequency and magnitude), vessel strikes, pollution, and ingestion of marine debris (Tiwari et al. 2013; NMFS and FWS 2020a). Below, we summarize the main anthropogenic threats facing each population (i.e., West Pacific and East Pacific). We start with a general discussion about the impacts of climate change on Pacific leatherbacks. Past and ongoing threats to this species within the action area are discussed in more detail below in the Environmental Baseline section of this biological opinion.

Climate change represents a threat to both the East and West Pacific leatherback populations. The impacts of climate change include: increases in temperatures (air, sand, and sea surface); sea level rise; increased coastal erosion; more frequent and intense storm events; and changes in oceanographic regimes and currents. A warming climate and rising sea levels can impact leatherback turtles through changes in beach morphology and sand temperature (Benson et al. 2015). Leatherback sea turtles are probably already beginning to be affected by impacts associated with anthropogenic climate change given low hatch success due to lethal beach temperatures and beach erosion (Bellagio Sea Turtle Conservation Initiative 2008; NMFS and FWS 2013; NMFS and FWS 2020a; Tapilatu and Tiwari 2007). West Pacific leatherback turtles have evolved to sustain changes in beach habitats given their proclivity to select highly dynamic and typically narrow beach habitats, and therefore at the population level can likely sustain some level of nest loss (NMFS and FWS 2020a). However, the increasing frequency of storms and high water events, perhaps as a result of climate change, can result in increased and perhaps unnatural loss of nests. In recent years, management and conservation practices have included relocating erosion-prone nests in Indonesia, Papua New Guinea, and the Solomon Islands to bolster hatchling production (NMFS and FWS 2020a).

Similar to other sea turtles, leatherback hatchling sex is determined by nest incubation temperature, with higher temperatures producing a greater proportion of females (Mrosovsky 1994). Sand temperatures fluctuate between 28.6 and 34.9 °C at Jamursba Medi and between 27.0 and 32.7 °C at Wermon (Tapilatu and Tiwari 2007). At Wermon, the sand is black, yet beach temperatures are lower perhaps because peak nesting coincides with the monsoon season (Tapilatu and Tiwari 2007). High average sand temperatures are indicative of a female-biased West Pacific leatherback population at Jamursba Medi nesting beaches (Tapilatu et al. 2013; Tapilatu and Tiwari 2007). A significant female bias was also reported by Binckley et al. (1998) for East Pacific leatherback hatchlings at the Playa Grande nesting beach in Costa Rica (Plotkin 1995). In addition to impacts on Pacific leatherback nesting success and sex ratios, the impacts of a warming ocean may also affect the environmental variables of their pelagic migratory and foraging habitat, which may further exacerbate population declines (NMFS and FWS 2020a).

Natural threats include predation on sea turtle hatchlings by birds and fish (see Vose and Shank 2003). Reported predation of leatherback hatchlings includes tarpons (Nellis and Henke 2000), gray snappers (Vose and Shank 2003), ghost crabs, great blue and yellow-crowned herons, and crested caracaras (Santidrian Tomillo et al. 2010). Adult leatherbacks are preyed upon by large predators, such as jaguars, tigers, killer whales, sharks, and crocodiles (reviewed by Eckert et al. 2012).

Major anthropogenic threats to the species are fisheries bycatch, direct harvest and alteration of nesting habitat, which are briefly described below (NMFS and FWS 2020a). In addition, habitat changes attributed to changing environmental conditions, pollution and marine debris are also threats to this species (Tiwari et al. 2013).

The primary threat to the West Pacific population is the legal and illegal harvest of leatherback turtles and their eggs. The removal of nesting females from the population reduces both abundance and productivity; egg harvest reduces productivity and recruitment. Leatherback turtles are protected by regulatory mechanisms in all four nations where this population nests, but laws are largely ignored and not enforced (NMFS and FWS 2020a). This is due to the extreme remoteness of beaches, customary and traditional community-based ownership of natural resources (which includes sea turtles), and overall lack of institutional capacity and funding for enforcement (Gjertsen and Pakiding 2011; Kinch 2006; Von Essen et al. 2014).

Directed killing of nesting females, and male and female juvenile and adult leatherbacks in their foraging areas has been documented in all four countries where this population nests (Bellagio Sea Turtle Conservation Initiative 2008; Jino et al. 2018; Kinch J et al. 2009; Petro et al. 2007; Suarez and Starbird 1995; Tiwari et al. 2013). While a number of relatively recent NMFS and FWS funded programs are working to quantify and reduce directed take, egg and turtle harvest is a well-documented past and current threat and is prolific throughout the West Pacific leatherback range (Bellagio Sea Turtle Conservation Initiative 2008; NMFS and FWS 2013; Tapilatu et al. 2017; Tiwari et al. 2013). In Indonesia, the direct harvest of turtles and eggs likely persists, although this threat has been minimized at Jamursba Medi, Wermon, and Buru Island beaches due to the presence of monitoring programs and associated educational outreach activities (NMFS and FWS 2020a). In the Maluku islands of Indonesia, several villages of the Kei islands have engaged in an indigenous hunt (directed fishery) of juvenile and adult leatherback turtles foraging in coastal habitats for decades.

While recent programmatic efforts are working to monitor and reduce this impact, the hunt was historically estimated to take over 100 leatherback turtles annually (NMFS and FWS 2020a; Suarez and Starbird 1996; WWF 2018). In Papua New Guinea, egg harvest and killing of nesting females is still a major threat despite the fact that leatherback turtles have been protected since the 1976 Fauna (Protection and Control) Act. The killing of nesting females and directed harvest of eggs in Vanuatu and the Solomon Islands is also well documented (Bellagio Sea Turtle Conservation Initiative 2008; NMFS and FWS 2013, 2020a).

The primary cause of the historical decline of the East Pacific leatherback population was the legal and illegal (post conservation measures) harvest of nesting females and eggs. The extensive and prolonged effects of comprehensive egg harvest levels of nearly 90 percent for about two decades have depleted the leatherback turtle population in Costa Rica and Mexico (Martínez et al. 2007; Santidrian Tomillo et al. 2008; Wallace and Saba 2009). To reduce the harvest of turtles and eggs, several regulatory mechanisms and protections have been established in the three nations hosting nesting beaches. In Mexico, the harvest of turtles and eggs is now prohibited as a result of national legislation. In Costa Rica, establishment of Parque Nacional Marino Las Baulas in 1991 ensured increased protection at three nesting beaches (Playa Grande, Playa Ventanas, and Playa Langosta), greatly reducing egg poaching in the area. Though conservation efforts have reduced the levels of both, egg poaching remains high and affects a large proportion of the East Pacific breeding population.

Fishery bycatch in coastal and pelagic fisheries is also a major threat to the West Pacific population, which is exposed to domestic and international fisheries throughout its extensive foraging range. Bycatch of leatherback turtles has been documented for a variety of gillnet and longline fisheries in the Pacific Ocean, but little is known about the total magnitude or full geographic extent of mortality (NMFS and FWS 2020a). Detailed bycatch data are available for U.S.-managed pelagic fisheries operating in the central and eastern Pacific Ocean due to regulatory mandates and high levels of observer coverage.

For West Pacific Ocean nesting populations, several areas of high risk were identified in the north and central Pacific, with the greatest risk adjacent to primary nesting beaches in tropical seas of Indo-Pacific islands, in the Sulu, Sulawesi, and South China Seas. In the central Pacific region, the area with the highest probabilities of capturing turtles in Hawaii longline fisheries was predicted to occur southwest of the Hawaiian Islands, between the equator and up to 15°to 20° N, from January through March. The North Pacific Transition Zone between 30° N and 35° N poses moderate risk to leatherbacks at this time. From October through December there is a moderate risk from 140° W to 120° W, and patchy areas of risk broadly distributed throughout the North Pacific from April through December (Roe et al. 2014). For eastern Pacific nesting populations, the greatest risk was identified in the South Pacific Gyre.

The summer nesting component of the population exhibits strong site fidelity to the central California foraging area (Benson et al. 2011), which puts migrating leatherbacks at risk of interacting with U.S. and international pelagic longline fleets operating throughout the Central and North Pacific oceans. Fishery observer data collected between 1989 and 2015 of 34 purse seine and longline fleets across the Pacific documented a total of 2,323 sea turtle interactions, of which 331 were leatherback turtles (Clarke 2017). Two bycatch hotspot areas were identified: one in central North Pacific (which likely reflects the 100 percent observer coverage in the Hawaii shallow-set longline fishery) and a second hotspot in eastern Australia. These data are unlikely to be representative of all bycatch hotspots as the data are driven by the presence of fishery observer programs, which are not extensive and are concentrated in certain nations fishing fleets.

From the 2023 biological opinion for the American Samoa longline fishery (NMFS 2023a), the estimated number of anticipated future interactions for leatherbacks is 48 captures, with 31 mortalities over five years.

The 2004 management measures introduced to the Hawaii longline fisheries have demonstrably reduced leatherback sea turtle interaction rates by 83% (Gilman et al. 2007a; WPRFMC 2009b). Since the Hawaii SSLL fishery re-opened in 2004, 22 leatherback sea turtles have been estimated to have been killed in the fishery. All of the leatherback sea turtles caught were released alive; mortality estimates come from applying the NMFS post-hooking mortality criteria to interactions (Ryder et al. 2006). NMFS authorized the incidental take of 21 leatherback sea turtles annually with an expectation that 3 would die (NMFS 2019a).

From 1990 to 2009, there were 24 observed leatherback turtle interactions in the California drift gillnet fishery based on 15.6 percent per year observer coverage (Martin et al. 2015). Genetic analyses indicated that almost all originated from the West Pacific DPS (Dutton et al. 1999) (NMFS SWFSC unpublished data). In 2001, NMFS implemented regulations (i.e., a large time/area closure offshore central California) that reduced interactions by approximately 80 to 90 percent (NMFS and FWS 2020a). Since bycatch reduction regulations have been in place to protect leatherbacks, only one leatherback has been observed taken in the California drift gillnet fishery, and it was released alive (NMFS West Coast observer program, unpublished data through 2021). Based on the latest biological opinion (NMFS 2013), the California and Oregon drift gillnet fishery is anticipated to result in an estimated 3 leatherback interactions mortalities per year, up to 10 interactions over a 5-year period, and up to 7 mortalities over a 5-year period.

In 2020, NMFS issued a final rule implementing a rolling 2-year hard cap of no more than two dead or injured leatherback sea turtles (as well as other sea turtle and marine mammal species). In 2021, this rule was vacated by a court order (Burke v. Coggins, 521 F. Supp. 3d 31 [D.D.C. 2021]). More recently, the Driftnet Modernization and Bycatch Reduction Act, which will phase out the use of large-mesh drift gillnets in federal waters off of California, was included in the 2023 Congressional Omnibus spending bill and became law in January 2023.

The U.S. tuna purse seine fishery operating in the Western and Central Pacific Ocean interacted with approximately 16 leatherback turtles between 2008 and 2015 based on observer coverage ranging from 20 to 100 percent (NMFS and FWS 2020a). The anticipated future interactions of leatherbacks for this fishery is estimated to be 11 sub-lethal interactions per year, and mortalities are not anticipated from this fishery.

Historically, significant leatherback bycatch was documented in the North Pacific high seas driftnet fishery, which expanded rapidly during the late 1970s and was banned in 1992 by a United Nations resolution (summarized in Benson et al. 2015). High seas driftnet fishery bycatch was likely a significant contributor to the population declines observed at nesting beaches during the 1980s and 1990s (Benson et al. 2015). Bycatch in small-scale coastal fisheries has also been a significant contributor to leatherback population declines in many regions (Alfaro-Shigueto et al. 2011; Kaplan 2005), yet there is a significant lack of information from coastal and small-scale fisheries, especially from the Indian Ocean and Southeast Asian region (Lewison et al. 2014).

In summary, West Pacific leatherbacks are exposed to high fishing effort throughout their foraging range, and likely in coastal waters near nesting beaches or en route to and from nesting beaches and foraging habitats, though very little fisheries data are available for coastal areas (NMFS and FWS 2020a). Bycatch rates in international pelagic and coastal fisheries are thought to be high, and these fisheries have limited management regulations despite hotspots of high interactions, for example in Southeast Asia (Lewison et al. 2004; Alfaro-Shigueto et al. 2011; Wallace et al. 2013b; Clarke et al. 2014; Lewison et al. 2014; Clarke 2017). Annual interaction and mortality estimates are only available for U.S.-managed pelagic fisheries, which operate under fisheries regulations that are designed to minimize interactions with and mortalities of endangered and threatened sea turtles (NMFS and FWS 2013, 2020a; Swimmer et al. 2017).

Bycatch in commercial and recreational fisheries, both on the high seas and nearshore, is considered a primary threat to the East Pacific leatherback population (NMFS and FWS 2020a). Juvenile and adult leatherbacks are exposed to high fishing effort throughout their foraging range and in coastal waters near nesting beaches. Mortality is also high in some fisheries, with reported mortality rates of up to 58 percent due in part to the use of gillnets and as well as consumption of bycaught turtles in Peru (NMFS and FWS 2020a).

Eckert and Sarti (1997) speculated that the swordfish gillnet fisheries in Peru and Chile contributed to the decline of the leatherback sea turtle in the Eastern Pacific as the decline in the nesting population at Mexiquillo, Mexico occurred at the same time that effort doubled in the Chilean driftnet fishery; although ongoing leatherback sea turtle bycatch in gillnet and longline fisheries of South America off Peru and Chile continues to impact adults and subadults (Alfaro-Shigueto et al. 2007, 2011; Donoso and Dutton 2010). An assessment of fisheries bycatch impacts on sea turtle populations globally found that bycatch in net gear (i.e. bottom-set nets, fixed nets, drift nets, etc.) appears to have the highest population-level impact on the East Pacific population, followed by longlines (Wallace et al. 2013a). Roe et al. (2014) highlights potential longline fishery bycatch hotspots in the Pacific that may affect populations at various stages of their life history. While efforts by individual nations and regional fishery management organizations have, to some extent, mitigated and reduced bycatch, this stressor remains a major threat to the East Pacific leatherback population (NMFS and FWS 2020a).

Conservation

Considerable effort has been made since the 1980s to document and address leatherback sea turtle bycatch in fisheries around the world. In the United States, observer programs have been implemented in most United States federally managed fisheries to collect bycatch data, and several strategies have been pursued to reduce both bycatch rates and post-interaction mortality. These include developing gear solutions to prevent or reduce capture (e.g., circle hooks in combination with fin-fish bait for longline fisheries) or to allow turtles to escape without harm (e.g., turtle exclusion devices in trawl fisheries), implementing seasonal time-area closures to prevent fishing when turtles are congregated, modifying existing gear (e.g., reducing mesh size of gillnets), and developing and promoting Sea Turtle Handling Guidelines (Stokes and Bergmann 2019).

For example, switching to large circle hooks and mackerel-type bait in 2004 with complimentary fishery-based outreach and education resulted in an 84% reduction in the leatherback sea turtle interaction rate in the Hawaii SSLL fishery (Swimmer et al. 2017). In addition, in 2020, NMFS issued a final rule for the SSLL that reduced the annual interaction limit from 26 to 16 for leatherbacks and included trip (not more than 2 leatherbacks per vessel trip) and vessel (vessels that reach the trip limit twice in a calendar year are prohibited from the fishery for the remainder of the year) limits (85 FR 57988). NMFS and FWS developed a 5-year action plan, identifying the top five recovery actions to support over the next five years: (1) reduce fishery interactions; (2) improve nesting beach protection and increase reproductive output; (3) international cooperation; (4) monitoring and research; and (5) public engagement (NOAA 2016). Reduced effort and time area closures of the California drift gillnet fishery have decreased leatherback captures from an average of 14.5 per year from 1990 to 2000 to an average of 0.6 per year from 2010 to 2020 (Caretta et al. 2022).

Community-based conservation projects at Wermon and Jamursba Medi beaches on the Birdshead Peninsula of Indonesia, Buru and Kei Islands within the Maluku Province of Indonesia, Isabel Island of the Solomon Islands, and Epi and Ambrym Islands of Vanuatu in the West Pacific, and in Mexico, Costa Rica and Nicaragua in the East Pacific have been developed that monitor nesting activity and protect nests and females from harvest and predation. These projects have increased our understanding of population nesting trends, and work to bolster the production of hatchlings from nesting areas and reduce harvest. Specifically, a nest monitoring project has been conducted in the Buru Islands since 2017. In 2017, 59% of leatherback nests were poached and nesting females were known to be harvested. By the 2021/2022 season, only one nest out of 181 was poached with no direct harvest of nesting females observed (WWF 2022). The Kei Island project works to reduce the harvest of subadult and adult leatherback turtles foraging within waters of the Kei Islands. In this project the direct harvest of foraging leatherbacks has declined by an average of 85.8% since 2017 (PIFSC unpublished data).

Conservation efforts at Jamursba-Medi and Wermon beaches have continued and expanded (Pakiding et al. 2020).

The conservation and recovery of leatherback sea turtles is facilitated by a number of regulatory mechanisms at international, regional, national and local levels, such as the Food and Agriculture Organization's (FAO) Technical Consultation on Sea Turtle-Fishery Interactions, the Inter-American Convention for the Protection and Conservation of Sea Turtles, CITES, and others. In 2008 the WCPFC adopted CMM 2008-03 to mitigate the impacts on turtles from shallow-set longline swordfish fisheries in the Western Central Pacific Ocean. In 2018, the WCPFC adopted CMM 2018-04 to ensure the safe handling, release, and reporting of captured sea turtles for all vessels targeting species covered by the Convention, to mitigate the impacts of the purse seine fisheries on sea turtles, and to require circle hooks and finfish bait for shallow-set longline vessels.

As a result of these designations and agreements, many intentional impacts on sea turtles have been reduced: harvest of eggs and adults have been reduced at several nesting areas through nesting beach conservation efforts (although significantly more effort is needed to reduce harvest pressure), and a number of community-based initiatives have helped reduce the harvest of turtles in foraging areas (NMFS and FWS 2013).

Summary of the Status

In this section of this biological opinion, we explained that the leatherback sea turtle, which is endangered throughout its range, is a global metapopulation represented by 7 extant subpopulations (a metapopulation in this respect is a set of discrete populations or subpopulations of the same species in which migration between those populations is significantly restricted). The Pacific Ocean portion of the metapopulation was once comprised of three spatially and genetically distinct populations. One population, Terengganu, Malaysia, is considered functionally extinct. The remaining two populations, the East Pacific and West Pacific Ocean populations, are critically endangered. Presently the East Pacific leatherback nesting is likely comprised of approximately 1,007 adult females, with only 106 to 503 females nesting annually per year at primary and secondary nesting beaches (NMFS and FWS 2020a). Populations have declined by 97.4% during the past three generations (Wallace et al. 2013b). Based on current trends, we estimate the population will continue to decline at 8.1% per year. The West Pacific Ocean leatherback sea turtle was estimated to have declined by as much as 78.3% from 1984 to 2011 (Tapilatu et al. 2013) and was estimated to be declining at a rate of approximately 6% per year through 2017 (Martin et al. 2020a). Martin et al. (2020a) estimates that there were approximately 790 adult females in 2017 at the two primary nesting beaches in Indonesia, which represent approximately 75% of the total population.

At this time, it is uncertain if the conservation efforts have been able to ameliorate the observed asymptotic declines for Pacific leatherback turtle populations and additional years of data, as well as data from newly established monitoring programs, are needed to assess the current status of these populations. For the West Pacific population, we have some data indicating that the conservation efforts are increasing hatchling production and nesting female survival.

2.2.12 North Pacific Loggerhead

Distribution and Population Structure

Loggerhead sea turtles are distributed globally with nine DPS' that are listed under the ESA (NMFS 2011). North Pacific loggerhead sea turtles, listed as endangered, are the only DPS of loggerhead sea turtle in the Action Area of the proposed action.

North Pacific loggerhead sea turtles occur north of the equator in the Pacific Ocean. Like other sea turtle DPSs, the North Pacific loggerhead sea turtle exhibits a complex life cycle: egg, hatchling, juvenile, subadult, and adult. Juvenile and subadult life stages are also frequently distinguished according to whether they occur in neritic or pelagic waters.

Almost all North Pacific loggerhead nesting occurs in Japan (Kamezaki et al. 2003), although some nesting may occur at low levels outside of Japan in areas surrounding the South China Sea (Chan et al. 2007; Conant et al. 2009). Hatchlings emerge in late summer and fall, and will eventually migrate to forage in parts of the central and eastern North Pacific Ocean using the Kuroshio and North Pacific Currents to assist their movement. Foraging North Pacific loggerhead sea turtles occur in the eastern Pacific as far south as Baja California Sur, Mexico (Seminoff et al. 2004; Peckham et al. 2007) and in the Western Pacific in the South and East China Seas and as far south as the Philippines and Malaysia (Limpus 2009; Kobayashi et al. 2008, 2011), and the mouth of Mekong River, Vietnam (Sadoyama et al. 1996). After years spent foraging and maturing, some recruit to neritic habitats and others remain in pelagic waters prior to reaching maturity. North Pacific loggerheads return to waters off their natal beaches to mate. There appears to be a meaningful substructure in the North Pacific loggerhead sea turtle such that it is comprised of at least three subpopulations. Matsuzawa et al. (2016) found significant differences in mtDNA haplotype frequencies for the DPS, and concluded that it is comprised of at least three regional management units or subpopulations represented by the three distinct nesting areas identified by:

- Ryukyu Archipelago, which includes turtles on Okinawa, Okinoerabu, and Amami islands
- Yakushima Island
- Japan Mainland, which includes Bousou, Enshu-nada, Shikoku, Kii, and Eastern Kyushu

The genetic data that define the three subpopulations are characterized by at least two common mtDNA haplotypes that occur at different frequencies but have too much overlap to differentiate in the fishery using a mixed stock analysis from the sample size from the Hawaii DSLL fishery (P. Dutton pers. comm. April 30, 2018). The Ryuku subpopulation is the smallest and the most genetically distinct. This spatial and genetic structure (NMFS and FWS 2020b) is important to the DPS' extinction risk because the genetic exchange that occurs between these three subpopulations is low, suggesting that genetic exchange among these subpopulations would be unlikely to alleviate the loss of nesting females from one subpopulation to the other.

Status and Trends

As described in Sections 1.4.2 and 2.2.11 for leatherback sea turtles, Martin et al. (2020a) used a Bayesian state-space model to estimate a long-term trend from nest count data for North Pacific loggerhead sea turtles. They used data from three index beaches in Japan, Maehama, Inakahama, and Yotsusehama, from 1985 to 2015. The nest count data was converted to nester count data by dividing the number of nests each year by the mean clutch frequency (4.6 nests per female; Hatase et al. 2013). As with West Pacific leatherback sea turtles, these trends will not necessarily represent the true growth rate of the population because annual nester counts, which represent the bulk of data on turtles, only represent a portion of the population, specifically, adult females. However, as this is the only estimate of population growth available for this DPS, we consider it the best available scientific information to describe trends.

As with leatherback sea turtles, Martin et al. (2020a) used a Bayesian inference model to estimate the range of intrinsic population growth rates or *r*. Drawing from the resulting distribution of *r* values as well as the distributions of the nesting female population size at the end of the time series and a process error term, Martin et al. (2020a) projected population trends 100 years into the future, conducting 10,000 simulations to capture the variability in projections. In the future projections, Martin et al. (2020a) computed the proportion of simulations for which the projected number of annual nesters fell below (and remained below) 50%, 25%, and 12.5% of the estimated abundance of nesters within specific time intervals. For the set of runs ending below a threshold, they calculated the mean, median, and 95% credible interval for the number of years until the population fell below the threshold. They also calculated the probability of the projected total reproductive females falling below each threshold at 5, 10, 25, 50, and 100 years in the future.

Results of the PVA model suggest that the adult female portion of the North Pacific loggerhead sea turtle population is increasing at a rate of 2.3% per year (95% CI: -1.1 to 15.6%), and the probability of the population as indicated by the index beaches falling to less than half of its current abundance within 100 years is 33% (Martin et al. 2020a). For those simulations in the PVA that indicated a decline, 50% of current population size was reached in a mean of 25.2 years (95% CI 5 to 82 years). PVA modeled estimates suggest the modeled population presently consists of a minimum of 4,541 adult female loggerhead sea turtles (95% CI: 4,074-5,063). It is estimated that there are approximately 328,744 juvenile (year 1-25) North Pacific loggerhead sea turtles (T. Jones pers. comm. 2019). Using the estimate of 4,541 females nesting in Yakushima which represents 52% of nesting females, the total number of North Pacific loggerhead nesting females is 8,733 (4,541*100/52). Using a sex ratio of 65% female (Martin et al. 2020a) suggests the DPS is comprised of approximately 13,435 (8,733*100/65) adults, or a total population size of approximately 342,179 (328,744+13,435.

Population Dynamics

The population dynamics of loggerhead sea turtles, like all of the other sea turtles this biological opinion considers, are usually based on adult females, rather than their male counterparts or the entire population. The spatial structure of male sea turtles and their fidelity to specific coastal areas is unknown; however, we discuss sea turtle populations based on the nesting beaches that female sea turtles return to when they mature. Because the patterns of increase or decrease in the abundance of sea turtle nests over time are determined by internal dynamics rather than external dynamics, we make inferences about the growth or decline of sea turtle populations based on the status and trend of their nests.

The age at which loggerhead sea turtle mature is uncertain. Several authors placed the age of first reproduction at 25 years (Conant et al. 2009; Jones et al. 2012, 2018) while Turner-Tomaszewicz et al. (2015) placed the age of first maturity at between 3 and 24 years (based on skeletochronological analysis). In a reanalysis of the Turner-Tomaszewicz et al. (2015) data, Martin et al. (2020a) estimate 37.9 years to maturity. Remigration intervals vary, but range from about 2 to 4 years in Pacific populations. Females are expected to lay eggs for 1 to 12 years (mean = 4), breed between 1 and 5 times per season, and produce 1 to 5 clutches. Clutch sizes range from about 78 to 135 eggs, with emergence success ranging from 43 to 80%.

Diving and Social Behavior

Tagging studies indicate that juvenile loggerhead sea turtles are shallow divers, less than 100 m, spending 40–80% of time at surface and 90% of time at depths less than 15 m (Howell et al. 2010). Narazaki et al. (2015) obtained approximately 19,500 (10,000 in summer and 9,500 in winter) dive profiles from six turtles off the East coast of Japan. Results showed the median daily dive depth ranged from 6 to 24 m, with occasional deep dives from 20 to 65 m (Narazaki et al. 2015). Furthermore, daily dive profiles were influenced by season and water temperatures with the deepest dive profiles, by all individuals, reported in winter (Narazaki et al. 2015). Maximum depths were reported over 340 m deep for two of the six turtles (Narazaki et al. 2015).

Threats to the Species

The various habitat types loggerhead sea turtles occupy throughout their lives exposes these sea turtles to a wide variety of natural and anthropogenic threats. The beaches on which loggerhead sea turtles’ nest and the nests themselves are threatened by hurricanes and tropical storms as well as the storm surges, sand accretion, and rainfall that are associated with hurricanes. For example, in 1992, all of the eggs over a 90-mile length of coastal Florida were destroyed by storm surges on beaches that were closest to the eye of Hurricane Andrew (Milton et al. 1994). Hatchlings are hunted by predators like herons, gulls, dogfish, and sharks. Adult loggerhead sea turtles are also killed by sharks and other large, marine predators. Loggerhead sea turtles are also killed by cold stunning and exposure to biotoxins.

A wide variety of human activities adversely affect hatchlings and adult female turtles when they are on land, including beach erosion, beach armoring and nourishment; artificial lighting; beach cleaning; human presence on nesting beaches; beach driving; coastal construction and fishing piers that alter patterns of erosion and accretion on nesting beaches; exotic dune and beach vegetation; and poaching. As the size of the human population in coastal areas increases, that population brings with it secondary threats such as exotic fire ants, feral hogs, dogs, native species that tolerate human presence (*e.g.*, raccoons, armadillos, and opossums) which feed on turtle eggs.

When they are in coastal or marine waters, loggerhead turtles are affected by a completely different set of human activities that include discharges of toxic chemicals and other pollutants into the marine ecosystem; underwater explosions; hopper dredging, offshore artificial lighting; entrainment or impingement in power plants; entanglement in marine debris; ingestion of marine debris; boat collisions; poaching, and interactions with commercial fisheries. Of these, interactions with fisheries represents a primary threat because of the number of individuals that are captured and killed in fishing gear each year (Conant et al. 2009).

Loggerhead sea turtles are also captured and killed in commercial fisheries. In the Pacific Ocean, between 2,600 and 6,000 loggerhead sea turtles were estimated to have been captured and killed in longline fisheries in 2000 (Lewison et al. 2004). The Hawaii-based longline fisheries are estimated to have captured and killed several hundred loggerhead sea turtles before the shallow set portion of the fishery was closed in 2001. When it was re-opened in 2004, with substantial modifications to protect sea turtles, these fisheries were estimated to have captured and killed fewer than 5 loggerhead sea turtles each year as a result of those mitigation measures. Between 2004 and 2018, the SSLL was estimated to have captured about 177 North Pacific loggerhead sea turtles, killing about 28 of these sea turtles (NMFS 2019a). In the West Coast DSLL fishery, NMFS exempted one interaction and associated mortality per 10-year period (NMFS 2016a).

Two exempted fishery permits were authorized in this fishery exempting take for one additional interaction and associated mortality over a 2-year period. However, as noted in 85 FR 48205, "issuance of this EFP was followed by litigation in the United States District Court for the Northern District of California, in which the plaintiffs alleged, among other claims, that NMFS had not used the best scientific information available in its NEPA analysis or ESA section 7 consultation, and that as a result, NMFS' issuance of the EFP violated both NEPA and the ESA. *Center for Biological Diversity, et al.* v. *Ross, et al.*, 4:19-cv-03135-KAW (N.D. Cal.). On December 20, 2019, the Court ruled in favor of the plaintiffs, and vacated and set aside the EFP, EA and FONSI, and 2018 Biological Opinion. No SSLL or DSLL fishing activity has occurred within the West Coast EEZ under the EFP since the Court's ruling." These are the only two United States fisheries expected to overlap the Action Area which may have effects to the North Pacific loggerhead other than this fishery.

However, a number of other United States fisheries outside of the Action Area are expected to effect the North Pacific loggerhead sea turtle. NMFS authorized the incidental take of up to 36 North Pacific loggerhead sea turtles annually with an expectation that 6 of those turtles would die in the United States WCPO purse seine fishery (NMFS 2021a). Additionally, 30 interactions are estimated to occur in the Eastern Tropical Pacific purse seine fishery, with 1 mortality expected every 7 years (NMFS 1999, 2004b). Lastly, in the California/Oregon drift gillnet fishery, NMFS exempted take for 3 interactions and 2 mortalities over any 1-year period, or 7 interactions with 4 mortalities over 5 years (NMFS 2013).

Conservation

Considerable effort has been made since the 1980's to document and reduce loggerhead bycatch in Pacific Ocean fisheries, as this is the highest conservation priority for the DPS. NMFS has formalized conservation actions to protect foraging loggerheads in the North Pacific Ocean which were implemented to reduce loggerhead bycatch in United States fisheries. Observer programs have been implemented in federally managed fisheries to collect bycatch data, and several strategies have been pursued to reduce both bycatch rates and post-hooking mortality. These include developing gear solutions to prevent or reduce capture (e.g., circle hooks) or to allow the turtle to escape without harm (e.g., turtle exclusion devices), implementing seasonal time-area closures to prevent fishing when turtles are congregated, modifying existing gear, and developing and promoting "Sea Turtle Handling Guidelines" (NMFS and FWS 2007a).

For example, switching to large circle hooks and mackerel bait in 2004 reduced the interaction rate by approximately 90% in the Hawaii shallow-set longline fishery (Gilman et al. 2007a; WPRFMC 2009b). In 2003, NMFS implemented a time/area closure in southern California during forecasted or existing El Niño-like conditions to reduce the take of loggerheads in the California/Oregon drift gillnet fishery (68 FR 69962; December 16, 2003). NMFS has also developed a mapping product known as TurtleWatch that provides a near real time product that recommends areas where the deployment of pelagic longline shallow sets should be avoided to help reduce interactions between Hawaii-based pelagic longline fishing vessels and loggerhead sea turtles (Howell et al. 2008, 2015) and implemented trip limits in the SSLL fishery (see NMFS 2019a).

Since loggerhead interactions and mortalities with coastal fisheries in Mexico and Japan are of concern and are considered a major threat to North Pacific loggerhead recovery, NMFS and United States non-governmental organizations have worked with international entities to: (1) assess bycatch mortality through systematic stranding surveys in Baja California Sur, Mexico; (2) reduce interactions and mortalities in bottom-set fisheries in Mexico; (3) conduct gear mitigation trials to reduce bycatch in Japanese pound nets; and (4) convey information to fishermen and other stakeholders through participatory activities, events and outreach. In 2003, Grupo Tortuguero's ProCaguama (Operation Loggerhead) was initiated to partner directly with fishermen to assess and mitigate their bycatch while maintaining fisheries sustainability in Baja California, Mexico.

ProCaguama's fisher-scientist team discovered the highest turtle bycatch rates documented worldwide and has made considerable progress in mitigating anthropogenic mortality in Mexican waters (Peckham et al. 2007, 2008). As a result of the 2006 and 2007 tri-national fishermen's exchanges run by ProCaguama, Sea Turtle Association of Japan (STAJ), and the Western Pacific Fisheries Management Council, in 2007 a prominent Baja California Sur fleet retired its bottom-set longlines (Peckham et al. 2008; Peckham and Maldonado-Diaz 2012).

Prior to this closure, the longline fleet interacted with an estimated 1,160-2,174 loggerheads annually, with nearly all (89%) of the takes resulting in mortalities (Peckham et. al. 2008). Because this fleet no longer interacts with loggerheads, conservation efforts have resulted in the continued protection of approximately 1,160-2,174 juvenile loggerheads annually (76 FR 58868; September 22, 2011). Additionally, stranding data collected since 2003 at Playa San Lazaro indicates a 60% reduction in standings' during 2010 compared to previous 2003-2009 averages (Peckham 2010). To date, 90% of the gillnet fleet has retired their gear (a total of 140 gillnets), 18 crews have converted to hook and line fishing (a more sustainable practice in the 'hotspot' area), and local government enforcement has increased to ensure compliance with local laws (Peckham pers. comm.).

In Japan, due to concerns of high adult loggerhead mortality in mid-water pound nets, researchers with the STAJ, ProCaguama, and NMFS have begun collaborations, together with local fishermen throughout several Japanese prefectures, to investigate and test pound net mitigation options to reduce the impact and mortality of sea turtle bycatch. This work is ongoing as of 2011 and has received high media attention both within Japan and internationally that has helped to raise public awareness and maintain momentum (Ishihara et al. 2014).

Led by the Mexican Wildlife Service, a federal loggerhead bycatch reduction task force, comprised of federal and state agencies and non-governmental organizations, was organized in 2008 to ensure loggerheads receive the protection they are afforded by Mexican law. In 2009, while testing a variety of potential solutions, ProCaguama's fisher-scientist team demonstrated the commercial viability of substituting bycatch-free hook fishing for gillnet fishing. ProCaguama, in coordination with the task force, is working to develop a market-based bycatch solution consisting of hook substitution, training to augment ex-vessel fish value, development of fisheries infrastructure, linkage of local fleets with regional and international markets, and concurrent strengthening of local fisheries management (Conant et al. 2009).

Conservation efforts have also focused on protecting nesting beaches, nests, and hatchlings. Much of Japan's coastline is "armored" using concrete structures to prevent and minimize impacts to coastal communities from natural disasters. These structures have resulted in a number of nesting beaches losing sand suitable for sea turtle nesting, and nests often need relocating to protect them from erosion and inundation. In recent years, a portion of the concrete structures at a beach in Toyohashi City, Aichi Prefecture, was experimentally removed to create better nesting habitat (76 FR 58868; September 22, 2011). The STAJ, along with various other organizations in Japan, are carrying out discussions with local and Federal Government agencies to develop further solutions to the beach erosion issue and to maintain viable nesting sites. Recently, the Ministry of Environment has supported the local NGO conducting turtle surveys and conservation on Yakushima in establishing guidelines for tourism to minimize impacts by humans on nesting beaches (Y. Matsuzawa, STAJ, personal communication; Conant et al. 2009). Yet, beach erosion and armament still remain one of the most significant threats to nesting beaches in Japan (Conant et al. 2009).

From 2003 to 2013, the Council contracted with STAJ to protect loggerhead nests and increase hatchling survivorship at several nesting beaches in southern Japan, including at the two primary beaches on Yakushima Island. Beach management activities include conducting nightly patrols during the summer nesting season to relocate nests from erosion prone areas, protecting nests from predators and people with mesh and fences, and cooling nests with water and shading to prevent overheating during incubation. STAJ has developed techniques for nest relocation that now result in an average of 60% hatchling success rates (compared to nearly zero survival of the same nests laid in erosion prone areas). Nest relocation in 2004-08 resulted in an estimated 269,451 hatchlings being released that otherwise may have been lost (Ishizaki 2015; WPRFMC 2015).

The conservation and recovery of loggerhead turtles is facilitated by a number of regulatory mechanisms at international, regional, national, and local levels, such as the Food and Agriculture Organization's (FAO) Technical Consultation on Sea Turtle-Fishery Interactions, the Inter-American Convention for the Protection and Conservation of Sea Turtles (IAC), the Convention on International Trade in Endangered Species (CITES), and others. In 2008, the WCPFC adopted CMM 2008-03 to mitigate the impacts on turtles from longline swordfish fisheries in the western central Pacific Ocean. The measure includes the adoption of FAO guidelines to reduce sea turtle mortality through safe handling practices and to reduce bycatch by implementing one of three methods by January 2010. The three methods to choose from are: 1) use only large circle hooks, or 2) use whole finfish bait, or 3) use any other mitigation plan or activity that has been approved by the Commission.

CMM 2018-04 further enhanced mitigation measures to protect sea turtles by improving survivorship, handling requirements, and reporting standards for participating countries. As a result of these designations and agreements, many of the intentional impacts on sea turtles have been reduced: harvest of eggs and adults have been slowed at several nesting areas through nesting beach conservation efforts and an increasing number of community-based initiatives are in place to slow the take of turtles in foraging areas. Moreover, as shown by the above examples from Hawaii, Japan, and Baja Mexico, international efforts are growing to reduce sea turtle interactions and mortality in artisanal and industrial fishing practices (Gilman et al. 2007b; Peckham et al. 2007; NMFS and FWS 2007a; Ishihara et. al. 2014).

Summary of the Status

In this section of this biological opinion, we explained that the North Pacific loggerhead sea turtle is endangered, and that the DPS' trend appears to be increasing in abundance. We used our knowledge of the DPS' demography and population ecology to capture the primary factors that appear to determine the loggerhead sea turtle's population dynamics.

While this DPS faces both natural and anthropogenic threats at all life cycle stages, the number of recruits into the population appears to balance the loss from the various stressors.

2.2.13 Olive Ridley Sea Turtle

Distribution and Population Structure

Olive ridley turtles occur in the tropical waters of the Pacific and Indian Oceans from Micronesia, Japan, India, and Arabia south to northern Australia and southern Africa (NMFS and FWS 2014). In the eastern Pacific Ocean, olive ridley turtles are found from the Galapagos Islands to California. They are found in coastal waters of over 80 countries (Abreu-Grobois and Plotkin 2008).

In the eastern Pacific, olive ridley sea turtles are highly migratory and appear to spend most of their nonbreeding life cycle in the oceanic zone (Cornelius and Robinson-Clark 1986; Arenas and Hall 1992; Pitman 1991, 1993; Plotkin 1994, 2010; Plotkin et al. 1994, 1995; Beavers and Cassano 1996). Polovina et al. (2003, 2004) tracked 10 olive ridley sea turtles caught in the Hawaii pelagic longline fishery. The olive ridley sea turtles identified as originating from the eastern Pacific populations stayed south of major currents in the central North Pacific-southern edge of the Kuroshio Extension Current, North Equatorial Current, and Equatorial Counter Current. Olive ridley sea turtles identified from the Western Pacific were associated with these major currents, suggesting that olive ridley sea turtles from different populations may occupy different oceanic habitats (Polovina et al. 2003, 2004).

Population structure and genetics are poorly understood for this species, but populations occur in at least the eastern Pacific, western Pacific, eastern Indian Ocean, central Indian Ocean, western Indian Ocean, West Africa, and the western Atlantic (NMFS and FWS 2014). Several studies found moderate to high genetic differentiation among regional rookeries separated by more than 500 km, but low differentiation for rookeries in closer proximity (e.g., Suriname and French Guiana: Hahn et al. 2012; Northern Territory, Australia: Hahn et al. 2012; Jensen et al. 2013). However, other studies found little genetic differentiation between rookeries over larger areas. In the Indian Ocean, Shanker et al. (2004) detected no population subdivision along 2,000 km of east India coastline. In the east Pacific Ocean, rookeries in Costa Rica and Mexico, separated by more than 500 km, were not genetically distinct (Bowen et al. 1998; Hahn et al. 2012). Wallace et al. (2010a) identified two distinct regional management units in the east Pacific (arribada and solitary nesters) based on a meta-analysis of genetic and other relevant data on olive ridley life history and biogeography. They believed the arribada and solitary nesting assemblages warranted separate management considerations, given there were differences in genetic diversity, trends, and abundance between the two types of nesting behaviors.

Based on the genetic samples from olive ridley sea turtles that have interacted with the DSLL fishery we expect that individuals from both the endangered Mexico population and the threatened subpopulation in the eastern Pacific as well as the threatened subpopulation in the

Western Pacific interact with the fishery. Results of this genetic analysis indicate that approximately 75% of olive ridley sea turtles interacting with the fishery are from either the endangered Mexico population or threatened subpopulation in the eastern Pacific, and 25% are from the threatened Western Pacific subpopulation of olive ridley sea turtles (NMFS unpublished data).

Status and Trends

Olive ridley sea turtle breeding populations on the Pacific coast of Mexico are listed as endangered under the ESA; all other populations are listed as threatened. The IUCN has classified the olive ridley turtle as "vulnerable" (Abreu-Grobois and Plotkin 2008).

Where population densities are high enough, nesting takes place in synchronized aggregations known as arribadas. The largest known arribadas in the eastern Pacific are off the coast of Costa Rica (~475,000 - 650,000 females estimated nesting annually) and in southern Mexico (~1,000,000 nesting females per year at La Escobilla, in Oaxaca, Mexico [Abreu-Grobois and Plotkin 2008; Valverde et al. 2012; NMFS and FWS 2014]). The population size of the endangered Pacific coast of Mexico population is therefore a minimum of 1,000,000 individuals (NMFS and FWS 2014). All but one of the nesting beaches discussed in NMFS and FWS (2014) have stable to increasing populations (seven stable, four increasing, one declining and three unknown). Therefore, we consider the overall trend for the endangered Mexico olive ridley sea turtle to be at least stable.

At the nesting site in Ostional, Costa Rica, the average arribada size in the main nesting beach increased from 75,000 turtles in 1980 to 125,000 turtles in 2003 (Chaves et al. 2005 as cited in NMFS and FWS 2014). The number of arribadas per year ranged from 7 to 16 and averaged 11.17 ± 2.29 (Chaves et al. 2005 as cited in NMFS and FWS 2014). From 2006-2010, arribadas ranged from 3,564 to 476,550 nesting females using the Valverde and Gates method (Valverde et al. 2012). Trends are considered to be increasing at this nesting site (NMFS and FWS 2014).

In Nicaragua, two arribada beaches still exist (Ruiz 1994): Playa Chacocente (located in the Chacocente Wildlife Refuge) and Playa La Flor (located in a private wildlife refuge; NMFS and FWS 2014). Population trends for Playa Chacocente are unknown. The nesting population at Playa La Flor is thought to be depleted although the most recent census suggest 521,440 annual nesting females, with stable population trends (Abreu-Grobois and Plotkin 2008). Overall, there are at least 1,000,000 nesting females in the threatened population in the East Pacific and therefore we consider this a minimum population estimate. While there are smaller nesting sites with declining trends, given that the two largest nesting sites are either stable or increasing, we consider the overall trend to be at least stable for threatened East Pacific olive ridley sea turtles.

In the western Pacific, information on the size of olive ridley nesting aggregations and their trends are limited (NMFS and FWS 2014). In Indonesia, olive ridley sea turtles nest on beaches in the West Papua Province, on Jamursba Medi beach, the number of nests from 1993 to 2001 increased from 4 to 227 (Hitipeuw and Maturbongs 2002). Mazaris et al. (2017) estimate the trend over that time period to be 47.3% per year although some of this increase may have been due to increased survey effort. In the Manokwari region the number of nests recorded from 2008 through 2011 ranged from 53 to 236, however survey effort was limited and likely not consistent across years (Suganuma et al. 2012).

Olive ridley sea turtle nesting has been monitored at Alas Purwo National Park, East Java, Indonesia since 1983 where they have increased from less than 10 annual nesters to more than 1,100 in 2018 (Kurniawan and Gitayana 2020; Figure 27). We digitized Figure 1 from Kurniawan and Gitayana (2020; Figure 27 below) and estimate the trend at 9.8% per year (95% CI: 8.1 to 11.6%). Assuming a 2 year remigration interval (van Buskirk and Crowder 1994) and 50% sex ratio (Maulany et al. 2017), the total adults for this location is approximately 4,400. On Hamadi beach, Jayapura Bay in June 1999, an estimated several hundred ridleys were observed nesting (NMFS and FWS 2014). On Buru Island, Indonesia leatherback and olive ridley sea turtles have recently been discovered nesting, with 211 olive ridley sea turtle nests identified in 2017 (WWF 2018).

In eastern Java, olive ridley sea turtle nesting was documented from 1992-1996 that ranged from 101 to 169 nests. In Australia, olive ridley sea turtle nesting is scattered throughout northern Australia, with a few thousand females nesting annually (Limpus 2008). The breeding population in northern Australia may be the largest population remaining in the Western Pacific region, although a full evaluation of their distribution and abundance is needed (Limpus 2008; NMFS and FWS 2014). The most recent recovery plan for sea turtles in Australia indicates more than 5,000 olive ridley females nest each year in northern Australia (Commonwealth of Australia 2017) suggesting that nest numbers are likely at least stable and potentially increasing for those beaches. We balance the available information on nest trends with the acknowledgment that olive ridley turtles in Indonesia and other areas continue to face hunting and egg collection pressures (Tapilatu et al. 2017) and assume that West Pacific olive ridley sea turtle population trends are stable.

Olive ridleys have a mean remigration interval of about 2 years (Pritchard 1969; Cornelius 1986; Plotkin 1994; Pandav et al. 2000; Whiting et al. 2007a), therefore we estimate a minimum 10,000 nesting females for the western Pacific Ocean, primarily represented by the breeding aggregation in Australia. Sex ratios are not known for this population, assuming a 50% sex ratio we estimated the adult population at 20,000. Combining this nesting population with the Alas Purwo National Park, East Java, Indonesia nesting population suggests a minimum adult population size of 24,400 for Indo-West Pacific olive ridley sea turtles. Maulany et al. (2017) conducted a PVA for the Alas Purwo National Park, Indonesia nesting aggregation using population metrics for models developed for Kemp's ridley (Coyne 2000, Heppell et al. 2005). From these they determined that 4.4% of the total population are breeding adults and, based on Heppell et al. (2005), excluding 0 to 1 year old hatchlings, 7.7% of the total population are breeding adults.

Based on the 4.4% we estimate a total population size of 554,545, and of those, 316,883 are juveniles and adults older than 1 year old. Therefore, eliminating 0 to 1 year old hatchlings as this size class is not captured in the DSLL, we estimate the population size to be approximately 316,883 juveniles and adults based on an adult population size of 24,400 with 7.7% of the population occurring in that stage (24,400/[7.7/100]).

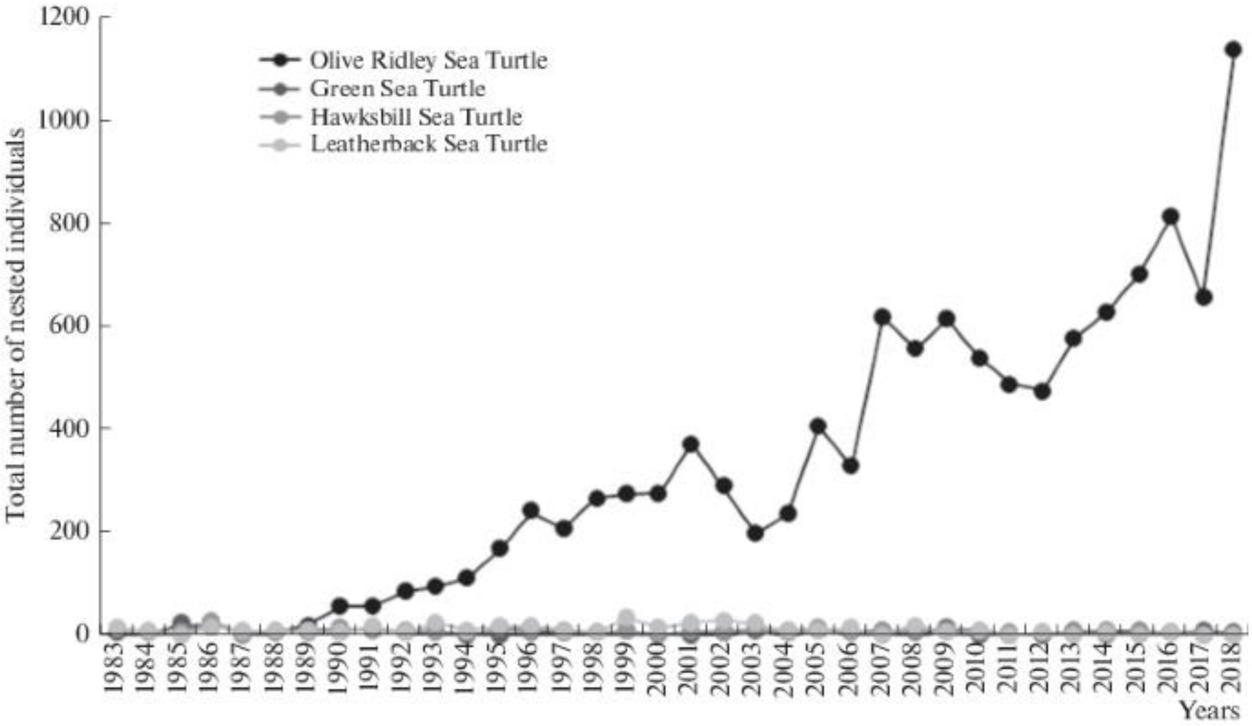

Figure 27. Trends in annual nesting sea turtles at Alas Purwo National Park, East Java, Indonesia (Kurniawan and Gitayana 2020).

Population Dynamics

Mature males measured at breeding grounds range in size from 63.3-67.1 cm SCL (Plotkin et al. 1996; Shanker et al. 2003). Measurements of nesting females range from 58.5 to 70 cm SCL (Plotkin and Bernando 2003; Shanker et al. 2003; Whiting et al. 2007b). Zug et al. (2006) estimate age at sexual maturity as approximately 13 years old at 60 cm SCL (range 10 to 18 years) and that the minimum size of maturity is 53 cm SCL. Significant growth spurts are identified at approximately 10 to 12 years old with somatic growth leveling out around 15 years of age (Zug et al. 2006).

Consecutive-year nesting is common in olive ridley sea turtles, however remigration intervals up to 8 years also occur and the overall mean remigration interval is about 2 years (Pritchard 1969; Cornelius 1986; Plotkin 1994; Pandav et al. 2000; Whiting et al. 2007a), although ranges from 1 to 8-year remigration intervals have been reported (Da Silva et al. 2007; Tripathy and Pandav 2007). In general, individual olive ridley sea turtles may nest one, two, or three times per season but on average two clutches are produced annually, with approximately 100-110 eggs per clutch (Pritchard and Plotkin 1995). However, smaller females may produce fewer eggs per clutch (Harfush et al. 2008). Generation length has not been determined for the species but is estimated at 20 years by Abreu-Grobois and Plotkin (2008).

Individual olive ridley sea turtles exhibit three different reproductive behaviors: mass or arribada nesting, dispersed or solitary nesting, and a mixed strategy of both (Kalb 1999; Bernardo and Plotkin 2007; Fonseca et al. 2013). Reproductive characteristics may differ between arribada and solitary nesters. Multiple paternity (i.e., more than one male fertilizing eggs in a clutch) was significantly greater in nests from arribada beaches, which may be attributed to population size and the associated increase in male encounter rates (Jensen et al. 2006). At Nancite Beach, Costa Rica, arribada nesters produced significantly larger clutches (i.e., more eggs) compared to solitary nesters, although other characteristics such as female size, egg size, or within-clutch variability in egg size, were not different between the groups (Plotkin and Bernardo 2003).

Olive ridley sea turtles exhibit temperature-dependent sex determination, and warmer incubation temperatures produce more females (reviewed by Wibbels 2003, 2007). The middle third of the incubation period is when the developing embryo's sex determination is sensitive to temperatures (Merchant-Larios et al. 1997). The temperature at which a nest will produce 50% males/females was estimated to be 29.95°C for nesting populations in Mexico (Sandoval Espinoza 2011 as cited in Hernandez-Echeagaray et al. 2012), approximately 30-31°C for nesting populations in Costa Rica, and less than 29°C in Gahirmatha, India (reviewed by Wibbels 2007). Pivotal temperatures likely vary within and among populations and generalizations should be applied with caution. Studies on sex ratios of olive ridley hatchlings are few and non-existent for juvenile and adults. Hernandez-Echeagaray et al. (2012) found a slight female-bias sex ratio (55%) for the 2010- 2011 nesting season at La Escobilla, Mexico. Sex ratios may also change over the nesting season. In Mexico, a female-biased hatchling sex ratio was found at most nest sites (La Escobilla was not included) at the beginning of the nesting season, and a male-biased ratio at the end of the season (Sandoval-Espinoza 2011 as cited in Hernandez-Echeagaray et al. 2012).

Nest success varies in time and space. On solitary nesting beaches, where density-dependent mortality is not a factor, hatching rates are significantly higher (Gaos et al. 2006; Dornfeld and Paladino 2012). Conversely, survivorship is low on high density arribada nesting beaches because of density-dependent mortality (Cornelius et al. 1991).

Diving and Social Behavior

Although olive ridley turtles are probably surface feeders, they have been caught in trawls at depths of 80-110 meters (NMFS and FWS 1998), and a post-nesting female reportedly dove to a maximum depth of 290 meters. The average dive length for an adult female and adult male is reported to be 54.3 and 28.5 minutes, respectively (Plotkin 1994).

Threats to the Species

Impacts from climate change, especially due to global warming, are likely to become more apparent in future years (NMFS and FWS 2014; IPCC 2022). Based on the available information, climate change will affect olive ridley habitat and biology. Based on substantial new evidence, observed changes in marine systems are associated with rising water temperatures, as well as related changes in ice cover, salinity, oxygen levels, and circulation. These changes include shifts in ranges and changes in algal, plankton, and fish abundance (IPCC 2018, 2022), which could affect olive ridley prey distribution and abundance. However, olive ridleys in the east Pacific Ocean are highly vagile, and seemingly adaptable to fluctuating environmental conditions. They possess the ability to shift from an unproductive habitat to one where the waters are biologically productive, which may minimize the impacts of climate change (Plotkin 1994, 2010). As with leatherback turtles nesting in the eastern Pacific, olive ridley sea turtles may also be affected by the occurrence of El Nino events. It is possible that the variation in numbers of turtles in the Ostional arribadas are also affected by changes in productivity in their foraging areas, because olive ridley females also need time to amass sufficient nutrients to support their metabolic, migratory, and reproductive activities (Valverde et al. 2012).

The various habitat types olive ridley sea turtles occupy throughout their lives exposes these sea turtles to a wide variety of natural threats. The beaches on which olive ridley sea turtles nest and the nests themselves are threatened by hurricanes and tropical storms as well as the storm surges, sand accretion, and rainfall that are associated with hurricanes. Predators like herons, gulls, dogfish, and sharks hunt hatchlings. Sharks and other large marine predators also kill adult olive ridley sea turtles. In addition, some female olive ridley turtles nesting in Costa Rica have been found afflicted with the fibropapilloma disease (Aguirre et al. 1999). The main egg predators were domestic dogs and vultures (*Coragyps atratus* and *Cathartes aura*; Ruiz 1994).

In India, uncontrolled mechanized fishing in areas of high sea turtle concentration, primarily illegally operated trawl fisheries, has resulted in large-scale mortality of adult olive ridley turtles during the last two decades. Since 1993, more than 50,000 olive ridleys have stranded along the coast, at least partially because of near-shore shrimp fishing (Shanker and Mohanty 1999). Fishing in coastal waters off Gahirmatha was restricted in 1993 and completely banned in 1997 with the formation of a marine sanctuary around the rookery. However, mortality due to shrimp trawling reached a record high of 13,575 ridleys during the 1997-1998 season and none of the approximately 3,000 trawlers operating off the Orissa coast use turtle excluder devices in their nets despite mandatory requirements passed in 1997 (Pandav and Choudhury 1999).

Historically, an estimated 10 million olive ridleys inhabited the waters in the eastern Pacific off Mexico (Cliffton et al. 1982 as cited in NMFS and FWS 1998d). However, human-induced mortality caused this population to decline. From the 1960s to the 1970s, several million adult olive ridleys were harvested by Mexico for commercial trade with Europe and Japan (NMFS and FWS 1998d). Although olive ridley meat is palatable, it was not widely sought after; its eggs, however, are considered a delicacy. Fisheries for olive ridley turtles were also established in Ecuador during the 1960s and 1970s to supply Europe with leather (Green and Ortiz-Crespo 1982).

The nationwide ban on commercial harvest of sea turtles in Mexico, enacted in 1990, has improved the situation for the olive ridley. Surveys of important olive ridley nesting beaches in Mexico indicate increasing numbers of nesting females in recent years (Marquez et al. 1996; Cornelius et al. 2007). Annual nesting at the principal beach, Escobilla Beach, Oaxaca, Mexico, averaged 138,000 nests prior to the ban, and since the ban on harvest in 1990, annual nesting has increased to an average of 525,000 nests (Salazar et al. 2000). At a smaller olive ridley nesting beach in central Mexico, Playon de Mismalayo, nest and egg protection efforts have resulted in more hatchlings, but the population is still seriously decremented and is threatened with extinction (Cornelius et al. 2007).

Olive ridley nests in Indonesia are subject to extensive hunting and egg collection (Cornelius et al. 2007). In combination with rapid rural and urban development, these activities have reduced the size of the nesting population in the region as well as their nesting success.

The following United States fisheries effect olive ridley sea turtles: ASLL, DSLL, SSLL, WCPO purse seine fishery, and the California Oregon drift gillnet. The ASLL fishery interacts with an estimated 27 olive ridley sea turtles or less each year (McCracken 2019b) and approximately two thirds of those are likely to be mortalities (NMFS unpublished data). Based on the genetics information from the ASLL fishery, those turtles are likely from both the Eastern and Western

Pacific. Olive ridley sea turtles that have interacted with the ASLL fishery have haplotypes consistent with both eastern Pacific (40%) and Western Pacific (60%) olive ridley sea turtles (NMFS unpublished data).

Since the SSLL fishery re-opened in 2004 there have been 10 observed interactions that have resulted in a hooking or entanglement and all animals were released alive. Based on the 10 olive ridley sea turtles that were caught by the SSLL fishery, NMFS predicts that the Hawaii SSLL fishery will interact with up to five olive ridley sea turtles per year, (or up to 11 over a 3-year period; NMFS 2019a). One of these olive ridley sea turtles may die in any year (NMFS 2019a).

The United States WCPO purse seine fishery is expected to interact with fewer than 7 olive ridley sea turtles per year, and up to 1 of those are expected to die from their interaction (NMFS 2021a). The California Oregon drift gillnet fishery has an incidental take statement for up to 2 anticipated olive ridley interactions and 1 anticipated estimated mortality every five years (NMFS 2013). Since 2001 no olive ridley sea turtles have been captured in the California Oregon drift gillnet fishery and only one has been observed since 1990 (NMFS unpublished data).

Historically, 212 olive ridleys were observed captured in the DSLL fishery from 2004 to 2021. When adjusted to account for the percentage of observer coverage, about 1,055 olive ridley turtles are likely to have been captured in the fishery in the 18-year period from 2004 to 2021 (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a). About 150 of the observed captures would have represented olive ridley sea turtles from the eastern Pacific nesting aggregations (or 740 when adjusted for observer coverage) and 55 would have been from Western Indo-Pacific nesting aggregations (or 270 when adjusted for observer coverage). About 92.2% (95% CI = 87.6 to 95.1%) of olive ridley turtles captured in the DSLL fishery have been dead when they were hauled back to the fishing vessel.

Conservation

The conservation and recovery of olive ridley sea turtles is facilitated by a number of regulatory mechanisms at international, regional, national and local levels, such as the FAO Technical Consultation on Sea Turtle-Fishery Interactions, the IAC for the Protection and Conservation of Sea Turtles, CITES, and others. As an Appendix I species under CITES, the international trade of skins from the species which fueled the large-scale commercial exploitation of the Olive Ridley from the 1960s into the 1980s was effectively halted. In 2008 the WCPFC adopted CMM 2008-03 to mitigate the impacts on turtles from longline swordfish fisheries in the Western Central Pacific Ocean. In 2018, the WCPFC adopted CMM 2018-04 to further mitigate the impacts of purse seine and longline fisheries on sea turtles.

Summary of the Status of the Olive Ridley Sea Turtle

In this section of this biological opinion, we explained that Mexico's Pacific coast breeding population is listed as endangered, and all other populations are listed as threatened. We used our knowledge of the species' demography and population ecology to capture the primary factors that appear to determine the olive ridley sea turtle population dynamics.

As previously mentioned, sea turtles face various threats throughout each stage of their respective life cycles. While this species faces both natural and anthropogenic threats at all life cycle stages, the number of recruits into the population appears to balance the loss from the various stressors.

2.2.14 Sperm Whale

Distribution and Population Structure

Sperm whales occur in every ocean except the Arctic Ocean. Sperm whales are found throughout the North Pacific and are distributed broadly from tropical and temperate waters to the Bering Sea as far north as Cape Navarin. Mature, female, and immature sperm whales of both sexes are found in more temperate and tropical waters from the equator to around 45˚ N throughout the year. These groups of adult females and immature sperm whales are rarely found at latitudes higher than 50˚ N and 50˚ S (Reeves and Whitehead 1997). Sexually mature males join these groups throughout the winter.

During the summer, mature male sperm whales are thought to move north into the Aleutian Islands, Gulf of Alaska, and the Bering Sea. Several authors have proposed population structures that recognize at least three sperm whale populations in the North Pacific for management purposes (Kasuya 1991; Bannister and Mitchell 1980). At the same time, the International Whaling Commission's (IWC) Scientific Committee designated two sperm whale stocks in the North Pacific: a western and eastern stock or population (Donovan 1991). The line separating these populations has been debated since their acceptance by the IWC's Scientific Committee. For stock assessment purposes, NMFS recognizes three discrete population centers of sperm whales in the Pacific: (1) Alaska, (2) California-Oregon-Washington, and (3) Hawaii.

Sperm whales are widely distributed throughout the Hawaiian Islands throughout the year and are the most abundant large whale in waters off Hawaii during the summer and fall (Shallenberger 1981; Lee 1993; Mobley et al. 2000). Sperm whale clicks recorded from hydrophones off Oahu confirm the presence of sperm whales near the Hawaiian Islands throughout the year (Thompson and Friedl 1982). The primary area of occurrence for the sperm whale is seaward of the shelf break in the Hawaiian Islands.

Sperm whales have been sighted in the Kauai Channel, the Alenuihaha Channel between Maui and the island of Hawaii, and off the island of Hawaii (Lee 1993; Mobley et al.1999; Forney et al. 2000). Twenty-one sperm whales were sighted during aerial surveys conducted in Hawaiian waters from 1993 through 1998. Sperm whales sighted during the survey tended to be on the outer edge of a 50 - 70 km distance from the Hawaiian Islands, indicating that presence may increase with distance from shore. However, from the results of these surveys, NMFS has calculated a minimum abundance of sperm whales within 46 km of Hawaii to be 43 individuals (Forney et al. 2000).

Status and Trends

Sperm whales were listed globally as endangered under the ESA in 1970. The sperm whale is the most abundant of the large whale species. The most recent estimate indicated a global population of between 300,000 and 450,000 individuals (Whitehead 2002; NMFS 2015c). In the North Pacific, Barlow and Taylor (2005) analyzed acoustic and visual detections in the temperate Northeastern Pacific ocean to estimate sperm whale abundance in this area of 26,300 to 32,100 individuals.

There are no trend assessments for sperm whales in the Pacific. The only study that references a trend was for the portion of Pacific sperm whales found off the California/Oregon/Washington coasts. The abundance of this population was estimated at 2,281 (no 95% confidence interval provided) individuals in 2014, with data from 1991 to 2014 showing a non-significant increasing trend and a significant increase in individuals traveling in small groups of 1 or 2 (Moore and Barlow 2017).

Population Dynamics

Female sperm whales reach sexual maturity around 9 years of age when they are roughly 29 ft. long. At this point, growth slows, and they produce a calf approximately once every five to seven years. After a 14 to 16-month gestation period, a single calf about 13 ft. long is born. Although calves will eat solid food before one year of age, they continue to nurse for several years. Females are physically mature around 30 years and 35 ft. long, at which time they stop growing.

For about the first 10 years of life, males are only slightly larger than females, but males continue to exhibit substantial growth until they are well into their 30s. Males reach physical maturity around 50 years and when they are 52 ft. long. Unlike females, puberty in males is prolonged, and may last until between ages 10 to 20 years old. Even though males are sexually mature at this time, they often do not actively participate in breeding until their late twenties. Their average lifespan is estimated to be between 80 and 90 years of age.

Diving and Social Behavior

Sperm whales have been documented to dive to depths of 2,000 m (6,562 ft.) for upwards of an hour (Watkins et al. 1993) with the longest dives on record lasting over 2 hours (Watkins et al. 2002). However, depending on the type of dive profile employed by the animal, typical foraging dives last approximately 40 to 45 minutes, half of which is spent at depth (Watkins et al. 1992; Amano and Yoshioka 2003; Watwood et al. 2006). Most active foraging dives appear to occur from 800 to 1000 m (Amano and Yoshioka 2003; Watwood et al. 2006) although geographical bathymetrics play a part in observed dive profiles (Aoki et al. 2007) and may be influenced by prey abundance (Davis et al. 2007; Irvine et al. 2017).

Sperm whales live in matrilineal groups of related females that associate with other female groups (Whitehead 1995; Richard et al. 1996; Conner et al. 1998; Gero et al. 2009) often creating long-term cooperative social units (Konrad et al. 2018). Behavioral and social structure vary among regions and ocean basins (Gero et al. 2014). However, typical group size in the Pacific ranges from 27 to 49 individuals and are larger than social groups documented in the Atlantic (Whitehead et al. 2012; Gero et al. 2014). Male calves usually leave these groups when they are roughly 6 years old and move northward to forage living solitary lifestyles until they are ready to breed. Sperm whales have the most extreme case of sexual size dimorphism documented among cetaceans. Males grow almost 1.5 times greater in size, both in length and weight, than females (Conner et al. 1998).

Threats to the Species

Sperm whales are affected by natural and anthropogenic threats throughout its range. Climate change is affecting all oceans where warming temperatures and ocean acidification could affect prey abundance (Quinn and Neibauer 1995; Mackas et al. 1998), or change ocean circulation where it can affect the distribution of prey (Whitehead 1997; Ruiz-Cooley et al. 2013). Increases in sea surface temperatures from the Pacific equatorial waters affect sperm whale feeding success and, in turn, calf production rates are negatively affected (Smith and Whitehead 1993; Whitehead 1997). Other natural threats include predation, disease, and food availability.

Direct harvest was the main cause of the initial depletion of sperm whales and other large whales. The total commercial killing of sperm whales worldwide between 1800 and 1909 has been estimated as close to 700,000 and between 1910 and 1973, as close to 605,000 (Best 1983). The IWC's moratorium on commercial whaling has largely mitigated this threat, however, the possibility of resumed whaling remains and has the potential to affect sperm whale populations. Members of the IWC reported harvesting 388 sperm whales under reservation or objection to the moratorium from 1986 through 2013; however, the last reported harvest occurred in the 1987/88 Antarctic season (IWC 2021). Although historical whaling activities were responsible for the depletion of sperm whales worldwide, they are now hunted only by Japan and in small numbers for scientific research, and therefore, the threat of overutilization by direct harvest is currently low (NMFS 2015b).

Sperm whales are impacted by both United States and foreign fisheries, primarily due to depredation of longline fisheries but also through capture and entanglement with gillnets, bottom-set longline and driftnet fisheries (NMFS 2015b). In the southern hemisphere, sperm whales depredate the Patagonian toothfish fisheries that operate off the coasts of Chile, South Georgia, Falkland Islands, Prince Edward Islands and Crozet and Kerguelen Islands (Clark and Agnew 2010; Goetz et al. 2010; Tixier et al. 2010). While detrimental to the fishery, sperm whale depredation can also lead to injury or entanglement of the whales (Hamer et al. 2012). Sperm whales have been found as bycatch in artisanal gillnets targeting sharks and pelagic fishes off the Pacific coasts of South America, Central America and Mexico (Palacios and Gerrodette 1996 as cited in NMFS 2015b).

In United States waters in the Pacific, sperm whales are known to have incidentally interacted with California thresher shark/swordfish drift gillnet operations, in which ten sperm whales were observed killed or seriously injured from 1990 to 2017 (Carretta et al. 2019). Of the eight sperm whales observed interacting with the California/Oregon drift gillnet fishery from 1990 to 2000, three were released alive and uninjured (37.5%), one was released injured (12.5%), and four were killed (50%) (Carretta et al. 2020). Therefore, approximately 63% of captured sperm whales could be accidentally killed or injured (based on the mortality and injury rate of sperm whales observed interacting with the United States fleet from 1990-2000). Based on past fishery performance, sperm whales are not observed interacting with this fishery every year; sperm whales were observed once between 2001 and 2012 (20% observer coverage; Carretta et al. 2020).

Bycatch of sperm whales in the California swordfish drift gillnet fishery has rarely been documented since the inception of the observer program in 1990. Between 1990 and 2017, 8,956 drift gillnet sets were observed. Observation of ten sperm whales entangled during six different observed sets, yielded a bycatch rate of approximately 1 sperm whale per 1,000 observed sets. All of the entanglements occurred during October through December in waters deeper than 1,500 m (820 fathoms), in proximity to steep continental shelf bathymetry (Carretta 2019).

Interactions between longline fisheries and sperm whales in the Gulf of Alaska have been reported (Rice 1989; Hill and DeMaster 1999; Straley et al. 2015; Muto et al. 2021). Observers aboard Alaskan sablefish and halibut longline vessels have documented sperm whales feeding on longline-caught fish in the Gulf of Alaska (Hill and DeMaster 1999; Muto et al. 2021; NMFS unpublished data) and in the South Atlantic (Ashford et al. 1996). The available evidence does not indicate sperm whales are being killed or seriously injured as a result of these interactions, although the nature and extent of interactions between sperm whales and long-line gear is not clear. Ashford et al. (1996) suggested that sperm whales pluck, rather than bite, the fish from the long-line. Mean estimated mortality of sperm whales in the Gulf of Alaska sablefish longline fishery was 3.7 per year from 2011-2015 (Muto et al. 2021).

In the United States WCPO purse seine fishery, four sets have interacted with a total of five sperm whales since 2005. After proportioning the unidentified marine mammals and accounting for unobserved sets the estimated total number of interactions with sperm whales from 2008-2018 is 6 (95th percentile: 13) (NMFS 2021a). Furthermore, sperm whales occur within the geographical range of the ASLL, however no interactions have occurred to date (NMFS unpublished data). Likewise, no interactions were noted in any United States West Coast highly migratory species fisheries to date (Carretta et al. 2017, 2020; NMFS 2021a).

In addition to fisheries, the status review (NMFS 2015) also included vessel interactions, pollutants, entanglement, research, and anthropogenic noise as potential anthropogenic threats to sperm whales. Most of these threats were considered low but noise from sonar and oil and gas testing are unknown and should be studied further, and climate change could potentially be high.

Conservation

Protection for the sperm whale occurs under both the ESA and the MMPA in the United States. The sperm whale was listed as endangered throughout its range under the precursor to the ESA, the Endangered Species 47 Conservation Act of 1969 (35 FR 8491; June 2, 1970) and remained on the list of threatened and endangered species after the passage of the ESA in 1973. Sperm whales are also listed in Appendix I of CITES and the Convention on the Conservation of Migratory Species of Wild Animals (CMS). The CITES classification is intended to ensure that commercial trade in the products of sperm whales does not occur across international borders. The CMS, also known as the Bonn Convention, aims to conserve terrestrial, aquatic, and avian migratory species throughout their range. Commercial whaling for this species ended in 1988 with the implementation of a moratorium against whaling by the IWC, and while it is often assumed that the worldwide population of sperm whales has increased since the moratorium, there are insufficient data on population structure and abundance of inhabited ocean basins to accurately determine population trends.

Summary of the Status

In this section of this biological opinion, we explained that the sperm whale is endangered, and that the species' trend appears to be uncertain (Moore and Barlow 2017). We used our knowledge of the species' demography and population ecology to capture the primary factors that appear to determine the sperm whale's population dynamics.

2.2.15 Main Hawaiian Island Insular False Killer Whale

Distribution and Population Structure

False killer whales are found worldwide mainly in tropical and warm-temperate waters (Stacey et al. 1994). NMFS determined MHI IFKWs constitute a DPS on September 18, 2012 (Oleson et al. 2012), and listed them as endangered (77 FR 70915, 28 November, 2012).

Evaluation of photographic and genetic data from MHI IFKW individuals suggests the occurrence of four separate social clusters (Martien et al. 2011; Baird et al. 2012, 2023; Figure

28). Mating occurs primarily, though not exclusively, within these clusters (Martien et al. 2011; Baird et al. 2023).

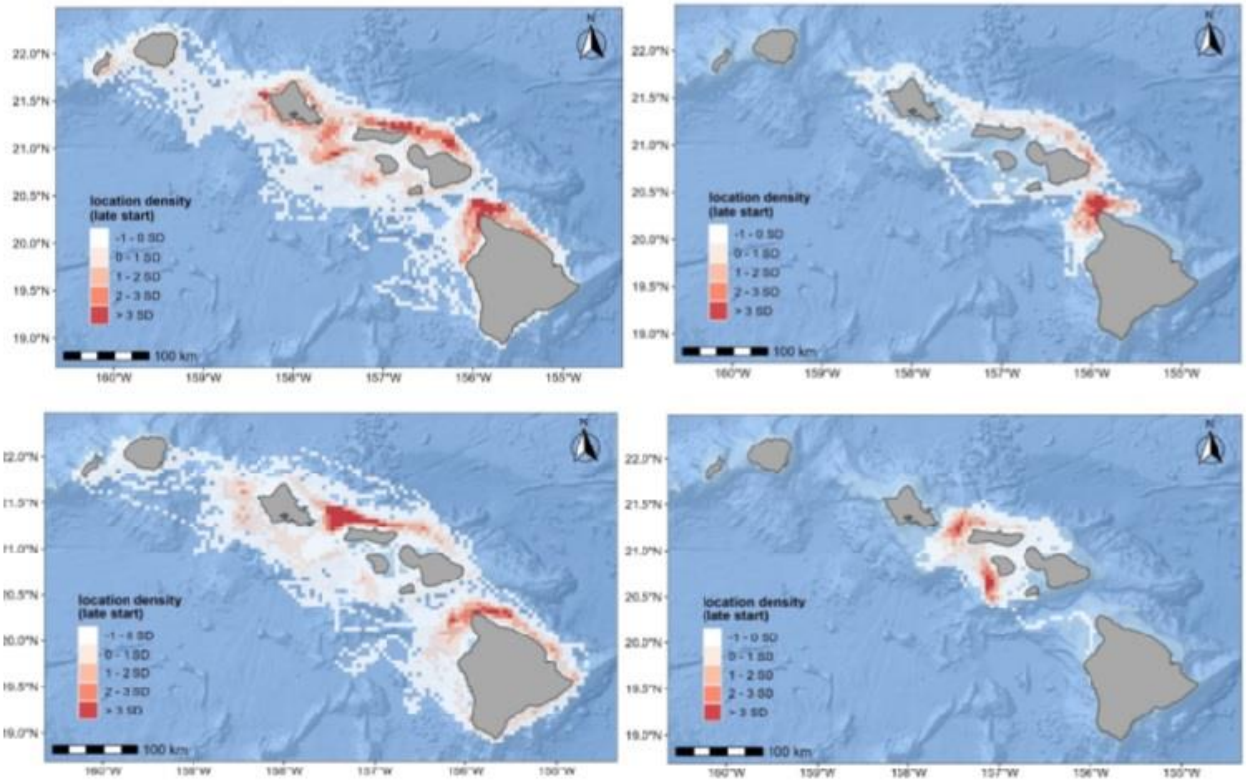

Figure 28. MHI IFKW density by clusters representing photo-identification data and 49 satellite-tagged individuals deployed through December 2022, using locations generated at 1-hour time intervals from a crawl mode (Baird et al. 2023). Upper left is Cluster 1, upper right is Cluster 2, lower left is Cluster 3 and lower right is Cluster 4.

The range and boundaries of the MHI IFKW have been assessed using data collected from satellite-linked telemetry tags as well as from ship and aerial surveys (Bradford et al. 2015). Satellite telemetry location data from 7 groups of individuals tagged off the islands of Hawaii and Oahu indicate that individuals move widely and quickly among the MHI and use waters up to 112 km offshore (Baird et al. 2010; Forney et al. 2010; Baird et al. 2023). Regular movement of individuals throughout the MHI has also been identified to occur over many years (Baird et al. 2005; Baird 2009; Baird et al. 2010; Baird et al. 2023). Movements between islands in both the leeward and windward waters can occur over the course of a few days (Baird 2009; Baird et al. 2010).

Ship survey sightings with photographs of individuals also confirm that MHI IFKW occur on both the windward and leeward sides of the MHI (Forney et al. 2010). Individuals from all tagged groups eventually range widely throughout the MHI, including movements to the west of Kauai and Niihau (Baird 2009; Forney et al. 2010; Baird et al. 2023). Based on locations obtained from 20 satellite tagged MHI IFKWs, the minimum convex polygon range was estimated to encompass 77,600 km^2 (M.B. Hanson unpublished data as cited in Oleson et al. 2010).

The greatest offshore movements occur on the leeward sides of the islands, although on average, similar water depths and habitat were utilized on both the windward and leeward sides of all islands (Baird et al. 2010). Tagged, as well as sighted individuals, utilize habitat overlaying a broad range of water depths varying from shallow (< 50 m) to deep (> 4000 m) (Baird et al. 2010). It is likely that movement patterns of the whales vary over time depending on the density and movement patterns of their prey species (Baird 2009).

Status and Trends

In December 2012, NMFS listed the MHI IFKW as endangered (77 FR 70915, 28 November, 2012). Abundance was previously estimated to be 167 (SE=23; 95% CI=128–218) animals within the surveyed area in 2015 (Bradford et al. 2018) and is based on encounter data from dedicated and opportunistic surveys for MHI IFKWs from 2000 to 2015 to generate annual mark-recapture estimates of abundance over the survey period. Using encounter data from dedicated and opportunistic surveys for MHI IFKWs from 1999 to 2021 with a capture-recapture model, Badger et al. (in review) estimated abundance to be 138 (95% credible interval [CRI]=120–160) animals in 2021. Badger et al. (in review) analyzed survey efforts and animal location data from telemetry deployments using kernel density estimators then compared the degree of overlap between those two processes using the overlap as a covariate for detection probability within a Jolly-Seber open population model in a Bayesian framework. Badger et al. (in review) determined the population has declined throughout the study period with an estimated annual trend of -3.51 (95% CRI: -8.40 to 2.04) over the entire time series and -5.53, (95% CRI = -9.91 to -1.61) over the past 10 years. Badger et al. (in review) also found that abundance appeared to vary by cluster, with the largest subset of the population being members of cluster 1 and fewest in cluster 4.

Population Dynamics

Female false killer whales reach sexual maturity between the ages of 8 and 11, while males mature 8 to 10 years later. Gestation ranges from 11 to 16 months, and lactation occurs for 1.5 to 2 years. Ferreira (2008) compared the large samples sizes from the Japanese drive fisheries (fisheries that intentionally strand pods for harvest) and a stranding event on the Atlantic coast of South Africa and found the following general trends: females were found to be approximately 84% of the length of males and had reached maturity at 9.2 years; males between the ages of about 8 and 18 were not found in the groups sampled, indicating dispersal of subadult males until they reach full sexual maturity; both males and females stopped growing between 25 and 30 years of age. The oldest estimated age (based on growth layers in teeth) was 63 for females and 58 for males, with females becoming reproductively senescencent at about age 44. The sex ratio was biased towards females at about 0.63. Understanding of their mating system remains poor, however it is suggested that they may be polygynous based on the males' large testes size (Ferreira 2008). The only reported birth interval, 6.9 years between calves, is from Japan (Kasuya 1986).

Diving and Social Behavior

From what we know of the limited available information about their diving behavior, maximum dive depth was estimated at 500 m (Cummings and Fish 1971). Time depth recorders have been deployed on four false killer whales (R. Baird et al. pers. comm. as cited in Oleson et al. 2010) totaling approximately 44 hours. The deepest dive recorded during a 22-hour deployment was estimated to have been as deep as 700 m (estimate based on duration past the recorders 234-m limit and ascent and descent rates). However, only seven dives were to depths greater than 150 m, all of them accomplished in the daytime. Other day dives ranged from 5–20 m and lasted for a minute or less, while dives that occurred at nighttime were shallow but longer (30-40 m max at approximately 6-7 minutes).

Overlap between the fishery's target depth range includes areas where MHI IFKW's forage. Indirect evidence of dive depths by false killer whales can be inferred from prey, where mahi mahi has been noted as a prominent prey item (Baird 2009). Based on the catch rates of longlines instrumented with depth sensors and capture timers, mahi mahi are caught closer to the surface in the daytime than other longline caught fish, primarily in the upper 100 m. Other prey species, such as bigeye tuna, typically occur much deeper, to at least 400 m (Boggs 1992). The deepest dives by the instrumented false killer whales approach the daytime swimming depth of swordfish (*Xiphias gladius*) near 700 m (Carey and Robinson 1981).

This DPS has been described as having a matrilineal social system (Ferreira 2008). However, this is not consistent because of two factors: 1) males leaving their natal group when they become sexually mature; and 2) finding females within a single group with different haplotypes, indicating that even among females, groups are composed of more than near-relatives (Chivers et al. 2010).

False killer whales are most commonly observed in smaller groups of approximately 10 - 20 individuals (Wade and Gerrodette 1993; Baird 2009; Baird et al. 2010), but have also been found in widespread aggregations of small groups of hundreds of individuals (Wade and Gerrodette 1993; Baird 2009; Reeves et al. 2009). These large aggregations can be spread over tens of kilometers yet appear to have coordinated movement directions (Baird et al. 2008). Mass strandings of large groups of false killer whales (range 50 to 835; mean = 180) have been documented in many regions (Ross 1984). Groups of 2–201 individuals (mean = 99) have also been driven ashore in Japanese drive-fisheries (Kasuya 1986). Analysis of age, sex, and maturity status from these mass mortality events indicates that these large groups include about equal numbers of males and females of various sizes (Odell and McClune 1999).

During a survey conducted in June and July 1989 on the leeward sides of Hawaii, Lanai, and Oahu to determine the minimum population size of false killer whales in Hawaiian waters, whales were observed on 14 occasions with three large groups (group sizes 470, 460, and 380) reported close to shore off the island of Hawaii on three different days (Reeves et al. 2009). Although the animals seen during the 1989 surveys are assumed to come from the insular population, based on their sighting location near the Big Island, it is possible that they represent a short-term influx of pelagic animals to waters closer to the islands. The average group size during the 1989 survey (195 animals) is larger than the typical average group size for the insular population (25 animals for encounters longer than two hours) during more recent surveys (Baird et al. 2005), and is larger than that observed for the pelagic population (12 animals) (Barlow and

Rankin 2007). However, because of the location of the sighting, it is likely that this group was part of the insular population (see page 5 in NMFS 2022a for further discussion).

Individuals have been known to form strong and long-term social bonds based on the studies done on the social organization of smaller groups near the MHI (Baird et al. 2008). As noted above in the Distribution and Population Structure section, MHI IFKWs appear to be comprised of 4 social clusters which consist of family members and regular associates displaying cluster specific spatial use patterns (Baird et al. 2008, 2012, 2023; Martien et al. 2019). False killer whales are also known to associate with other cetacean species, especially bottlenose dolphins (Leatherwood et al. 1988). Records also show false killer whales attacking other cetaceans, including sperm whales and bottlenose dolphins (Palacios and Mate 1996; Acevedo-Gutierrez et al. 1997).

Threats to the Species

Hooking, entanglement, or intentional harm caused by commercial and recreational fisheries, competition for food with commercial fisheries (Boggs and Ito 1993; Reeves et al. 2009), exposure to environmental contaminants (Ylitalo et al. 2009), and small population size are identified as the most substantial threats to the population (Carretta et al. 2019).

Effects of other threats such as nearshore hook and line fishing and environmental contaminants have yet to be assessed. There is significant geographic overlap between various nearshore fisheries and MHI IFKW distribution, and evidence of interactions with hook-and-line gear (e.g., Baird et al. 2015), such that these fisheries may pose a threat to the DPS. Five MHI IFKWs have recently stranded, including four from a single social cluster (Cluster 3), a high rate for a small demographic unit (Caretta et al. 2019).

Lastly, recent research has indicated that concentrations of polychlorinated biphenyls (PCBs) exceeded proposed threshold levels for health effects in 84% of sampled IFKW (Foltz et al. 2014).

Conservation

Effort has been made in the last several years to mitigate and reduce the impacts from the longline fisheries on false killer whales, including the MHI IFKW. As previously mentioned, on July 24, 2018, NMFS published the final rule designating critical habitat for the MHI IFKW by designating waters from the 45 m depth contour to the 3,200 m depth contour around the MHI from Niihau east to Hawaii (83 FR 35062).

A Take Reduction Team was established in January 2010 (75 FR 2853, 19 January 2010) after M/SI of false killer whales exceeded the potential biological removal (PBR[5]) in the fishery. The Team was tasked with developing recommendations to reduce incidental mortality and serious injury of the Hawaii pelagic, MHI insular, and Palmyra false killer whale populations in Hawaii-based longline fisheries. NMFS published a final FKWTRP based on the Team's

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[5] The PBR level is defined by the MMPA as the maximum number of animals, not including natural mortalities, that may be removed from a marine mammal stock while allowing that stock to reach or maintain its optimum sustainable population.

recommendations (77 FR 71260, 29 November 2012) which eliminated the seasonal contraction of the Longline Exclusion Zone (LLEZ) around the MHI. The TRP became effective December 31, 2012, with gear requirements effective February 27, 2013. These measures were not in effect during 2008-2012, a portion of the period for which bycatch was estimated in this report. 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 mortality and serious injury rate (see below, McCracken 2016). The implementation of the FKWTRP benefits the MHI IFKW. Specifically, the elimination of the seasonal contraction of the MHI Longline Fishing Prohibited Area effectively protects MHI IFKW from being exposed to the majority of the effort of the DSLL fishery. Currently 5.4% of the DPS' range overlaps with the DSLL fishery, accounting for approximately 4,200 square miles where interactions with the DPS may still occur. When the SEZ is closed, this area is closed to fishing removing any chance of the fishery from interacting with the DPS. Critical habitat is solely within the longline exclusion zone, and longline fishing does not occur within the geographical range of critical habitat.

Summary of the Status

In this section of this biological opinion, we explained that the MHI IFKW is endangered, and that the DPS' population trend is declining (Badger et al. in review). MHI IFKWs appear to be comprised of 4 social clusters which consist of family members and regular associates displaying cluster specific spatial use patterns (Baird et al. 2008, 2012, 2023; Martien et al. 2019). Abundance appears to vary by cluster, with the largest subset of the population being members of cluster 1 and fewest in cluster 4. Current estimates determined abundance to be only 138 individuals (CRI]=120–160; Badger et al. in review). Hooking, entanglement, or intentional harm caused by commercial and recreational fisheries, competition for food with commercial fisheries (Boggs and Ito 1993; Reeves et al. 2009), exposure to environmental contaminants (Ylitalo et al. 2009), and small population size are identified as the most substantial threats to the DPS (Carretta et al. 2019).

3 ENVIRONMENTAL BASELINE

By regulation, the Environmental Baseline refers to the condition of the listed species or its designated critical habitat in the Action Area, without the consequences to the listed species or designated critical habitat caused by the proposed action. The environmental baseline includes the past and present impacts of all state, federal or private actions and other human activities in the Action Area, anticipated impacts of all proposed federal projects in the Action Area that have already undergone formal or early section 7 consultation, and the impact of state or private actions which are contemporaneous with the consultation in process (50 CFR 402.02). The consequences to listed species or designated critical habitat from ongoing agency activities or existing agency facilities that are not within the agency's discretion to modify are part of the Environmental Baseline. The purpose of describing the environmental baseline in this manner in a biological opinion is to provide context for effects of the proposed action on listed species.

The past and present impacts of human and natural factors leading to the status of the 15 endangered and threatened species addressed in this biological opinion within the Action Area include the atmospheric, oceanic, ecological, and coastal effects of climate change; fisheries interactions; surface vessel traffic; pollution, and marine debris. These categories of activities and their impact on endangered and threatened species in the Action Area summarized in the narratives that follow.

3.1 Threats Posed by A Changing Global Climate

Globally averaged annual surface air temperatures have increased by about 1.8 ºF (1.0 ºC) over the last 115 years (1901 to 2016; Wuebbles et al. 2017). The earth's climate is now the warmest in the history of modern civilization. All of the relevant evidence points to human activities, particularly emissions of greenhouse gases since the mid-20th century, as the probable cause of this warming pattern (Wuebbles et al. 2017). Without major reductions in emissions, the increase in annual average global temperature relative to preindustrial times could reach 9 ºF (5 ºC) or more by the end of this century (Wuebbles et al. 2017). With significant reductions in emissions, the increase in annual average global temperature could be limited to 3.6 ºF (2 ºC) or less (Wuebbles et al. 2017). There is broad consensus that the further and the faster the earth warms, the greater the risk of potentially large and irreversible negative impacts (Wuebbles et al. 2017).

Increases in atmospheric carbon and changes in air and sea surface temperatures can affect marine ecosystems in several ways including changes in ocean acidity, altered precipitation patterns, sea level rise, and changes in ocean currents. Global average sea level has risen by about seven to eight inches since 1900, with almost half of that rise occurring since 1993. It is very probable that human-caused climate change has made a substantial contribution to sea level rise, contributing to a rate of rise that is greater than during any preceding century in at least 2,800 years (Wuebbles et al. 2017). Global average sea levels are expected to continue to rise by at least several inches in the next 15 years, and by one to four feet by 2100 (Wuebbles et al. 2017). Climate change can influence ocean circulation for major basin wide currents including intensity and position of western boundary currents (Gennip et al. 2017). These changes have potential for impact to the rest of the biological ecosystem in terms of nutrient availability as well as phytoplankton and zooplankton distribution (Gennip et al. 2017).

Effects of climate change on marine species include alterations in reproductive seasons and locations, shifts in migration patterns, reduced distribution and abundance of prey, and changes in the abundance of competitors or predators. Variations in sea surface temperature can affect an ecological community's composition and structure, alter migration and breeding patterns of fauna and flora and change the frequency and intensity of extreme weather events. For species that undergo long migrations (e.g., sea turtles), individual movements are usually associated with prey availability or habitat suitability. If either is disrupted, the timing of migration can change or negatively impact population sustainability (Simmonds and Eliott 2009). Over the long term, increases in sea surface temperature can also reduce the amount of nutrients supplied to surface waters from the deep sea leading to declines in fish populations (EPA 2010), and, therefore, declines in those species whose diets are dominated by fish. Acevedo-Whitehouse and Duffus (2009) proposed that the rapidity of environmental changes, such as those resulting from global warming, can harm immunocompetence and reproductive parameters in wildlife, to the detriment of population viability and persistence.

Climate change is projected to have substantial direct and indirect effects on individuals, populations, species, and the community structure and function of marine, coastal, and terrestrial ecosystems in the near future (McCarty 2001; IPCC 2014, 2018, 2022). Climate change will likely have its most pronounced effects on vulnerable species whose populations are already in tenuous positions (Williams et al. 2008). As such, we expect the risk of extinction for ESA-listed species to rise with the degree of climate shift associated with global warming. Increasing atmospheric temperatures have already contributed to documented changes in the quality of freshwater, coastal, and marine ecosystems and to the decline of endangered and threatened species populations (Mantua et al. 1997; Karl et al. 2009).

Elasmobranch species ranges are expected to shift as they align their distributions to match their physiological tolerances under changing environmental conditions (Doney et al. 2012). Climate-related shifts in range and distribution have already been observed in some marine mammal populations (Silber et al. 2017). Specialized diets, restricted ranges, or reliance on specific substrates or sites (e.g., for pupping) make many marine mammal populations particularly vulnerable to climate change (Silber et al. 2017). Marine mammals with restricted distributions linked to water temperature may be exposed to range restriction (Learmonth et al. 2006; Isaac 2009). MacLeod (2009) estimated that, based upon expected shifts in water temperature, 88% of cetaceans would be affected by climate change, 47% would be negatively affected, and 21% would be put at risk of extinction.

Hazen et al. (2012) examined top predator distribution and diversity of top marine predators in the Pacific Ocean in light of rising sea surface temperatures using a database of electronic tags and output from a global climate model. Hazen et al. (2012) predicted up to a 35% change in core habitat area for some key marine predators in the Pacific Ocean, with some species predicted to experience gains in available core habitat and some predicted to experience losses. Notably, leatherback sea turtles were predicted to *gain* core habitat area, whereas loggerhead sea turtles and blue whales were predicted to lose core habitat area. Such range shifts could affect marine mammal and sea turtle foraging success as well as sea turtle reproductive periodicity (Kashner et al. 2011; Pike 2014).

Significant impacts to marine mammals and sea turtles from ocean acidification may be indirectly tied to foraging opportunities resulting from ecosystem changes (Busch et al. 2013; Haigh et al. 2015; Chan et al. 2017). Nearshore waters off California have already shown a persistent drop in pH from the global ocean mean pH of 8.1 to as low as 7.43 (Chan et al. 2017). The distribution, abundance and migration of baleen whales reflects the distribution, abundance and movements of dense prey patches (e.g., copepods, euphausiids or krill, amphipods, and shrimp), which have in turn been linked to oceanographic features affected by climate change (Learmonth et al. 2006). Ocean acidification may cause a shift in phytoplankton community composition and biochemical composition that can impact the transfer of essential nutrients to predators that eat plankton (Bermudez et al. 2016).

Blue whales, as predators that specialize in eating krill, are likely to change their distribution in response to changes in the distribution of krill (Payne et al. 1986, 1990; Clapham et al. 1999). Krill have been shown to suffer decreased larval development and survival under lower pH conditions (McLaskey et al. 2016). Krill also have lower metabolic rates after both short-term and long-term exposure to low pH (Cooper et al. 2016). Increased ocean acidification may also have serious impacts on fish and larval development and behavior (Raven et al. 2005, Bignami et al. 2013) that could impact fish populations (Munday et al. 2009). In turn this could negatively impact piscivorous threatened and endangered species that rely on those populations for food.

Sea turtles have temperature-dependent sex determination, and many populations already produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Newson et al. 2009; Patrício et al. 2017; Jensen et al. 2018). Altered sex ratios have been observed in sea turtle populations worldwide (Mazaris et al. 2008; Reina et al. 2008; Robinson et al. 2008; Fuentes et al. 2009). Female-biased green sea turtle sex ratios have been reported for East Pacific green sea turtles at foraging locations in San Diego Bay, California (Allen et al. 2015). For North Central Pacific green sea turtles, Chaloupka et al. (2008b) did not report gender bias and Franke and Turtle Research Program Staff (2015) came to the same conclusion for green sea turtles in Hawaii which would primarily represent the Central North Pacific species,. A fundamental shift in the demographics of species such as sea turtles may lead to increased instability of populations that are already at risk from several other threats. In addition to altering sex ratios, increased temperatures in sea turtle nests can result in reduced incubation times (producing smaller hatchling), reduced clutch size, and reduced nesting success due to exceeded thermal tolerances (Fuentes et al. 2009; Fuentes et al. 2010a; Fuentes et al. 2011; Azanza-Ricardo et al. 2017).

Other climatic aspects, such as extreme weather events, precipitation, ocean acidification and sea level rise also have potential to affect marine turtle populations. Changes in global climatic patterns will likely have profound effects on the coastlines of every continent, thus directly impacting sea turtle nesting habitat (Wilkinson and Souter 2008). In some areas, increases in sea level alone may be sufficient to inundate turtle nests and reduce hatching success by creating hypoxic conditions within clutches (Caut et al. 2010; Pike et al. 2015). Flatter beaches, preferred by smaller sea turtle species, would likely be inundated sooner than would steeper beaches preferred by larger species (Hawkes et al. 2014).

Moderately small increases in sea level can result in the loss of a large proportion of nesting beaches in some locations. For example, a study in the northwestern Hawaiian Islands predicted that up to 40% of green turtle nesting beaches could be flooded with a 0.9 m sea level rise (Baker et al. 2006). The loss of nesting beaches would have catastrophic effects on sea turtle populations globally if they are unable to colonize new beaches that form, or if the newly formed beaches do not provide the habitat attributes (sand depth, temperature regimes, refuge) necessary for egg survival. Poloczanska et al. (2009) noted that extant marine turtle species have survived past climatic shifts, including glacial periods and warm events, and therefore may have the ability to adapt to ongoing climate change (e.g., by finding new nesting beaches). However, the authors also suggested since the current rate of warming is very rapid, expected changes may outpace sea turtles' ability to adapt.

Changing patterns of coastal erosion and sand accretion, combined with an anticipated increase in the number and severity of extreme weather events, may further exacerbate the effects of sea level rise on turtle nesting beaches (Wilkinson and Souter 2008). After Hurricane Walaka went through the Central North Pacific in October of 2018 much of East Island in French Frigate Shoals was left largely inundated. Roughly 50% of the Central North Pacific green sea turtles nest at East Island and the future impacts from the inundation of the majority of East Island is unclear at this time. Climate change is expected to affect the intensity of hurricanes through increasing sea surface temperatures, a key factor that influences hurricane formation and behavior (EPA 2010). Extreme weather events may directly harm sea turtles, causing "mass" strandings and mortality (Poloczanska et al. 2009). Studies examining the spatio-temporal coincidence of marine turtle nesting with hurricanes, cyclones and storms suggest that cyclical loss of nesting beaches, decreased hatching success and hatchling emergence success could occur with greater frequency in the future due to global climate change (Hawkes et al. 2009).

Habitat for many shark and ray species is comprised of open ocean environments occurring over broad geographic ranges, large-scale impacts such as climate change may impact these species. Chin et al. (2010) conducted an integrated risk assessment to assess the vulnerability of several shark and ray species on the Great Barrier Reef to the effects of climate change. Scalloped hammerhead sharks for instance were ranked as having a low overall vulnerability to climate change, with low vulnerability to each of the assessed climate change factors (i.e., water and air temperature, ocean acidification, freshwater input, ocean circulation, sea level rise, severe weather, light, and ultraviolet radiation). In another study on potential effects of climate change to sharks, Hazen et al. (2012) used data derived from an electronic tagging project and output from a climate change model to predict shifts in habitat and diversity in top marine predators in the Pacific out to the year 2100. Results of the study showed significant differences in habitat change among species groups, but sharks as a whole had the greatest risk of pelagic habitat loss.

Environmental changes associated with climate change are occurring within the Action Area and are expected to continue into the future. Marine populations that are already at risk due to other threats are particularly vulnerable to the direct and indirect effects of climate change. The oceanic whitetip shark and giant manta ray considered in this opinion have likely already been impacted by this threat through the pathways described above.

3.2 Fisheries Interactions

The various countries throughout the North Pacific Ocean engage in a wide variety of commercial, artisanal, subsistence, and recreational fisheries that target an equally wide variety of species that include anchoveta, anchovies, barracudas, breams, carangids, catfishes, clams and squids, cockles, crabs, croakers, herring, mackerel, mullets, pilchard, ponyfishes, prawns, sardinellas, scads, sharks and rays, shrimps and squids, snappers, and highly migratory species such as tuna, marlin, and swordfish. Although many of these fisheries interact with endangered and threatened species, and affect the status of several of the species within the Action Area, the overwhelming majority of these fisheries occur in coastal areas of the Pacific and do not occur in the Action Area.

The main fisheries that occur in the Action Area and that have measurable effect on the endangered and threatened species considered in this biological opinion are fisheries for highly migratory species using longline and purse seine gear. Key target species for these fisheries are tuna (albacore, bigeye, skipjack and yellowfin), swordfish and marlin. Fisheries that use both types of gear have occurred in the western and central Pacific Ocean since the 1950s (for longline fisheries; 1980s for purse seine fisheries), and fisheries managers have, since 2004, been collecting robust data that can be used to assess the impacts of these fisheries on endangered and threatened species in the North Pacific Ocean generally, and in the Action Area specifically.

Nevertheless, the early stages of these fisheries probably had effects on the abundance and dynamics of populations of imperiled species in the Action Area. Given the longevity and long generation times of the endangered and threatened species that occur in the Action Area, those impacts are likely to still influence the dynamics of populations of these species. We note, given MHI IFKW’s range is entirely within the Hawaii EEZ, we do not expect foreign fleets to fish within the range of the species.

The Action Area overlaps with the management areas of two Regional Fishery Management Organizations (Figure 29): in the western portion of the Action Area, the WCPFC manages fisheries for highly migratory species targeted by 26 nations, 7 territories, and 7 cooperating non-member nations. In the eastern portion of the Action Area, the IATTC manages fisheries for highly migratory species targeted by 21 nations and 5 cooperating non-member nations.

Many of the impacts of these fisheries on the endangered and threatened species that occur in the Action Area have already been summarized in the narratives presented in the Status of the Listed Resources section of this biological opinion. However, data from the WCPFC and IATTC provide additional insight into the impacts of fisheries for highly migratory species on endangered and threatened species in the Action Area.

The number of vessels that participate in domestic and foreign longline fisheries in the western and central Pacific region has fluctuated between 3,000 and 6,000 over the last 30 years, which includes the 100 to 145 permitted vessels in the Hawaii longline fisheries (WPRFMC 2018). The four main target species are yellowfin tuna, bigeye and albacore tuna, and swordfish.

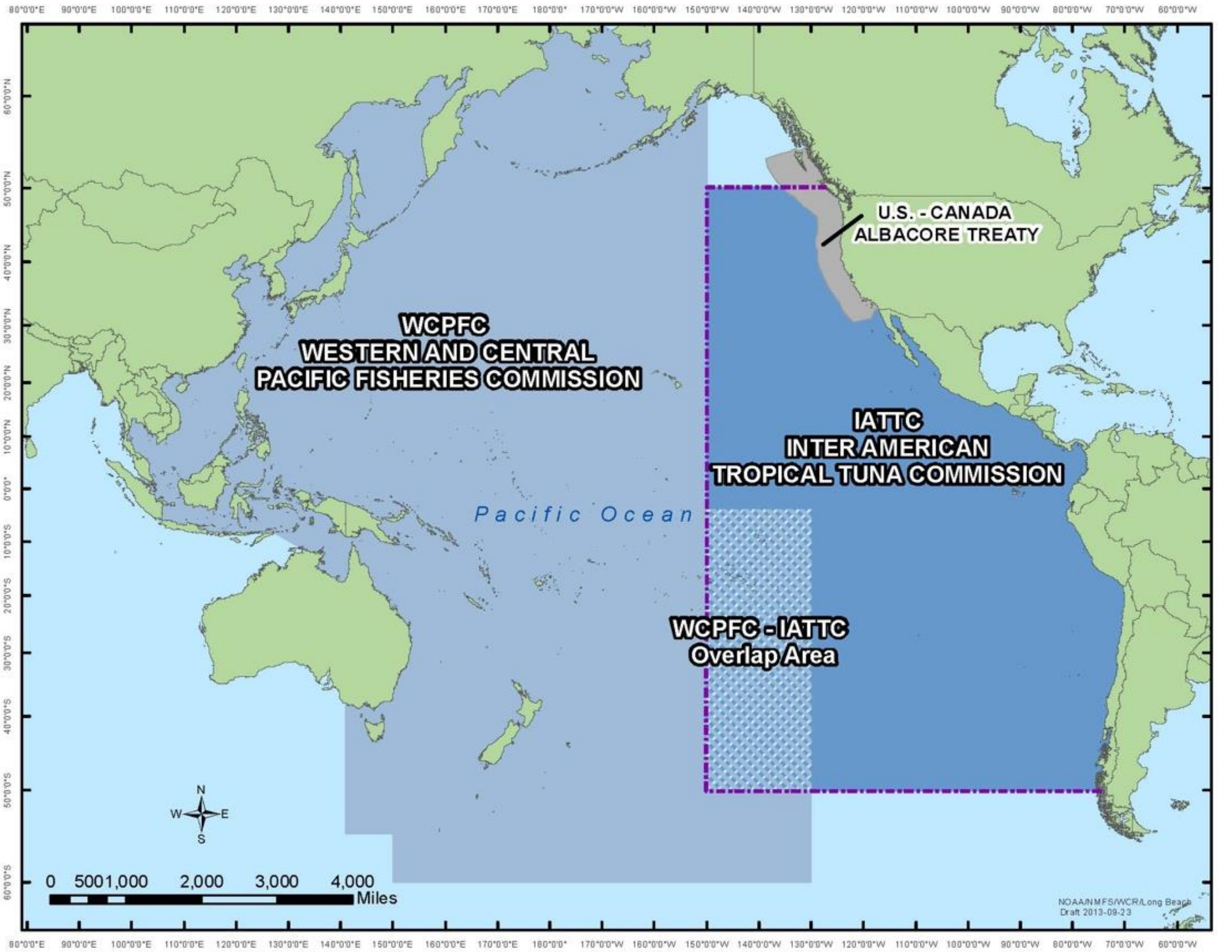

Figure 29. The boundaries of the Western Central Pacific Fisheries Commission (west of the line) and the Inter-American tropical tuna Commission (east of the line).

Table 6 presents the estimated number of the different species of sea turtles captured in fisheries operated by 16 countries in the western and central Pacific Ocean from 1989 to 2015 (the data available do not allow us to distinguish between the different species of green turtles listed pursuant to the ESA so they are grouped under one heading). Other authors provide different estimates: Lewison et al. (2004) estimated that 2,600 to 6,000 loggerhead and 1,000 to 3,000 leatherback sea turtles were captured in 2000 while Peatman et al. (2018b) estimated that from 6,619 to 41,180 loggerhead sea turtles and a median of about 9,920 leatherback turtles were captured in fisheries in the North Pacific Ocean between 2003 and 2017. Beverly and Chapman (2007) produced estimates that differ from both of these sources. These various estimates cover different time intervals, were produced by a variety of different assumptions, and rely on data collected from fisheries with limited observer coverage, so their differences are not surprising. Nevertheless, they capture the approximate scale of the number of sea turtles that have been captured by fisheries in the Action Area. While some of these interactions occur outside of the Action Area, mortalities resulting from those interactions impact the condition of the species within the Action Area.

Table 6. Estimates of the number of the different species of sea turtles captured in fisheries operated by 16 countries in the western and central Pacific Ocean from 1989 to 2015. The "estimated" number of captures expand the reported numbers to account for the 5% observer coverage used by the WCPFC (NMFS unpublished data).

Totals	**Leatherback Sea Turtle**	**Loggerhead Sea Turtle**	**Green Sea Turtle**	**Olive Ridley Sea Turtle**
Total reported	331	549	325	762
Total estimated	6,620	10,980	6,500	15,240

Finally, bycatch estimates of sea turtles were summarized from annual reports by the WCPFC (2021a). Data for sea turtles are summarized in Table 7. This information is not exclusive of United States fishery data, based on total hooks per year, the Hawaii DSLL fishery represents 5 to 6% of the total hooks set by WCPO longline fisheries. Figure 30 shows the trends in CPUE (captures per hook) from 2013 to 2019 for sea turtles captured in WCPO longline fisheries. CPUE was calculated as observed turtle captures divided by observed hooks.

Table 7. Average and (95% confidence interval) of the annual numbers of sea turtles reported captured/killed from 2013 to 2020 by participating countries reporting catch data to the Western and Central Fisheries Commission for longline fisheries operating in the WCPO, data were reported in 5°x 5°bins. Estimated Interactions and Mortalities calculated using the Expansion Factor ([100/Observer Coverage] x [Observed Interactions] (WCPFC 2021a)*.

Species	**Observed Interactions**	**Observed Mortalities**	**Estimated Interactions**	**Estimated Mortalities**
Green Sea Turtle	34 (28 – 39)	17 (14 – 21)	1,260 (1,091 - 1,429)	645 (531 – 758)
Leatherback Sea Turtle	18 (14 – 22)	1 (1 – 3)	722 (468 – 976)	76 (16 – 136)
Loggerhead Sea Turtle	69 (32 – 106)	11 (8 – 13)	2,387 (1,318 – 3,457)	390 (327 – 452)
Olive Ridley Sea Turtle	92 (46 – 138)	71 (32 – 110)	3,101 (1,955 – 4,246)	2,337 (1,405 – 3,268)

*Percent Observer Coverage (2013-2020): 2.8% (2.2 – 3.4%)

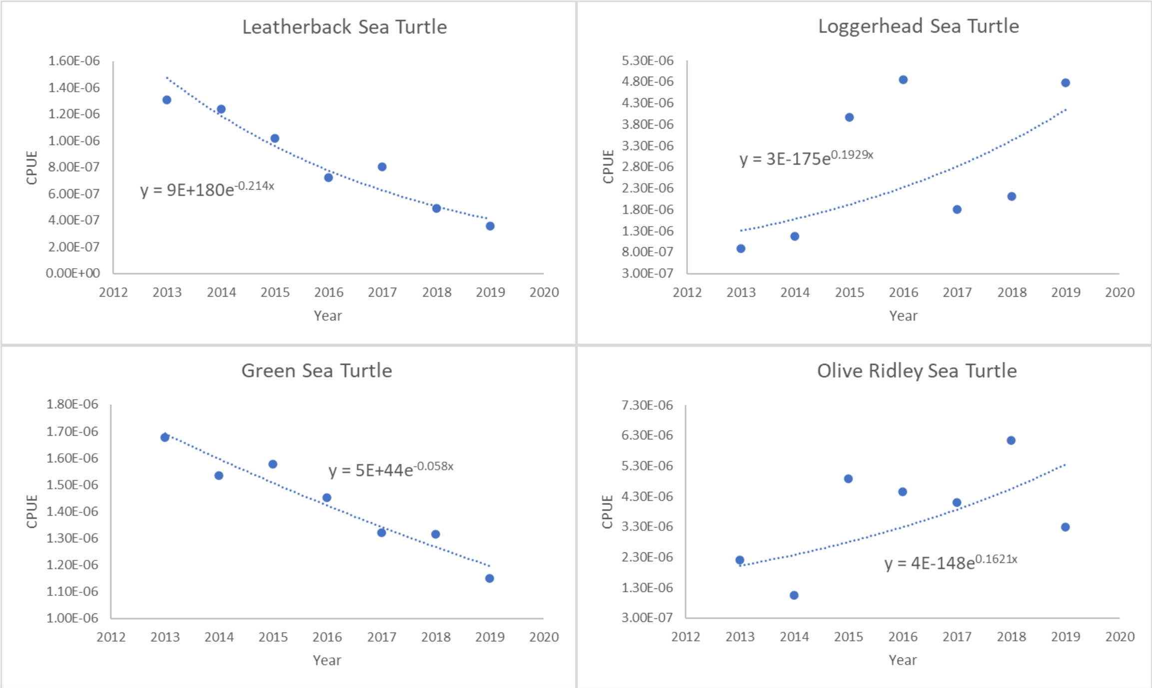

Figure 30. Trends in CPUE for sea turtles captured in the WCPO longline fisheries. Dotted lines are fitted exponential curves, equations describe these curves and the exponent in the equations is the mean percent per year change in CPUE.

Without an accompanying estimate of the number of times individual sea turtles are captured in these fisheries, the number of these sea turtles that were dead or seriously injured when captured, the number released with serious injuries, and the number that were likely to die or fail to reproduce after they were released, it is difficult to assess the precise impact of these fisheries on sea turtles. However, given the estimated mortality values, capturing this number of sea turtles in fisheries in the Action Area would cause populations of these endangered and threatened sea turtles to experience further declines in their abundance and reproductive output.

Oceanic whitetip sharks were once one of the most abundant pelagic shark species encountered in the western and central Pacific Ocean (Molony 2007). Substantial and sustained declines in catch-per-unit-of-effort have been documented for the oceanic whitetip shark population within the western and central Pacific region and have been reported to exceed 90% declines (Clarke et al. 2011a, 2011b, 2012; Lawson 2011; Rice and Harley 2012; Rice et al. 2015; Young et al. 2017). To attach numbers to these declines, Peatman et al. (2018b) estimated that about 1,470,000 oceanic whitetip sharks were captured in longline fisheries in the area managed by the WCPFC between 2003 and 2017. Peatman et al. (2018a) estimated that about 13,882 (median estimate) oceanic whitetip sharks were captured by the purse seine fleet managed by the WCPFC from 2003 to 2017. These are median catch estimates based on data collected from fisheries with limited observer coverage, so the estimates have wide confidence intervals. Nevertheless, these estimates capture the approximate scale of the interactions between longline fisheries and oceanic whitetip sharks in portions of the Action Area. Again, as these numbers represent the entire WCPFC boundaries, we cannot parse out the number of individuals expected to be taken from the Action Area. However, at this time, this is the best available science regarding numbers of sharks harvested within the Region.

As noted in the Status of Listed Resources section, Rice et al.'s (2020) assessment of future trends in oceanic whitetip sharks in the WCPO is dependent on trends in fishery captures and mortalities since 2016. We use the WCPFC bycatch data (WCPFC 2021a) to assess these trends and compare it to trends in the Hawaii DSLL fishery (Figure 31). Comparing the average CPUEs from 2017 to 2019 to those of 2016, total CPUE declined by 25% in the WCPO longline fisheries and by 41% in the Hawaii DSLL fishery. At-vessel mortality CPUE declined by 29% in the WCPO longline fisheries and by 50% in the Hawaii DSLL fishery. Note that we did not include data for 2020 due to the unusual circumstances presented by the COVID-19 pandemic that changed observer rates and patterns. Also note that the DSLL shifted from wire to monofilament leaders as of May 2021 which is expected to reduce the number of sharks that are brought to the vessel by 30% as these sharks are expected to bite off and escape (Bigelow and Carvalho 2021). It is expected that this will further reduce overall mortalities. In a recent assessment, Bigelow et al. (2022) suggest the recent initiatives that prohibit retention, improve handling and release conditions, and shifts to monofilament leaders are likely to results in increasing trends for WCPO oceanic whitetip sharks.

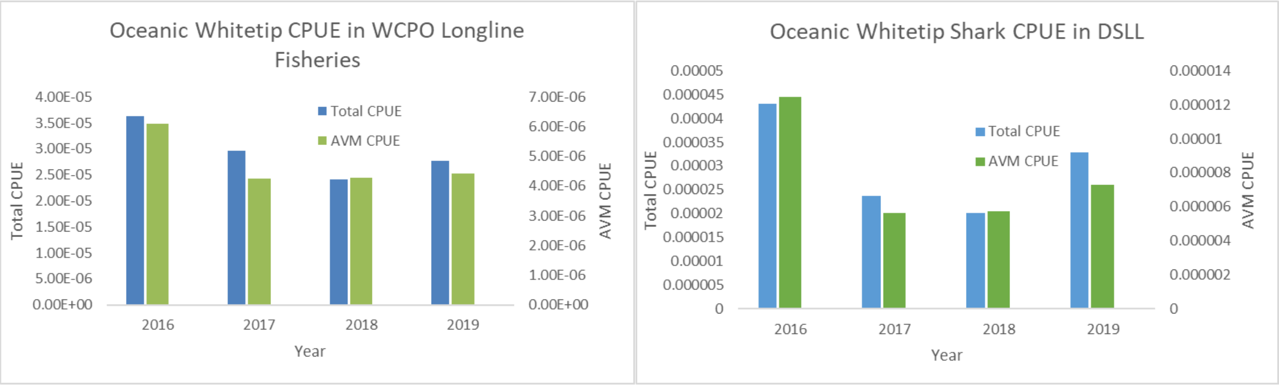

Figure 31. Total observed oceanic whitetip shark catch per observed hook (Total CPUE) and observed at-vessel mortality per observed hook (AVM CPUE) for all WCPO longline fisheries (inclusive of United States; exclusive of fisheries occurring in Vietnam and archipelagic waters of Papua New Guinea, Indonesia and the Philippines; WCPFC 2021a; left panel) and for the DSLL (right panel) from 2016 to 2019.

It is difficult to assess the precise impact these fisheries have had on oceanic whitetip sharks. However, capturing this number of oceanic whitetip sharks would have resulted in the death of a substantial number of individuals. Therefore, we are reasonably certain these fisheries contributed to the declines in oceanic whitetip shark. Data on Indo-West Pacific scalloped hammerheads contains significant uncertainty on current catch and reporting estimates given that many hammerhead species are reported in aggregate (Miller et al. 2014; WCPFC 2021b). Despite requirements for better reporting (see WCPFC 2011), recent catches of hammerheads have not been provided to the WCPFC for a number of longline fleets, including fleets from among the top twenty countries reporting Pacific shark catches to the FAO (Miller et al. 2014). Considering the lack of evidence, we cannot reliably determine the impact from foreign fleets on this species within the Action Area.

Rays are currently not considered key bycatch species and therefore have not been included in WCPFC bycatch summaries. However, their bycatch rates were summarized by Tremblay-Boyer and Brouwer (2016). Currently, observers in the WCPFC are not instructed to distinguish between mantas and mobulids. The total number of manta and mobula ray interactions documented by observers from 2003-2017 totals 1,800. The majority were discarded either alive-healthy-injured or alive-unknown. Many were discarded in an unknown condition while a smaller proportion were discarded alive-dying or dead.

According to Tremblay-Boyer and Brouwer (2016), giant manta rays are observed at a rate of 0.001-0.003 individuals per 1,000 hooks in the longline fisheries. The longline standardized CPUE data, while short, provides a more accurate representation of the species' abundance trend (due to traditional focus on species in longline observer programs) and indicate that giant manta rays are observed less frequently in recent years compared to 2000-2005 (Tremblay-Boyer and Brouwer 2016). The IATTC reports that there is significant uncertainty as to whether they receive all bycatch data from the high-seas longline fishery and consider the data of limited or no value for scientific purposes (IATTC 2021). Therefore, we cannot estimate the number or impact to giant manta rays within the IATTC region of the action area, other than NMFS collected data reported in this consultation.

Only 2 interactions have been reported with sperm whales in the WCPFC from 2015-2020 (WCPFC 2021c), however it is not clear if those interactions occurred within the Action Area or the RFMO’s greater geographical area. Additionally, given the lack of data reported by the IATTC, we cannot determine if any interactions have occurred with the species with foreign fleets that overlap the Action Area in the geographical range as managed by that RFMO.

3.2.1 Other Domestic Fisheries in the Action Area

In addition to the Hawaii DSLL fishery, the domestic federal Hawaii SSLL and West Coast DSLL fisheries also occur in the Action Area. The Hawaii SSLL fishery overlaps partially on the north side of the MHI and the West Coast DSLL fishery partially overlaps near the California EEZ.

In 2004, the Hawaii longline fishery was split into shallow and deep set sectors via a Regulatory Amendment. Since 2004, the Hawaii SSLL fishery has had 100% observer coverage, which provided a robust data set of the number of interactions (i.e. hooking and entanglement) between that fishery and threatened and endangered species over 15 years. Data on interactions with listed species are available for a longer period; however, numbers of sea turtle interactions dropped considerably following the implementation of gear changes adopted with the reopening of the Hawaii SSLL fishery in 2004. Since then, interactions have declined by 84% for leatherback sea turtles and 95% for loggerhead sea turtles (Swimmer et al. 2017). Interactions with green sea turtles and olive ridley sea turtles also dropped as a result of the change in operations.

The Hawaii SSLL fishery and the Hawaii DSLL fishery overlap on the north side of the MHI. From 2004 to 2022 there have been 395 records of sea turtles being captured (Table 8) in the Hawaii SSLL fishery.

Table 8. Number sea turtles, by species, interacting with the Hawaii SSLL fishery between 2004 and 2022, inclusive of both dead and live releases.

Observed turtle status	Green sea turtle	Leatherback sea turtle	Loggerhead sea turtle	Olive Ridley sea turtle	Grand Total
Hooked or entangled; released injured	11	121	248	15	395
Dead	-	-	3	1	4
Grand Total	11	121	251	16	399

Between 2004 and 2022, 988 oceanic whitetip sharks were caught in the Hawaii SSLL fishery with an estimated 21 interactions with giant manta rays (17 observed plus 4 unidentified but pro-rated giant manta rays; NMFS 2019a; NMFS unpublished data). Projected annual interaction rates are provided in Table 9 for the various species which may also be caught in this (the DSLL) fishery.

Table 9. The estimated annual number and amount of exempted take of sea turtles, oceanic whitetip shark and giant manta ray annual interaction and associated mortality with various species in the Hawaii SSLL fishery (NMFS 2019a).

Species	**Annual**	
	Number Captured	**Total Number Killed**
Leatherback sea turtle	21	3
Loggerhead sea turtle	36	6
*Olive ridley sea turtle (all species)	5	1
**Green sea turtle (all species)	5	1
Oceanic whitetip shark	102	32
Giant manta ray	13	4

*Mexican breeding population (endangered) and all other breeding populations (threatened) considered in NMFS (2019a).

**East Pacific, Central North Pacific, East Indian-West Pacific, Southwest Pacific, Central West Pacific, and Central South Pacific DPSs of green sea turtles considered in NMFS (2019a).

Interactions with MHI IFKW false killer whales are not anticipated with any U.S. longline fishery except the DSLL (Nitta and Henderson 1993; Oleson et al. 2010) given the lack of overlap between the SSLL fishery and the species' range. Lastly, the West Coast based DSLL fishery (NMFS 2016a) is expected to overlap the Action Area. As of 2016 only one vessel was participating in this fishery, but it is expected that the maximum number of hooks set in the fishery is about 800,000 per year (NMFS 2016a). This is based upon six vessels making 14 sets per trip, 5 trips per season, and setting 1,900 hooks per set (NMFS 2016a).

Since 2005, all of the fishing activity has occurred between the equator and 35° north latitude and between the United States and Mexico EEZ boundaries (200 nm from shore) and 140°W longitude. However, there is a seasonal prohibition on longline gear use from April 1 to May 31 in waters bounded on the south by 0° latitude, on the north by 15°N. latitude, on the east by 145°W longitude, and on the west by 180°W longitude. Table 10 shows the anticipated number of turtle interactions and mortality in this fishery. No interactions have been noted with oceanic whitetip sharks or giant manta rays in any United States West Coast highly migratory species fisheries to date (C. Villafana and C. Fahy pers. comm. to J. Rudolph; March 7, 2019). Additionally, we did not find any records or evidence of interactions with whale species considered herein.

Table 10. Number of anticipated turtle interactions and associated mortality (in parentheses) from the West Coast Region DSLL fishery Incidental Take Statement during the next ten consecutive years, beginning in 2016 (NMFS 2016a).

Species	**Estimated entanglements (mortalities over ten years**
East Pacific and Central North Pacific Green Sea Turtle	Up to 1 (1)
Leatherback	4 (2)
Olive Ridley	6 (6)
North Pacific Loggerhead	Up to 1 (1)

3.2.2 Hawaii Deep Set Longline Historical Effort

Although the effects of the Hawaii DSLL fishery are included in the above discussion, herein we provide a few more details specific to this fishery. The assessments in NMFS' 2014 and 2017 biological opinions assumed 128 vessels would participate in the fishery and make approximately 1,305 trips, with 18,592 sets, consisting of 46,117,532 hooks annually (NMFS 2014, 2017b). As Table 11 demonstrates, these two earlier opinions underestimated the number of vessels that would participate in the fishery each year from 2014 to 2017 as well as the number of trips, and the number of hooks set in three of the four years between 2014 and 2017. In addition, the number of sets the two prior opinions assumed would occur each year was exceeded once, was approximately correct once, and overestimated the number of sets twice.

These discrepancies are not surprising because the two prior opinions only had 10 to 13 years of data to support their projections. Although a 10- to 13-year interval is a long time in terms of lived experience, it represents a relatively small sample for the purposes of statistical analysis. More importantly, it is a relatively small statistical sample with a lot of year-to-year variation.

Table 11. The number of active vessels and fishing effort in the Hawaii DSLL fishery, 2004-2022 (includes effort in both WCPO and EPO). Blue shading denotes the lowest values in the table, yellow shading denotes the highest values (NMFS 2018b; NMFS unpublished data).

Year	**Vessels making deep-sets**	**Deep-set fishing effort (trips)**	**Deep-set fishing effort (sets)**	**Deep-set fishing effort (hooks)**
2004	125	1,522	15,902	31,913,246
2005	124	1,590	16,550	33,663,248
2006	127	1,541	16,452	34,597,343
2007	129	1,588	17,815	38,839,377
2008	127	1,532	17,885	40,083,935

Year	Vessels making deep-sets	Deep-set fishing effort (trips)	Deep-set fishing effort (sets)	Deep-set fishing effort (hooks)
2009	127	1,402	16,810	37,770,913
2010	122	1,360	16,085	37,244,432
2011	129	1,462	17,173	40,766,334
2012	128	1,356	18,069	43,965,781
2013	135	1,383	18,772	46,919,110
2014	139	1,350	17,777	45,464,747
2015	143	1,452	18,519	47,600,000
2016	142	1,354	17,988	47,400,181
2017	145	1,502	19,488	53,013,297
2018	143	1,643	21,012	58,600,000
2019	149	1,737	22,234	63,400,000
2020	146	1,644	20,785	59,700,000
2021	146	1,614	22,192	62,700,000
2022	144	1,506	20,949	62,200,000

As shown in Figure 32, active participation in the deep-set fishery since 2004 has remained below 164, but has increased from 2010 to 2019, decreasing from 149 vessels in 2019 to 144 vessels in 2022.

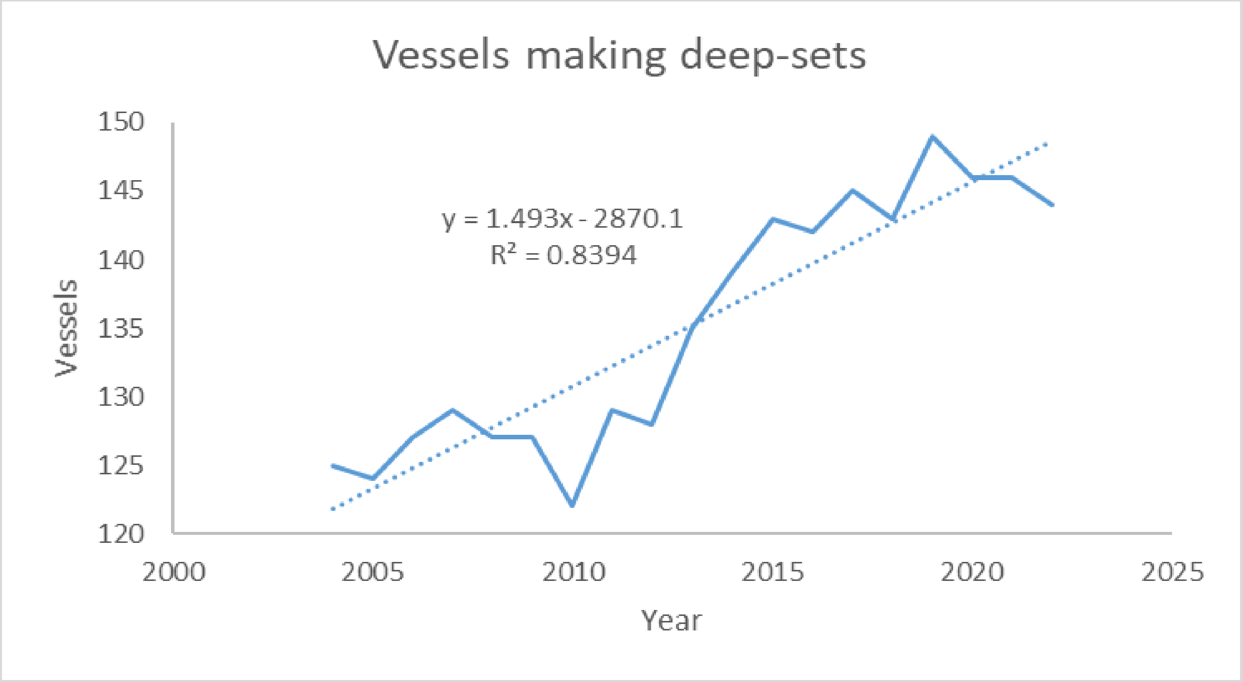

Figure 32. Summary of effort for the Hawaii DSLL fishery since 2004. See Table 11 for the data (data from NMFS 2018a and pers. comm. J. Makaiau to A. Garrett, May 19, 2022).

NMFS (2018b) indicated that the number of hooks deployed by the Hawaii DSLL fishery grew from around 32 million in 2004 to 53 million in 2017 (See also Table 11). This represented a 14.95% increase in the number of hooks compared to the 46,117,532 million hooks evaluated in the 2014 biological opinion and 2017 supplemental biological opinion. NMFS (2018a) anticipated this trend would potentially continue at rates similar to those in the past five years. Based on a 14.95% increase in hooks deployed by the deep-set fishery as compared to the 2014 Hawaii DSLL fishery biological opinion (NMFS 2014) and the 2017 supplemental biological opinion (NMFS 2017b), NMFS (2018a) anticipated under the proposed action that the fishery could potentially deploy up to 60,938,785 hooks over the next 5 years.

Current numbers of hooks per year indicate that the fishery is deploying increased hooks per year at a faster rate than NMFS (2018a) originally anticipated (Figure 33), and while this increase is expected to continue into the future (J. Makaiau pers. comm. to A. Garrett, May 19, 2022), there is an upper limit to the number of hooks per year the fishery can set. The Hawaii DSLL fishery does not impose any limits on the annual numbers of trips, sets or hooks deployed by vessels in a given year, however, the cap on number of permits and the cap on vessel size (101 ft.), which limits the amount of effort that can be expended in a given set, will ultimately limit annual fishery effort in terms of hooks per year. We note that the number of hooks per year peaked in 2019 and has been stable or decline since then.

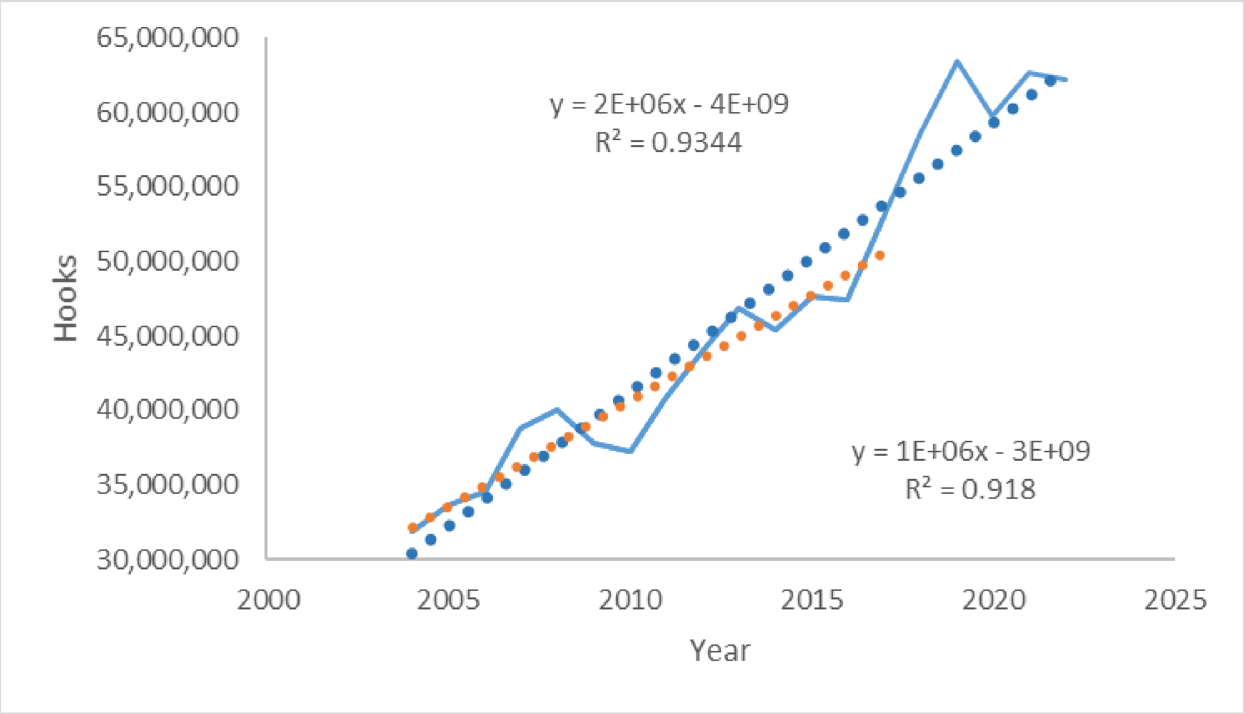

Figure 33. Hooks per year deployed by the Hawaii DSLL fishery (blue line). Orange dotted line and lower regression equation represents the relationship anticipated by NMFS (2018a); blue dotted line and upper regression equation represents the current rate of increase in hooks per year (NMFS unpublished data).

Part of the driver for the increase in hooks per year is the increase in vessels participating in the Hawaii DSLL fishery up to 2019; however, the number of hooks per vessel has also been increasing over time as well (Figure 34). Drilling down further, trips per vessel per year has not changed substantially over time and generally decreased since 2019 (Figure 34). Sets per vessel have been generally increasing since 2016 (Figure 34). Finally, hooks per set has increased over time, with 2022 hooks per set have the highest value. Hence the following factors are resulting in the increases in hooks per year in the Hawaii DSLL fishery: 1) increasing vessels over time (which will be capped at 164), although this has declined slightly since 2019; 2) increasing sets per vessel since 2016; and 3) increased numbers of hooks deployed per set.

There are no regulatory limits (although there are likely operational limits) to the second two factors impacting hooks per year and therefore we consider the possibility that the linear increase in number of hooks per year will continue into the future (Figure 35). This relationship suggests an annual increase of ~1,802,000 hooks per year (95% CI: 1,557,699 to 2,046,303). We acknowledge that there is likely some inflection point at which growth would be unlikely to occur at this same pace particularly given the limited entry and vessel size limits of this fishery; however, it is unclear when this would occur, or if it has already occurred as hooks per year has not increased since it peaked in 2019 (Figure 35), although we note that hooks per set has continued to increase since 2019 (Figure 34).

Figure 34. Effort metrics for the Hawaii DSLL fishery. Top left shows the increase in hooks per vessel from 2004 to 2022, dotted line represents a linear regression that is described by the equation. Top right shows the trends in trips per vessel (blue line) and sets per vessel (orange line) from 2004 to 2022. Bottom shows hooks per set from 2004 to 2022, dotted line represents a linear regression that is described by the equation.

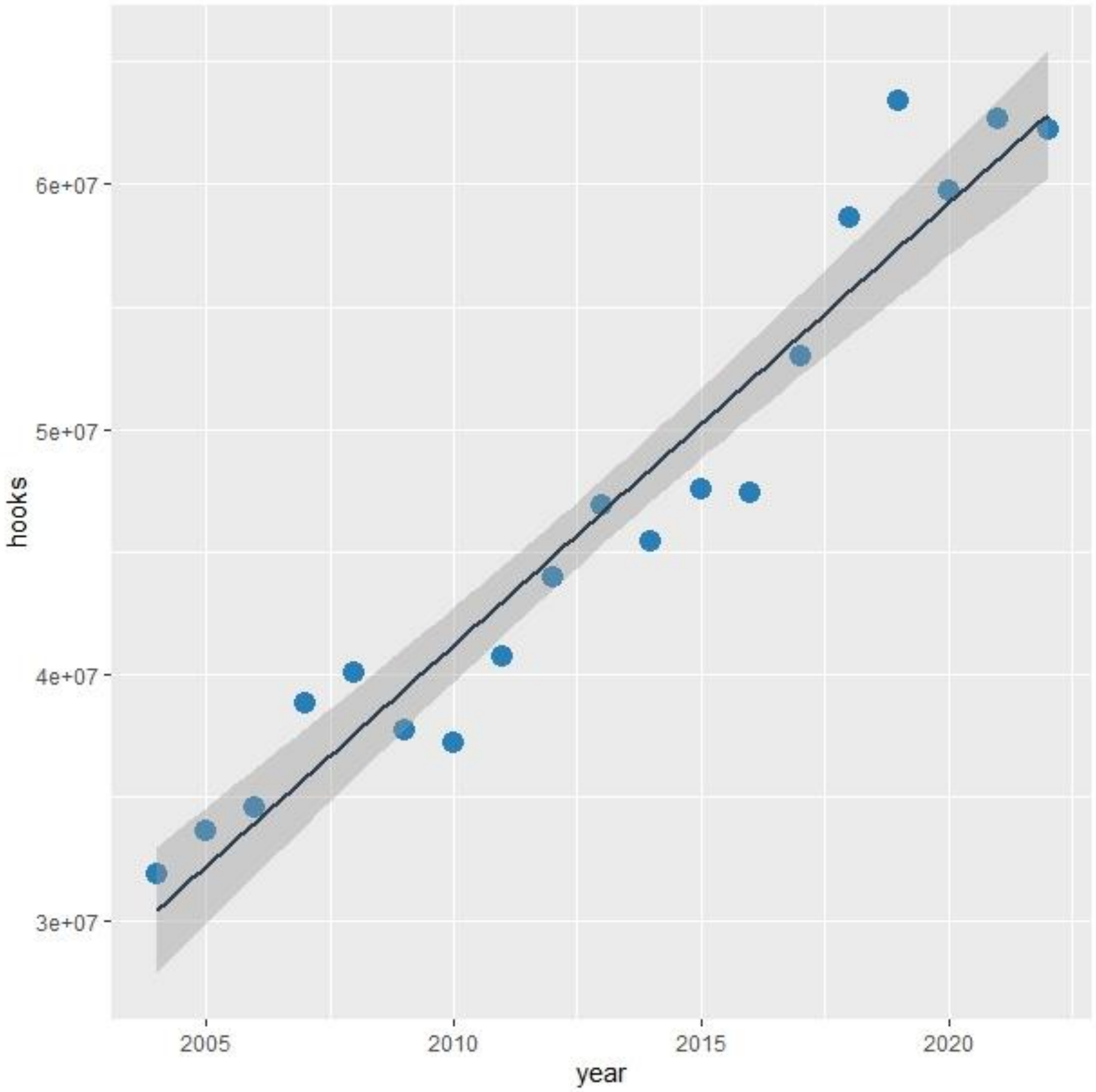

Figure 35. Trend in annual hooks per year for the DSLL from 2004 to 2022. Blue dots are the number of hooks in a given year, black line is a linear regression through the data, and the gray shaded area represents the 95% CI for the linear regression.

The confidence intervals we presented perform better than the mean, but even they would over- and underestimate the fishing activity that occurred over the past 14 years. For example, participation in 4 out of the past 14 years was greater than the upper 95% CI and was less than the lower 95% CI in 6 out of the past 14 years. The number of trips was greater than the upper 95% CI in 5 out of the past 14 years and was less than the lower 95% CI in 6 out of the past 14 years. The number of sets was greater than the upper 95% CI in 3 out of the past 14 years and was less than the lower 95% CI in 5 out of the past 14 years. The number of hooks was greater than the upper 95% CI in 5 out of the past 14 years and was less than the lower 95% CI in 6 out of the past 14 years. We also calculated 99% CIs for these estimates, which performed only marginally better at capturing all of the fishing activity over the past 14 years than the 95% intervals.[6]

Although we rely on the 95% CIs for the analyses we present in this biological opinion, it is important to note that levels of fishing activity in some years — the number of vessels that participate and the number of sets, trips, and hooks associated with those vessels — will be outside the upper and lower confidence intervals we used in the assessments presented in this biological opinion. As we noted above, mean estimates are not representative of fishing activity per se, but they split the difference between over- and underestimating levels of fishing activity *over multiple years*. Therefore, while confidence intervals should represent fishing activity in the DSLL fishery that might occur in a particular year, we turn to mean estimates to represent fishing activity over multiple years.

Although, the Pelagic FEP authorizes 164 vessels to participate in the Hawaii longline fishery and NMFS has issued all of those permits to eligible participants, not all fishing permits are actively used and have a low probability of being used[7]. However, since 2012, the number of vessels participating in the fishery has increased slightly almost every year (2016 was the exception) and has been higher than the participation anticipated in the 2014 and 2017 biological opinions. By 2017, the number of vessels participating in the fishery was about 13% higher than the participation the 2014 or 2017 opinions assumed (*n* = 128; NMFS 2014, 2017b).

Since 2010, fishing effort has increased as well over time. By 2017, fishing effort was almost 15% higher than the effort the 2014 and 2017 opinions assumed, and the number of sets was about 5% higher (NMFS 2018b). If this rate of increase continues unchanged, the fishery could potentially deploy up to 60,938,785 hooks over the next 5 years (a 14.95% increase; NMFS 2018b. That increase would represent 7,925,488 more hooks than the 2014 and 2017 biological opinions anticipated (NMFS 2018b). As the Pelagics FEP authorizes 164 vessels to participate in the Hawaii longline fisheries (DSLL or SSLL) and considering NMFS has issued all 164 permits to eligible participants; NMFS considered 164 vessels in the recent 2022 supplement (NMFS 2022b) and we consider 164 vessels in this analysis.

To summarize the historic impact of the DSLL, between 2004 and 2022, 49 giant manta rays were observed incidentally captured with an estimated 237 total interactions (McCracken 2019a;

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[6] For example, participation in the fishery also exceeded the upper 99% CI in 4 out the past 14 years, but was less than the lower 99% CI in 3 out of 14 years.

[7] Based on participation in the fishery over the past 14 years, the chances of 164 vessels participating in the DSLL fishery under the same conditions that have applied over the past 14 years is discountably small.

McCracken and Cooper 2020a, 2020b, 2021b, 2022b; NMFS 2018b). Additionally, there have been 4 documented Indo-west scalloped hammerhead sharks observed captured with an estimated total of 19 interactions and 5,668 oceanic whitetip sharks were observed, with an estimated 29,138 sharks (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021b, 2022b; NMFS 2018b).

Historically, the DSLL fishery has observed 25 green sea turtle interactions between 2004 and 2022. It's estimated 128 green sea turtles interacted with this fishery during this time frame (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a). A Bayesian mixed stock analysis indicated that the 14 turtles where genetic samples were collected from in this fishery originated mainly from rookeries in the East Pacific (mean=55.8%; 95% Credible Interval (CI) =31-80%). Approximately 20% were estimated to belong to the Central North Pacific (mean=20%; 95% CI=4-43%), the East Indian-West Pacific contributed an estimated 8.5% (mean=8.5%; 95% CI=0-28%) and 7.3% to the Southwest Pacific green sea turtle (mean=7.3%; 95% CI=0-25%). The Central West Pacific and Central South Pacific green sea turtles contributed estimated means of 2.8% (95% CI=0-18%) and 5.0% (95% CI= 0-24% respectively).

During this same time frame, 46 leatherbacks have been observed caught, with an estimated total of 246 interactions. Based on a combined mtDNA and nDNA analyses, 15 of the 16 genetic samples collected in this fishery are assigned to the western Pacific leatherback population (95.0%), and one to the East Pacific leatherback population (5.0%) (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a; NMFS 2018b). Also, between 2004 and 2022, 19 North Pacific loggerhead sea turtles were observed with an estimated total of 94 interactions (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a).

Additionally, 222 olive ridleys were observed captured, with an estimated 1,104 individuals caught between 2004 and 2022 (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a; NMFS 2018b). Analysis of collected genetic samples determined 73.3% belong to the eastern Pacific (Mexico/Costa Rica/Central America) and 26.7% western Pacific (western/Indo Pacific). Therefore, we expect overall that 809 of these olive ridleys were from either the endangered Mexico population or the East Pacific portion of the threatened olive ridley, with 295 coming from the West Pacific portion of the threatened olive ridley.

To date, there has been 1 interaction with a sperm whale since 2004 with an estimated total of 6 interactions accounting for observer coverage (McCracken 2019a). The branch line, leader, and hook remained attached to the animal after the mainline parted according to the observer's description. In addition, approximately 25-30 ft. of mainline may have been still attached resulting in a prorated serious injury determination of 0.75 using the L10 criterion (Bradford and Forney 2013; NMFS 2018b). No additional interactions have occurred through the end of 2022 since McCracken's (2019a) estimates.

Between 2004 and 2022, 123 false killer whales were observed taken in the DSLL fishery (NMFS unpublished data). In the DSLL fishery, nine false killer whales were taken within the Hawaiian EEZ through 2017. Two of those takes occurred in 2012 within the pelagic-NWHI overlap zone north of Kauai before this area was closed to longline fishing. All incidental interactions within the Hawaiian EEZ were within the range of the pelagic population, with four considered seriously injured, and three could not be determined based on the information provided by the observer. Outside of the Hawaii EEZ, one was observed dead, 12 were considered seriously injured, and four were considered not seriously injured. Of the four DSLL interactions, one occurred inside the Hawaii EEZ and was considered seriously injured, and three occurred outside the Hawaii EEZ, with one considered seriously injured, one considered not seriously injured, and one whose injury status could not be determined based on the information provided by the observer. The SEZ was closed in 2018 and 2019 after the fishery interacted with an additional 2 animals within the EEZ each year. The first SEZ closure occurred in July 2018 (83 FR 33848; July 17, 2018) and the area was closed through December 31, 2018. The second SEZ closure occurred in February 2019 (84 FR 5356; February 21, 2019) and the area was closed through August 24, 2020 (85 FR 50959; August 19, 2020; NMFS 2021c).

For the 5-yr period prior to the implementation of the Take Reduction Plan (TRP) in 2012, the average estimated mortality and serious injury to MHI IFKWs (0.21 animals per year) exceeded the PBR (0.18 animals per year; Carretta et al. 2019). Following implementation of the TRP a significant portion of the recognized population range is inside of the expanded year-round LLEZ around the MHI, providing significant protection for this population from longline fishing. Prior to that time, a seasonal contraction to the LLEZ potentially exposed a significant portion of the offshore range of the population to longline fishing. Because of the significant change in longline fishery activity relative to the MHI insular population under the TRP, the status of the population is assessed relative to the post-TRP period (2013-2015). For this period the estimate of mortality and serious injury (M/SI) (0.03) is below the PBR (0.30).

Lastly, we note that in October 2017, a 90 foot longliner (the F/V Pacific Paradise) grounded on a reef 450 yds. offshore of Waikiki within designated Hawaiian monk seal critical habitat. An emergency consultation (I-PI-18-1601-AG, PIR-2018-10305) was completed in 2018 to examine the effect of the removal and disposal of the vessel on designated monk seal critical habitat. NMFS determined that the removal and disposal of the grounded F/V Pacific Paradise was NLAA to designated monk seal critical habitat. Considering the one vessel grounding between 2004 and 2017 and all trips that occurred in the fishery during this period (20,394 trips; Table 11) results in a 4.90×10^{-3} chance a vessel may potentially ground. We discuss our expectations that vessel groundings from this fishery are NLAA in Section 9.5.14.

The effect of this continuing action on ESA-listed species is addressed in the Effects of the Action section of this biological opinion in greater detail.

3.3 Surface Vessel Traffic and Activity

The propensity of vessel strikes to go unnoticed or unreported by vessel operators impedes an accurate assessment of the magnitude this threat poses to ESA-listed species. Large container vessels represent a known threat to sea turtles and sperm whales (as well as other large whales). Vessel operated by the United States Navy and other governments are also a known threat to sea turtles and large whales. Ship strikes were identified as a source of mortality for green sea turtles in Hawaiian waters. Chaloupka et al. (2008) reported that 2.5% of green turtles found dead on Hawaiian beaches between 1982 and 2003 had been killed by ship strikes. Ship strikes have also been reported as a potentially important threat to sea turtle populations by researchers in other parts of the world including the Canary Islands (Oros et al. 2005), Italy (Caseale et al. 2010), and the Galapagos Islands (Parra et al. 2011; Denkinger et al. 2013). NMFS estimates that 250 green sea turtles are struck by vessels annually around Hawaii and the mortality for vessel strikes is 95 to 100% (NMFS 2018d).

NMFS estimated that Navy training activities in the Hawaii Range kills 20 Central North Pacific green sea turtles per year or 100 over a 5-year interval and harms approximately 7 turtles annually and 34 over a 5-year interval) due to vessel strikes which will continue to occur into the future (NMFS 2018d). The Mariana Islands training and testing activities intersects with the action area in the transit corridor of their action and they estimate that one green sea turtle (unknown species) is killed on an annual basis (NMFS 2017a).

All of the species covered in this opinion, except green sea turtles, occur mainly in pelagic waters where their density is sparse. They are large, mobile animals that spend little time near the surface. Therefore, we do not expect vessel strikes significantly contribute to the extinction risk of these species. Green sea turtles can be in high concentrations in near-shore habitats where there is overlap with high density boating. Green sea turtles are most vulnerable to small vessels (<15 m), traveling at fast rates (>10 knots) (Kelly 2020). NMFS (2008) estimated 37.5 vessel strikes of sea turtles per year from vessels of all sizes in Hawaii. More recently, we estimated as many as 200 green sea turtle strikes annually in Hawaii (Kelly 2020).

In addition to creating a risk of ship strike, much of the increase in sound in the ocean environment over the past several decades is due to increased shipping, as vessels become more numerous and of larger tonnage (NRC 2003; Hildebrand 2009; Mckenna et al. 2012). Shipping constitutes a major source of low-frequency (5 to 500 Hz) sound in the ocean (Hildebrand 2004), particularly in the Northern Hemisphere where the majority of vessel traffic occurs. While commercial shipping appears to be a primary source of anthropogenic noise pollution in the ocean, other sources of maritime traffic can also impact the marine environment. These include recreational boats, whale-watching boats, research vessels, and ships associated with oil and gas activities.

Seismic surveys, primarily for scientific research, have been conducted in the Action Area over the past several decades (NMFS 2018c). These surveys use high energy sound sources operated in the water column to probe below the seafloor with acoustic energy. Endangered and threatened sea turtles have been reported to exhibit a variety of responses when exposed to sound fields associated with seismic airguns and echosounders. Avoidance behavior and physiological responses from airgun exposure may affect the natural behaviors of sea turtles (McCauley et al. 2000). McCauley et al. (2000) conducted trials with caged sea turtles and an approaching-departing single air gun to gauge behavioral responses of green and loggerhead sea turtles. Their findings showed behavioral responses to an approaching airgun array at around two kilometers with avoidance occurring around one kilometer.

In 2018 NMFS completed a section 7 consultation on seismic surveys funded by the National Science Foundation in 2018 and 2019 and concluded that 7 North Central Pacific green, 65 leatherback, 61 North Pacific Ocean loggerhead, and 32 olive ridley sea turtles can be expected to be harassed during the survey around Hawaii (NMFS 2018c).The effect of noise produced by vessel traffic, Navy sonar, seismic surveys and other low-frequency sounds on endangered and threatened sharks and rays is generally unknown.

These species may be impacted by anthropogenic sound in various ways. It can produce direct physical harm or may illicit behavioral responses including, but not limited to, cessation of feeding, resting, or social interactions, changes in habitat to avoid areas of higher sound levels, or changes in diving behavior (MMC 2007). Although anthropogenic sounds may have adversely affected all the species considered herein, we expect that such sounds have contributed and will continue to contribute to the declines of cetaceans in the Action Area into the future. In contrast the effect of vessel traffic and ship strikes tends to be more apparent in coastal areas, and have contributed to the current status of the sea turtle species that exhibit more coastal distributions like green sea turtles.

3.4 Pollution and Marine Debris

Many different types of pollution can adversely affect endangered and threatened species and habitats within the Action Area. There are three main categories of marine pollution: oil pollution, contaminants and pesticides, and marine debris. In this section, we describe these three pollution categories, the, exposure pathways and anticipated effects on endangered and threatened resources.

Hawaii's geographic isolation results in more energy requirements than it produces. More than four-fifths of Hawaii's energy comes from imported petroleum, making it the most petroleum-dependent state in the nation (U.S. EIA 2019; EPA 2022). Imported crude oil is refined at the PAR Pacific’s refinery on Oahu for distribution and use (Figure 36). Total petroleum imports ranged from 54 million barrels in 2006 to approximately 30 million barrels in 2017 (State of Hawaii 2019) with significant increases in domestic petroleum sources attributed to this rate of decline in use (EPA 2022).

Past oil spills that impacted the offshore marine environment of Hawaii within the Action Area are listed in as reported by the EPA (2022). Where available, information on the impacts to natural resources is summarized.

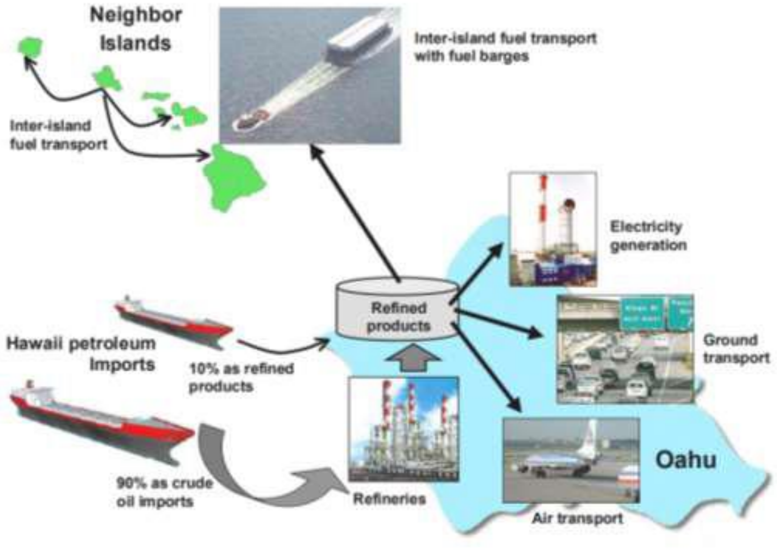


Figure 36. Hawaii Fuel Supply System (Hawaii Department of Transportation 2013; EPA 2022).

Table 12. Important historical and recent spills in the Action Area. From the Hawaii Area Plan Section 9000-9 (October 2015) except where otherwise noted (EPA 2022).

Date	Spill Name/Location	Oil Type and Volume	Natural Resource Impacts
05/13/87	Chevron Pipeline Oahu: Wildlife Refuge adjacent to Pearl Harbor	104,496 gallons Jet A fuel.	Hawaiian Stilt, Hawaiian duck and fish kill
01/20/87	Tank Barge Hana Offshore Oahu en route to Kahalui, Maui	42,000 gallons spilled; Bunker C.	Minimal impacts wildlife
03/02/89	M/V *Exxon Houston* Oahu: Germaines Luau Beach and Campbell Industrial park.	25,000 gallons crude spilled 92,000 gallons Bunker C fuel oil.	Impacts to wildlife were minimal.
11/06/90	T/V *Star Connecticut* Oahu: Barbers Point	Potential for 250,000 barrels; 0 discharged. Various refined products.	None
11/20/91	T/V *Yupex* Oahu: Honolulu Harbor	25,000 gallons diesel fuel	None
05/14/96	Chevron Pipeline oil spill Oahu: Waiau Stream and Pearl Harbor	41,244 gallons No. 6 fuel oil	Not reported
08/24/98	Tesoro Hawaii Corporation tanker Oahu: Barbers Point traveling to Kauai	4,914 gallons Bunker C fuel (Tesoro Oil Spill Natural Resource Trustees 2000).	Shoreline and waters of Kauai; 5 Hawaiian monk seals reported exposed to oil (animals were not assessed for injuries by NMFS, however 2 reported with potential significant respiratory concerns;

Date	Spill Name/Location	Oil Type and Volume	Natural Resource Impacts
			Tesoro Oil Spill Natural Resource Trustees 2000; NMFS 2007); 105 seabirds of various species *observed* (Tesoro Oil Spill Natural Resource Trustees 2000).
Recent Incidents (last 10 years) from NRC Reports			
09/10/13	Molasses Spill Oahu: Honolulu Harbor	233,000 gallons molasses	Various coral species, fish kills, other bottom crustaceans/creatures
01/16/15	Tug *Nalani* Oahu: Barbers Point	75,000 gallons diesel	Not reported
05/24/16	*USS HOPPER* 50 miles southwest of Barbers Point	2,000 gallons fuel oil 1D	Not reported
10/11/17	F/V *Pacific Paradise* Oahu: Kaimana Beach in Waikiki	1,500 gallons of fuel and oily water mix	Not reported

Fish (i.e. elasmobranchs) rely on passing water over their gills to respire and are the major route of exposure as they intake water (and pollutants including hydrocarbons, surfactants, pesticides, etc.) through these anatomical structures. Gills also play an important part in the regulation of mucous responsible for ion regulation (Agamy 2013a).

As discussed by Agamy (2013a), responses to chemical exposure through the gills includes "epithelial hyperplasia with lamellar fusion, epithelial hypertrophy, telangiectasia, edema with epithelial lifting, mucous and chloride cells proliferation and epithelial desquamation are typical histopathological lesions of gills in response to a wide range of contaminants, including oil compounds (Reviewed in Wood 2001; Au 2004)". Edema, cell lifting, shortening, lesions, hemorrhages, tissue necrosis, and alterations in blood chemistry values occur due to exposure as well (Agamy 2013a, 2013b). All of these modifications to gill epithelial tissue can result in deleterious gas exchange functions and decrease oxygen consumption in an exposed individual. Some studies have shown increased rates in mortality depending on the concentration of chemical exposure (for instance, Agamy 2013a). Additionally, PAHs are carcinogenic to fish and persist in their tissues for weeks after oil exposure (Danion et al. 2011) with immunosuppression occurring quickly after oil exposure (Omar-Ali et al. 2015).

PAHs are environmental contaminants that are naturally occurring and come from multiple anthropogenic sources, including oil spills. Some PAHs are carcinogenic, mutagenic (cause developmental abnormalities), genotoxic, immunotoxic, cause adverse respiratory effects, and larval mortality (U.S. Department of Health and Human Services 1995; Mastrangelo et al. 1996; Incardona et al. 2004; Bechmann et al. 2010; Balciog˘lu 2016; Zychowski and Godard-Codding 2017). PAHs can bioaccumulate in prey items leading to toxic effects in those species (Moore et al. 1989) which affect sea turtles (Camacho et al. 2012). Some PAHs provide serious adverse effects in the marine environment over 17 months after an oil spill event (Perez et al. 2008) and the EPA designated 16 PAHs as High Priority Pollutants because of their potential toxicity, prevalence, and persistence within the environment (40 CFR Part 423, Appendix A; Hussar et al. 2012). Therefore, PAHs are also considered Persistent Organic Pollutants, or POPs. PAHs appear to show greater partitioning and storage in tissues than other categories of POPs with some species at greater risk of long-term bioavailability although varying risks given life stage and sex (Munoz et al. 2021).

Additionally, POPs are maternally transferrable in some species, like sea turtles, and affect neonates (Ehsanpour et al. 2014; De Andres et al. 2016; Munoz and Vermenien 2020 and references cited therein). Barraza et al. (2020) showed that location specific variables can influence POP loading in an individual and may vary between populations given high site fidelity, even amongst close geographic populations. However, although poorly researched, the literature also suggests that species higher in the trophic chain have some ability to metabolize *some* PAHs. Long-term fitness impacts may occur from chronic exposure that can ultimately cause more harm to populations (Camacho et al. 2012). Hutchinson and Simmonds (1992) suggested fibropapillomatosis may also be influenced by PAH exposure in sea turtles while long-term effects from chronic PAH exposure can result in cancers (Collins et al. 1998; Camacho et al. 2012). However, data linking PAHs and POPs to cancers in elasmobranchs is lacking.

External oil exposure can result in coating of the entire body including epidermal surfaces where absorption occurs altering hematology and blood chemistry values (Vargo et al. 1986 and Lutcavage et al. 1995 as cited in Michelmore et al. 2017). Additionally, alterations to dermal characteristics through cellular modifications increases risk of infection in an individual, although long-term effects are unknown (Lutcavage et al. 1995). Lastly, Bembenek-Bailey et al. (2019) found that there were substantial alterations in metabolites that suggest crude oil and oil/dispersant exposure can lead to energy depletion and oxidative stress in those animals (EPA 2022). Synergistic effects of oil and dispersants are not well known, however have been documented by Adams et al. (2014) to affect salt gland, renal, and adrenal functions by increasing exposure to oil hydrocarbons.

PAHs adversely affect sea turtle embryos and hatchlings as well (CRRT Response Technologies Committee 2015 as cited by NMFS 2016c). Sea turtle nests are suspected to suffer the greatest effect due to the impacts of oil exposure on developing embryos and hatchling success. However, information in the literature is inconsistent. Lutcavage et al. (1995) reported observed effects of loggerhead sea turtles exposed to weathered crude oil that included alteration of blood chemistry, respiration and diving patterns, interference with salt gland functioning, and skin lesions (NMFS 2016). Lutcavage et al. (1995) further hypothesized that exposure to fresh oil would have been more harmful to the animals (NMFS 2016c). These effects are likely to apply to other species of sea turtle species under consideration and likely increase opportunities for predation and disease after exposure. Yet, Wallace et al. (2020) also discuss various exposure studies that had no effects to eggs or hatchlings, including studies in surrogate land species of turtles.

Studies on exposure to oil and dispersed oil in the available literature have mixed results from no effects to observed effects. Stacy (2012) and Stacy et al. (2017) did not observe effects noted in the literature by previous authors (Vargo et al. 1986; Lutcavage et al. 1995; Camacho et al. 2013; Harms et al. 2014). Yet later studies better address exposure to dispersants which resulted in statistically significant alterations for 15 different hematological variables associated osmotic, electrolyte, mineral, and hydration; which ultimately affected hatchling weights (Harms et al. 2019). Shaver et al. (2021) on the other hand documented increased abnormalities in embryos and nesting trends in the same population assessed by Stacy (2012) and Stacy et al. (2017).

Additionally, Harms et al. (2019) noted indications of a hypothalamic-pituitary-adrenal stress response and potential hepatocellular toxicity. The authors discuss that some results were inconsistent with previous findings particularly, in relation to anemia and hematological assessment methods used by previous authors. Ultimately, Harms et al. (2019) suggests anemia is likely a result of chronic exposure to dispersants and chemically dispersed oil. Three values suggested considerable concern for morbidity or potential pending mortality including uric acid, potassium, and phosphorus (Harms et al. 2019). Exposure to sea turtle hatchlings would be less likely given the number of nests expected to occur in Hawaiian waters during any given year when compared to adults in the Region.

In a productivity-susceptibility analysis and associated maximum entropy (MaxEnt) modeling approach, Romo-Curiel et al. (2022) assessed oil spill exposure to 24 large pelagic fish species in the Gulf of Mexico, given their productivity and susceptibility to exposure to determine potential for recovery. Both the oceanic whitetip and giant manta were species considered in their analysis. The oceanic whitetip shark had a medium vulnerability ranking and the giant manta ray had a high vulnerability ranking (Table 2 and Figure 2 in Romo-Curiel et al. 2022). The authors then used these rankings in conjunction with a hotspot analysis to determine an overall ranking given three geographic scenarios.

Ultimately, the oceanic whitetip shark had a low ranking whereas the giant manta ray ranked medium across all scenarios in deep oceanic waters. We recognize, we do not have rankings for portion of the species' population(s) which occur in the Central or greater Pacific. However, we can consider these rankings applicable in our Region since it is focused on the biological factors of the species, and because the areas considered (i.e., the GOM versus the Action Area) are both deep pelagic waters, and the two species preferred depth ranges across ocean basins is the same (Miller and Klimovich 2017; Young et al. 2017).

An elasmobranch or sea turtle's response to oil or contaminant exposure will be dependent on a large number of variables. However, length of exposure, concentration of the contaminant(s), and the specific type contaminant(s) are the most influential. Currently, due to the lack of available information in the literature, we cannot quantify the number of animals that may be exposed to releases that have previously, or may occur, in the Action Area.

Marine debris has become a widespread threat for a wide range of marine species that are increasingly exposed to it on a global scale. Plastic is the most abundant material type worldwide, accounting for more than 80% of all marine debris (Poeta et al. 2017). The most common impacts of marine debris are associated with ingestion or entanglement and both types of interactions can cause the injury or death of animals of many different species. Ingestion occurs when debris items are intentionally or accidentally eaten (e.g., through predation on already contaminated organisms or by filter feeding activity, in the case of large filter feeding marine organisms, such as whales) and enter in the digestive tract. Ingested debris can damage digestive systems and plastic ingestion can also facilitate the transfer of lipophilic chemicals (especially persistent organic pollutants) into the animal's bodies.

An estimated 640,000 tons of fishing gear is lost, abandoned, or discarded at sea each year throughout the world's oceans (Macfadyen et al. 2009). These "ghost nets" drift in the ocean and can fish unattended for decades (ghost fishing), killing large numbers of marine animals through entanglement. Fisheries observers in the Hawaii longline fisheries (SSLL and DSLL) document marine debris that is encountered during fishing trips. From 2008-2015 they documented 1,426 debris encounters with marine debris. Over 75% of the debris encountered consisted of nets and tangled debris. Almost 85% of debris was collected and brought back to the docks; this amounted to over 211,838 pounds of debris (NOAA 2015).

Of the different types of pollution that can adversely affect the sea turtles, ingestion and entanglement risk of marine debris may pose the most serious threat. The sub-tropical convergence zone (STCZ), a known area of marine debris aggregation (Kubota 2004; Pichel et al. 2007; Maximenko et al. 2012), is within the Action Area. Due to increased biological productivity in the STCZ, it has become a significant foraging and migration corridor for swordfish (*Xiphias gladius*) (Seki et al. 2002) and sea turtles (Polovina et al. 2004; Howell et al. 2008, 2010). The early developmental stages of all turtle species are spent in the open sea. During this time both juvenile turtles and their buoyant food are drawn by advection into fronts (convergences, rips, and drift lines).

The same process accumulates large volumes of marine debris, such as plastics and lost fishing gear, in ocean gyres (Carr 1987). An estimated four to twelve million metric tons of plastic enter the oceans annually (Jambeck et al. 2015. It is thought that some sea turtles eat plastic because it closely resembles jellyfish, a common natural prey item (Schuyler 2014). Ingestion of plastic debris can block the digestive tract which can cause turtle mortality as well as sub-lethal effects including dietary dilution, reduced fitness, and absorption of toxic compounds (Lutcavage et al. 1997; Laist et al. 1999).

Santos et al. (2015) found that a small amount of plastic debris was sufficient to block the digestive tract and cause death in sea turtles. They reported that 10.7% of green turtles in Brazilian waters were killed by plastic ingestion, while 39.4% had ingested enough plastic to have killed them. These results suggest that debris ingestion is a potentially important source of turtle mortality, one that may be masked by other causes of death. Gulko and Eckert (2003) estimated that between one-third and one-half of all sea turtles ingest plastic at some point in their lives. A more recent study by Schuyler et al. (2015) estimates that 52% of sea turtles globally have ingested plastic debris. Schuyler et al. (2015) synthesized the factors influencing debris ingestion by turtles into a global risk model, taking into account the area where turtles are likely to live, their life history stage, the distribution of debris, the time scale, and the distance from stranding location. They found that oceanic life stage turtles are at the highest risk of debris ingestion. Based on this model, olive ridley turtles are the most at-risk species; green, loggerhead, and leatherback turtles were also found to be at a high and increasing risk from plastic ingestion (Schuyler 2014).

The regions of highest risk to global turtle populations are off the east coasts of the United States, Australia, and South Africa, the East Indian Ocean, and Southeast Asia. In addition to ingestion risks, sea turtles can also become entangled in marine debris such as fishing nets, monofilament line, and fish-aggregating devices or FADs (NRC 1990a, 1990b; Lutcavage et al. 1997; Laist et al. 1999). Turtles are particularly vulnerable to ghost nets due to their tendency to use floating objects for shelter and as foraging stations (Kiessling 2003; Dagorn et al. 2013).

Marine mammals are also particularly susceptible to the threats associated with marine debris and many cases of ingestion and entanglement have been reported around the world (Poeta et al. 2017). Baulch and Perry (2014) found that the proportion of cetacean species ingesting debris or becoming entangled in debris is increasing. Based on stranding data, they found that recorded rates of ingestion have increased by a factor of 1.9 and rates of entanglement have increased by a factor of 6.5 over the last forty years (1970-2010). Ingestion of marine debris can also have fatal consequences for large whales.

In 2008, two male sperm whales stranded along the northern California coast with large amounts of fishing net scraps, rope, and other plastic debris in their stomachs. One animal had a ruptured stomach, the other was emaciated, and gastric impaction was suspected as the cause of both deaths (Jacbosen et al. 2010). Additionally, in February 2023, a sperm whale stranded in Kauai and marine debris was considered a contributing factor in its death (University of Hawaii News 2023). According to marine debris data compiled in NOAA's 2014 Marine Debris Program Report, three Guadalupe fur seals were reported to have evidence of entanglement; one with net markings, another with a polyfilament line around its neck, and one with hook and line (Hanni et al. 1997). Once entangled, Guadalupe fur seals may drag and swim with gear attached for long distances, ultimately resulting in fatigue, compromised feeding ability, or severe injury, which may lead to reduced reproductive success and death.

Plastics within the marine environment are also a threat to manta ray species. Filter feeders such as the giant manta ray are particularly susceptible to ingesting high levels of microplastics (Germanov 2018) and being exposed to toxins (Worm et al. 2017), due to their feeding strategies (Paig-Tran et al. 2013) and target prey (Setala et al. 2014). Jambeck et al. (2015) found that the Western and Indo-Pacific regions are responsible for the majority of plastic waste. These areas also happen to overlap with some of the largest known aggregations for giant manta rays.

Giant manta rays must filter hundreds to thousands of cubic meters of water daily to obtain adequate nutrition (Paig-Tran et al. 2013), therefore, they can ingest microplastics directly from the water or indirectly through their contaminated planktonic prey (Setala et al. 2014). Not only can microplastics prohibit adequate nutrient absorption and physically damage the digestive track (Germanov et al. 2018), they can harbor high levels of toxins and persistent organic pollutants and transfer these toxins to the animal once ingested (Worm et al. 2017). These toxins are known to bioaccumulate and have been shown to alter the functioning of the endocrine system of aquatic animals (Rochman et al. 2014). In addition, these toxins can be passively transferred from mother to embryo through yolk or milk production (Lyons et al. 2013), and species that have delayed sexual maturity, have more opportunities to accumulate toxins and are expected to offload higher levels of contaminants to their offspring (Lyons et al. 2013).

Plastic additives and persistent organic pollutants have been found in the muscle of basking sharks (Fossi et al. 2014), the blubber of fin whales (Fossi et al. 2012, 2014) and the skin of whale sharks (Fossi et al. 2017). However, studies have yet to confirm that filter feeders are directly affected by microplastic ingestion and plastic-associated toxins and additives (Germanov et al. 2018). While the ingestion of plastics is likely to negatively impact the health of the species, the levels of microplastics in manta ray feeding grounds, frequency of ingestion and the transfer of toxins are presently being studied to evaluate the impact on these species (Germanov 2015a, 2015b).

Marine debris may entangle or be ingested by oceanic whitetip sharks (Compango 1984), leading to injury or possibly starvation, and derelict fishing gear may cause entanglement and possibly drowning. However, data are not available to estimate the number of oceanic whitetip mortalities resulting from marine debris in the Action Area.

It is difficult to assess the precise impact pollution and marine debris have had on ESA-listed species under consideration. However, we are reasonably certain they have contributed to the declines in these species.

3.5 Synthesis of the Environmental Baseline

The listed resources considered in this biological opinion have been exposed to a wide variety of the past and present state, federal, and private actions in the Action Area, which includes of all proposed federal projects in the Action Area that have already undergone formal or early consultation, and state or private actions that are contemporaneous with this consultation. While we have quantitative data for many fishery activities, the impact of other activities, such as vessel traffic, pollution and marine debris, on the status, trend or the demographic processes of threatened and endangered species is largely unknown. Some of these are likely to have had and will continue to have lasting effects on the endangered and threatened species considered in this consultation.

The preceding section of this biological opinion addresses global climate change, fisheries and fisheries bycatch, and pollution from chemicals and marine debris, and effects these stressors have on listed resources. Some of these stressors have resulted in mortality or serious injury to individual animals (e.g., fishing), whereas other stressors (e.g., pollution) may induce sub-lethal responses like changes in behavior that could impact important biological functions such as feeding or breeding. Of the stressors considered herein, the cumulative effect of fisheries in the Action Area likely has had the greatest impact on the listed species considered herein, and the populations that comprise those species.

4 EFFECTS OF THE ACTION

Effects of the Action are all consequences to listed species or critical habitat that are caused by the proposed action, including the consequences of other activities that are caused by the proposed action. A consequence is caused by the proposed action if it would not occur but for the proposed action and it is reasonably certain to occur. Effects of the action may occur later in time and may include consequences occurring outside the immediate area involved in the action (50 CFR 402.02).

As we described in the Approach to the Assessment section of this biological opinion, we organize our effects' analyses using a stressor identification - exposure – response – risk assessment framework. The Integration and Synthesis section of this opinion follows the Effects of the Action and integrates information we presented in the Status of Listed Resources and Environmental Baseline sections of this biological opinion with the results of our exposure and response analyses to estimate the probable risks the proposed action poses to endangered and threatened species. Because NMFS has previously concluded that the proposed action is not likely to adversely affect several listed species and areas designated as critical habitat for listed species, these listed resources are not considered in the analyses that follow. Species and critical habitat not likely to be adversely affected by the proposed action are discussed in Section 2.1 of this biological opinion, Status of Listed Resources Not Considered Further and in Appendix A.

4.1 Potential Stressors

Potential stressors associated with the proposed action include:

1. Interaction with gear including capture of non-target species, such as listed species or their prey
2. Derelict gear
3. Introduction of oily discharges, cardboard, plastics, and other waste into marine waters
4. Collisions with vessels
5. Vessel noise
6. Vessel groundings
7. Vessel emissions

We determined that vessel noise, collisions with vessel, introduction of discharges and other wastes, derelict gear, vessel groundings, and vessel emissions are stressors that are not likely to adversely affect any species. Discussion of these conclusions can be found in Section 9/Appendix A (See Section 2.1; Listed Resources Not Considered Further and Appendix A for more details). As a result, in this section we focus on the stressor of interaction with fishing gear. This stressor results in hooking and entanglement of listed species and are thus likely to adversely affect them through mortality and injury, including latent mortality resulting from injury. We also briefly discuss slipped or unobserved catch, and depredation of bait and catch.

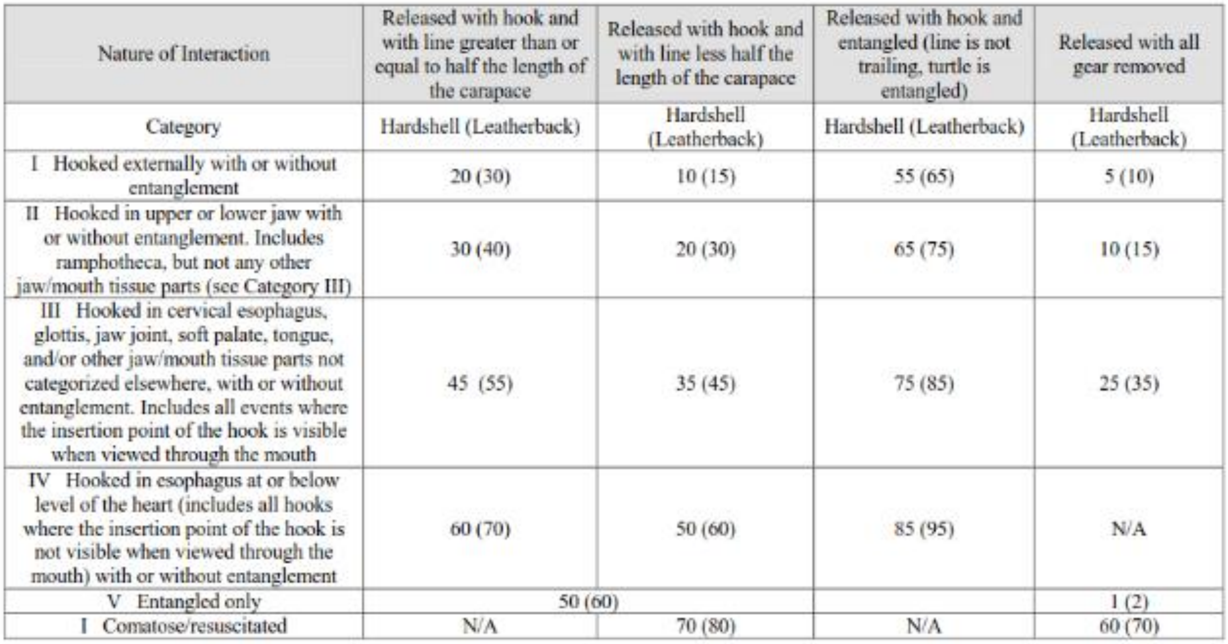
4.2 General Responses of ESA-listed Species to the Fishery

The most significant hazard the DSLL fishery presents to listed species results from hooking and entanglement by gear which can injure or kill sea turtles, elasmobranchs, and cetaceans. If hooked or entangled, air-breathing species can drown after being prevented from surfacing for air; alternatively, all listed species that are hooked or entangled, may not immediately die from their wounds but can suffer impaired swimming or foraging abilities, altered migratory behavior, and altered breeding or reproductive patterns, and latent mortality from their interactions.

Although survivability studies have been conducted on some listed species captured in longline fisheries, long-term effects are nearly impossible to monitor; therefore, a quantitative measure of the effect of longlining on sea turtles, sperm whales, MHI IFKW, Indo-west Pacific scalloped hammerhead shark, oceanic whitetip shark, and giant manta ray populations is very difficult. Even if listed species are not injured or killed after being entangled or hooked, these interactions can be expected to elicit stress- responses that can have longer-term physiological or behavioral effects. The following discussion summarizes the information on how sperm whales, MHI IFKW, sea turtles, Indo-west Pacific scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays are likely to respond to these interactions with fishing gear.

For sea turtles, we used Ryder et al. (2006) to assign post-interaction mortality rates to sea turtles that were alive when they were brought alongside the fishing vessel. Ryder et al. (2006) assign post-interaction mortality rates based on the location of the hook, whether the turtle was released with or without the hook, the amount of trailing line remaining at release, and whether or not the turtle was entangled. Table 13 specifies the release conditions and hook locations for the associated post-interaction mortality rates.

Table 13. Post-interaction mortality rates for sea turtles interacting with longline fisheries from Ryder et al. (2006)*.



Nature of Interaction	Released with hook and with line greater than or equal to half the length of the carapace	Released with hook and with line less half the length of the carapace	Released with hook and entangled (line is not trailing, turtle is entangled)	Released with all gear removed
Category	Hardshell (Leatherback)	Hardshell (Leatherback)	Hardshell (Leatherback)	Hardshell (Leatherback)
I Hooked externally with or without entanglement	20 (30)	10 (15)	55 (65)	5 (10)
II Hooked in upper or lower jaw with or without entanglement. Includes ramphotheca, but not any other jaw/mouth tissue parts (see Category III)	30 (40)	20 (30)	65 (75)	10 (15)
III Hooked in cervical esophagus, glottis, jaw joint, soft palate, tongue, and/or other jaw/mouth tissue parts not categorized elsewhere, with or without entanglement. Includes all events where the insertion point of the hook is visible when viewed through the mouth	45 (55)	35 (45)	75 (85)	25 (35)
IV Hooked in esophagus at or below level of the heart (includes all hooks where the insertion point of the hook is not visible when viewed through the mouth) with or without entanglement	60 (70)	50 (60)	85 (95)	N/A
V Entangled only	50 (60)			1 (2)
I Comatose/resuscitated	N/A	70 (80)	N/A	60 (70)

* Numbers in the table are the percent of turtles with the corresponding injury and release condition expected to die. For example, a loggerhead sea turtle is a hardshell turtle, and if one is hooked in a flipper (externally hooked) and released with the hook and an amount of line equivalent to less than half of its carapace length, there is a 10% chance that it will die from its injury. A leatherback sea turtle with the same injury and release condition would have a 15% chance of dying.

4.2.1 Entanglement in Longline Gear

Sea Turtles

Sea turtles are particularly prone to being entangled in fishing gear because of their body configuration and behavior. Records of stranded or entangled sea turtles reveal that fishing debris can wrap around the neck, flippers, or body of a sea turtle and severely restrict swimming or feeding. Over time, if the sea turtle is entangled when young, the fishing line will become tighter and more constricting as the sea turtle grows, cutting off blood flow, causing deep gashes, some severe enough to remove an appendage. Sea turtles have also been found with trailing gear that has been snagged on the bottom, thus causing them to be anchored in place (Balazs 1985).

Sea turtles have been found entangled in branch lines, mainlines and float lines. Longline gear is fluid and can move according to oceanographic conditions determined by wind and waves, surface, and subsurface currents, etc.; therefore, depending on both sea turtle behavior, environmental conditions, and location of the set, turtles could be entangled in longline gear. Entanglement in monofilament line or polypropylene (float line) could result in substantial wounds, including cuts, constriction, or bleeding on any body part. In addition, entanglement could directly or indirectly interfere with mobility, causing impairment in feeding, breeding, or migration. Sea turtles entangled by longline gear are most often entangled around their neck and fore flippers.

Sharks

Although most sharks tend be hooked by longline gear, they can sink the gear as they dive and if they begin rolling, can become entangled in the monofilament branch lines and mainline. An entanglement as such, could cause the shark to perish if it is unable to circulate water through its gills. The literature on sharks captured on longline gear is primarily focused on the effects of hooking, post release handling, and post hooking mortality, not entanglement in longline gear. However, marine debris data compiled in NOAA's 2014 Marine Debris Program Report reveals several accounts of sharks entangled in natural fiber rope and monofilament (NOAA Marine Debris Program 2014). A shortfin mako shark entangled in natural fiber rope, resulted in scoliosis, abrasions and was undernourished (Wegner and Cartamil 2012) and the monofilament found encircling a blacknose shark caused its spine to be deformed (Schwartz 1984). In general, entanglement could directly or indirectly interfere with the shark's mobility, causing impairment in feeding, breeding, or migration.

Giant Manta Ray

Similar to sea turtles, when giant manta rays interact with longline gear, they are particularly prone to being entangled in fishing gear because of their body configuration and behavior. The giant manta ray tends to be more vulnerable to entanglement and foul hooking as opposed to being hooked in the mouth (Sales et al. 2010; Domingo et al. 2012). If entangled in a monofilament branch line or polypropylene float line, the giant manta ray is at risk of severing of the cephalic and pectoral fin, severe injuries that can lead to a reduction in feeding efficiency and even death.

There is very little information on the evidence and impact of entanglement on the giant manta ray. However, there are data regarding the reef manta which is a reasonable surrogate species since, prior to 2009, the two manta species are so similar that they were categorized as a single species. Surveys of the reef manta from 2005-2009 at an aggregation site off Maui, Hawaii, revealed that 10% of the population had an amputated or non-functional cephalic fin (Deakos et al. 2011). Most of these injuries were attributed to entanglement in fishing line (most likely from recreational or nearshore fisheries) since the straight edge cut of all amputated cephalic fins resemble the severing effects of monofilament (Deakos et al. 2011). In fact, eight individuals had physical evidence of entanglement with fishing line; two individuals had hooks in the cephalic fin, two had monofilament wrapped around the cephalic fin and another two had scars from previous line entanglements, and two individuals had line that had begun to cut part way through the cephalic fin (Deakos et al. 2011).

Deakos et al. (2011) observed that individuals in this population with an amputated cephalic fin appeared healthy, however, considering the function of the cephalic fin to guide food into the manta's mouth, feeding efficiency is most likely reduced, and the absence of this fin may negatively affect size, growth rate and reproductive success. Lastly, Deakos et al. (2011) report that videos show two reef manta rays in Hawaii, which were entangled in mooring lines, perish, and become immediately consumed by sharks. Although mooring lines are not used in this fishery, the material is similar to polypropylene float line, entangles these animals, and shows that predators are quick to take advantage of an entangled animal.

Cetaceans

Entanglements with fishing gear occur when whales encounter lines and other parts of the gear that may be suspended in the water column. Often these encounters do not lead to entanglements. However, sometimes a portion of the gear, or a loop in the line, may catch on a body part of an animal. If that happens, then the gear may begin to wrap around the body (including the head/mouth area, pectoral and dorsal fins, and the tail/fluke region) as the animal moves through the water. Reacting to the contact of the gear, or the drag created by the gear, the animal may perform quick and powerful changes in directions or body movements attempting to shake the gear free. The specific dynamics of gear encounters that may be more likely to lead to an entanglement are not well documented, as very few entanglements have been witnessed in the initial stages. Furthermore, if the line parts, it may result in an animal trailing gear, or the animal can free themselves altogether. Often, we are unsure what causes a line part, so these interactions are unobserved. However, forensic review of entanglement cases suggests that knots/splices/leads and other potential sources of snags, along with loose or slack lines, are likely contributors to numerous entanglements. Other possible contributors include animal behavior (whether they encounter the gear while foraging or migrating) and ocean conditions (e.g., current, tide, wind) as well as the condition and/or life stage of the animal.

4.2.2 Hooking

Sea Turtles

In addition to being entangled in a longline, sea turtles are also injured and killed by being hooked. Sea turtles are either hooked externally - generally in the flippers, head, beak, or mouth - or internally, where the animal has attempted to forage on the bait, and the hook is ingested into the gastrointestinal tract, often a major site of hooking (E. Jacobson in Balazs et al. 1995a). Hooking can occur as a result of a variety of scenarios, some of which will depend on foraging strategies and diving and swimming behavior of the various species of sea turtles. For example, necropsied olive ridleys have been found with bait in their stomachs after being hooked;

therefore, they most likely were attracted to the bait and attacked the hook. Similarly, a turtle could concurrently be foraging in or migrating through an area where the longline is set and could be hooked at any time during the setting, hauling, or soaking process.

When a sea turtle is hooked and the hook is removed, which is often possible with a lightly hooked turtle, the hooking interaction is likely to result in injuries that can, in some cases, lead to death. The risk of mortality from hooking increases if the hook is lodged internally. Like most vertebrates, the digestive tract of the sea turtle begins in the mouth, through the esophagus, and then dilates into the stomach. The esophagus is lined by strong conical papillae, which are directed caudally towards the stomach (White 1994). The existence of these papillae, coupled with the fact that the esophagus snakes into an S-shaped bend further towards the tail make it difficult to see hooks, especially when deeply ingested. Not surprisingly, and for those same reasons, a deeply ingested hook is also very difficult to remove without significant injury to the animal. The esophagus is attached firmly to underlying tissue; therefore, when a hook is ingested, the process of movement, either by the turtle's attempt to get free of the hook or by being hauled in by the vessel, can traumatize the internal organs of the turtle, either by piercing the esophagus, stomach, or other organs, or by pulling the organs from their connective tissue. Once the hook is set and pierces an organ, infection may ensue, which may result in the death of the animal.

If a hook does not become lodged or pierce an organ, it can pass through to the colon, or even be expelled through the turtle (E. Jacobson in Balazs et al. 1995a). In such cases, sea turtles can pass hooks through the digestive tract with little damage (Work 2000). Of 38 loggerheads deeply hooked by the Spanish Mediterranean longline fleet and subsequently held in captivity, six loggerheads expelled hooks after 53 to 285 days (average 118 days; Aguilar et al. 1995). If a hook passes through a turtle's digestive tract without getting lodged, the chances are good that less damage has been done. Tissue necrosis that may have developed around the hook may also get passed along through the turtle as a foreign body (E. Jacobson in Balazs et al. 1995a).

Sharks

Sharks are incidentally captured when they bite baited hooks or depredate on catch. Injuries to sharks from longline hooks can be external-generally in the mouth, jaw, gills, roof of mouth, tail and fin or ingested internally, considered deeply hooked or gut-hooked.

As with other marine species, even if the hook is removed, which is often possible with a lightly hooked shark, the hooking interaction is believed to be a significant event. As previously mentioned, capture on a longline is a stressful experience. On average soak times in the DSLL are approximately 21 hours and may last longer. During capture, the amount of water flow over the gills is limited and biochemical recovery can take up to 2 to 7 days, and even longer for injured sharks (Campana et al. 2009). In addition, sharks are vulnerable to predation while being captured due to their restricted mobility, and after their release due to exhaustion and injury. Furthermore, handling procedures can cause additional damage (e.g., cutting the jaw, tail, gaffing, etc.), stress, or death.

A gut-hooked shark is at risk of severe damage to vital organs and excessive bleeding. Campana et al. (2009) found in a post-release mortality study that 33% of tagged blue sharks with extensive trauma such as a gut-hooking perished. Campana et al. (2009) attribute rapid post-release mortality of sharks to occur because of the trauma from the hooking rather than any interference with digestion or starvation.

Giant Manta Ray

The giant manta ray primarily feeds on planktonic organisms such as euphausiids, copepods, mysids, decapod larvae and shrimp, but some studies have noted their consumption of small and moderate sized fishes (Bigelow and Schroeder 1953; Carpenter and Niem 2001 as cited in Miller and Klimovich 2017; The Hawaii Association for Marine Education and Research Inc. 2005). Due to its foraging behavior the giant manta ray tends to be more vulnerable to foul hooking as opposed to being hooked in the mouth (Mas et al. 2015).

As with other marine species described in this section, even if the hook is external and removed, a captured giant manta ray is still at risk of post-release tissue and physiological trauma (Miller and Klimovich 2017). However, due to the large size of the animal, they are seldom boarded, so instead of removing the hook, fishermen tend to cut the branch line. This scenario is discussed further in the trailing gear section. If the giant manta ray does ingest the hook, the process of movement, either by the manta ray's attempt to get free of the hook or by being hauled in by the vessel, can traumatize the internal organs or pull the organs from their connective tissue. Once the hook is set and pierces an organ, infection may ensue, which may result in the death of the animal.

Cetaceans

The MHI IFKW population has a high rate of dorsal fin disfigurements and mouthline injuries consistent with injuries from unidentified fishing lines (Baird and Gorgone 2005; Baird et al. 2014, 2017; Beach 2015; Baird 2019). We note, it is not clear what fisheries are responsible for these injuries and the frequency of incidental take to MHI IFKW from the commercial fisheries, including the DSLL, is considered rare (NMFS 2021c). We discuss the information we have on both recreational and commercial fisheries to illuminate the potential effects to the species as they often use the same type of gear.

Recreational troll lines in Hawaii are typically 400-lb monofilament, which would likely be cut free by the angler before significant injuries could occur. Additionally, we note that the State of Hawaii previously did not require reporting for MHI IFKW interactions with troll fisheries to better inform these data. However, recent collaboration between the State, NMFS, and Cascadia Research, resulted in the inclusion of sightings and interaction data for protected sharks and marine mammals in the monthly catch reports for commercial marine license holders in Hawaii.

In addition to numerous self-reports (Boggs et al. 2015) and videos (e.g., Jouppi 2015) of false killer whale depredation in troll and handline fisheries, examination of a stranded MHI IFKW in October 2013 revealed that an individual had five fishing hooks and fishing line in its stomach (NOAA Fisheries PIR Marine Mammal Response Network, as cited in Carretta et al. 2018). Four of the hooks within the whale's stomach were not consistent with those currently allowed for use within the commercial longline fisheries and could have come from a variety of nearshore fisheries (Baird et al. 2014; Carretta et al. 2018). Although the fishing gear is not believed to have caused the death of the whale, the finding confirms that MHI IFKWs are consuming previously hooked fish or are interacting with hook and line fisheries. Furthermore, this highlights that while swallowing hooks and line was not established as the cause of death for these animals, the associated debris may have impeded their ability to forage and/or compromised their health (Kristi West, Hawaii Institute of Marine Biology, pers. comm. 2016 as cited in NMFS 2021c).

We are aware of one additional interaction where a MHI IFKW depredated a recreational troll fisherman's catch (a kawakawa; mackerel tuna) in 200 ft. of water just north of Kalaupapa, Molokai on 15 March 2021 (J. Peschon pers. comm. to J. Rudolph, March 17, 2021). The whale held the fish deep and was almost brought boat side (likely on its own accord as it was not hooked or entangled) before tearing off the body of the fish. The animal subsequently popped up next to the vessel consuming the fish after pulling it off.

As with the other species considered herein, pelagic longline gear interactions with sperm whales and MHI IFKWs may be affected by both soak time and the number of hooks fished, independent of overall fishing effort. The longer the soak time, the greater the chances these two species may encounter the gear, either in attempts at depredation of fish caught on the gear or in simply passing across the gear during normal movements. Gear left in the water longer likely also has a greater chance of having more fish captured on the lines, and may be a source of attraction for sperm whales and MHI IFKWs attempting to depredate captured fish, although Bayeless et al. (2017) found that 57% of depredation events in the Hawaii longline fishery occurred during the haul.

4.2.3 Trailing Gear (Line)

Sea Turtles

Trailing gear is any line left on a turtle after its release. Turtles are likely to swallow line trailing from an ingested hook, which may occlude their gastrointestinal tract, preventing or hampering the turtle when it feeds. As a result, trailing line can eventually kill a turtle shortly after the turtle is released or it may take a while for the turtle to die.

Trailing line can also become snagged on a floating or fixed object, further entangling sea turtles or the drag from the float can cause the line to constrict around a turtle's appendages until the line cuts through the appendage. With the loss of a flipper, a turtle's mobility is reduced, as is its ability to feed, evade predators, and reproduce. Crews that have hooked a turtle are directed to remove the hook if it is external. However, if the hooking is internal and would cause more damage to remove, crew are instructed to cut the line as close to the hook as possible to minimize the amount of trailing gear. Likewise, when larger turtles such as the leatherback, cannot be boarded, crew are instructed to cut the line as close to the hook as possible. Occasionally, the branch line breaks during a turtle interaction and a majority of the line may remain attached to the animal.

Sharks

Excessive trailing gear could directly or indirectly interfere with a shark’s mobility, causing impairment in feeding, breeding, or migration. Further, trailing line can also become snagged on a floating or fixed object, further entangling the shark or the drag from the float can cause the line to constrict around the body of the shark or its fins. Members of the Western and Central Pacific Fisheries Commission are required to regulate their vessels consistent with the conservation and management measures (CMM) for the oceanic whitetip shark. Pursuant to CMM 2011-04 (provisionally updated to CMM 2022-04), NMFS has implemented regulations (50 CFR 300.226 and 50 CFR 665.811) requiring vessels to release any oceanic whitetip shark that is caught as soon as possible after the shark is brought alongside the vessel, and to do so in a manner that results in as little harm to the shark as possible. In accordance with this measure, the amount of trailing gear shall be minimal as to cause as little harm as possible. Currently, there are no CMM's for Indo-west Pacific scalloped hammerhead sharks.

Giant Manta Ray

Given their size, giant manta rays are seldom boarded, and similar to leatherback sea turtles, fishermen are instructed to cut the line as close to the hook as possible after the observer views the animal on trips that are observed. Occasionally, the branch line breaks during an interaction and the majority of the line may remain attached to the animal. If entangled in trailing line, the giant manta ray is at risk of severing of the cephalic and pectoral fin, which are considered severe injuries that can lead to a reduction in feeding efficiency and even death. Trailing line can become snagged on a floating or fixed object, further entangling the giant manta ray or the drag from the float can cause the line to constrict around a manta's cephalic fin until the line cuts through the appendage.

Cetaceans

Similar to the other species, trailing line has the potential to be ingested by cetaceans, which may prevent or hamper their ability to forage or impede their health. As a result, trailing line can eventually kill a whale shortly after it is released, or it may take a while for the animal to die. Alternatively, trailing line can wrap around an animal, constricting its movement, cause it to become snagged or accumulate additional debris. Crew are directed to remove external hooks when possible. However, if the hooking is internal and would cause more damage to remove, crew are instructed to cut the line as close to the hook as possible to minimize the amount of trailing gear. Often, the branch line breaks during the interaction and a majority of the branch line may remain attached to the animal. Expected mortality and serious injury determinations under the MMPA reflect the hook and amount of trailing line which remains on the animal when it is released.

4.3 Species-Specific Analyses

The narratives that follow present our exposure and response analyses for the endangered and threatened species that we previously concluded are likely to be adversely affected by the fishery. For our exposure analyses we estimate the number, age (or life stage), and gender of the individuals that are likely to be exposed to stressors associated with the fishery if the data are available in sufficient detail.

McCracken (2019a) projected the probable number of the different species that are likely to be incidentally captured in the DSLL fishery in the future based on the number and rate of individuals captured in the fishery since 2002. Unlike the SSLL fishery, which has 100% observer coverage, the DSLL fishery has about 20% observer coverage, so McCracken (2019a) had to adjust the number of the different species reported to have been incidentally captured in a particular year to account for the percent observer coverage in each particular year. As noted in section 1.4.4, we calculated maximum 5-yr running averages and running sums based on the annual point estimates of McCracken (2019a), and McCracken and Cooper (2020a, 2020b, 2021a, 2021b, 2022a, 2022b). The following narratives include tables that present estimates of the number of the different species that are likely to be incidentally captured in the fishery based on these metrics.

For our response analyses, we try to assess how animals respond after being incidentally captured in DSLL gear. For the most part these analyses are informed by published observer reports of the animals' condition when they are brought aboard or alongside a fishing vessel. Although animals that are released alive can still die from wounds associated with their capture (i.e. post release mortality) or they can change their foraging patterns, migratory behavior, reproductive behavior because of being captured (Skomal and Mandelman 2012; Lewison et al. 2014; Wilson et al. 2014), there is almost no information on how the experience of being captured in DSLL fishing gear affects the long term fitness of individuals that have been released alive. As a result, our response analyses focus on the condition of individuals when they are brought to a fishing vessel, although we make every effort to consider longer-term effects by reviewing the available literature for metrics related to post release mortality and expected secondary and tertiary responses by animals considered herein.

We acknowledge that a certain proportion of individuals that interact with the DSLL will escape, either by biting or breaking the line or otherwise freeing themselves, before being brought to the vessel and are therefore unobservable (for example, Moyes et al. 2006; Murray 2011; and Warden and Murray 2011; Gilman et al. 2013).We have sufficient information to account for unobserved interactions by oceanic whitetip sharks. However, we do not currently have data to assess the frequency of these occurrences for other species. We are reasonably certain that post-interaction survival for unobserved individuals is high because they were likely in good condition at 'release' and do not endure the stress of the haul and being brought alongside the vessel (i.e. see Hutchinson et al. 2021 for sharks). We believe the proportion of unobserved interactions is very small and the short-term (survival) and long-term (fitness) effects are minimal. For this biological opinion, we have done everything possible to account for unobserved interactions and include their effects in our analyses (e.g. rounding up to the nearest whole number of mortalities, assuming all serious injuries result in mortality, identifying long-term effects such as alterations in blood chemistry or stress responses, etc.).

As noted in Section 3 – Environmental Baseline, there has been an overall increasing trend in hooks per year and hooks per set for the Hawaii DSLL fishery from 2004 to 2022, although hooks per year peaked in 2019 and have been stable or declining in the three years since then (Figure 33). While there is an upper limit on the number of vessels that can fish at any time given the limit on the number of permits, there are no limits on annual trips, sets or hooks per vessel and the number of hooks per year is expected to continue to increase similar to historic rates in the near term (Figure 35). There are, however, practical limits to the amount of effort the fishery can expend given the cap on permits and vessel size.

There are numerous environmental variables that drive species distribution and probability of being captured, and fishery effort is only one factor that is not likely to explain much of the annual variability in observed incidental captures. We checked this for each species using linear regression where the independent variable was annual observed hooks and the dependent variable was observed incidental captures. We found statistically significant relationships between observed hooks and observed incidental captures for green sea turtles and olive ridley sea turtles. For the rest of the species, R^2 was less than 0.07 indicating that less than 7% of the variability in observed incidental captures per year is explained by the number of hooks. We discuss the results for green and olive ridley sea turtles in the relevant sections below. We acknowledge that with the limited data available (19 years) and high year-to-year variability in observed incidental captures, there may be a more significant relationship between hooks and captures than we are currently able to detect for most species and we assert that this will need to be carefully monitored if the fishery effort continues to increase into the future.

Interpretation of the intervals presented in these analyses. The narratives that follow this introduction present and discuss intervals: the exposure estimates present mean values with 95th percentiles (all from McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2021b, 2022a, 2022b) and maximum 5-yr running averages and sums, while response and risk estimates present mean values with upper and lower 95% confidence intervals. It is important to note that all values in a confidence interval have the potential to occur, although those at the edges have lower likelihoods than those towards the center of the interval. The point estimate (in our case, the mean value) is the value that best describes anticipated effects over a longer term.

For example, Table 25 presents annual estimates of the number of oceanic whitetip sharks that are likely to be incidentally captured in the DSLL fishery (annual mean = 1,708; 95th percentile = 3,185). The estimates in Table 25 should be interpreted as “the data suggest that the number of oceanic whitetip sharks incidentally captured in any given year should average about 1,708 and rarely will be higher than 3,185; over any 5-year period we expect the average to be no more than 2,117.8 oceanic whitetip sharks with up to 10,589 over 5 years.” The 5-year maximum running average for oceanic whitetip sharks, 2,117.8, occurred from 2018-2022 when the mean was exceeded in four out of those five years. If our analysis only focused on the mean, 1,708 sharks, we would not have sufficiently analyzed the impact of the fishery on oceanic whitetip sharks over that 5-year timeframe. For this reason our analysis focuses on the maximum 5-year running average rather than the mean.

In our risk analysis, if data are available, we use the exponential growth equation to project population sizes 40 years into the future for species that are declining (20 years for West Pacific leatherbacks) as this is our analytical timeframe (see Section 1.3.1). The exponential population growth equation is expressed as:

$$N_t = N_0*\exp(rt)$$

where N_0 is the starting population size, N_t is the population size after time duration t, and r is the population growth rate. This equation is commonly used to describe small and depleted populations that are not subject to density-dependent factors (see discussion in Martin et al. 2020a). This equation can be rewritten as:

$$N_t/N_0 = \exp(rt) \text{ or } \ln(N_t/N_0)= rt,$$

which can be solved for population growth rate as:

$$r = \ln(N_t/N_0)/t.$$

We use this form of the exponential growth equation to assess population growth rates with and without the fishery. We assume historic population growth rates include the impact of the fishery. To estimate the population growth rates without the fishery, we add estimated fishery mortalities back to the population each year and assess the difference in population growth rates with and without the fishery mortalities in the near term (one year out) and over the long term (20 or 40 yrs out).

For species or DPS' with declining population trends, we assess the statistical significance of the differences in population growth rates using a power analysis (Cohen 1988; Appendix C. A power analysis is used to determine the sample size needed to detect a statistically significant difference in means based on the uncertainty around the mean (i.e. 95% CI or standard deviation). Small differences in means and large confidence intervals generally require large sample sizes to detect these differences. In our case, sample size would be the number of years of data required to detect the differences in the population growth rates with and without the fishery. In our power analysis we assess the number of years required to be 80% certain the population growth rates are statistically significantly different with probability less than 5%. We assume equal variances for the population growth rates with and without the fishery. We also assess the power to detect a difference over the 40 year analysis timeframe (20 years for West Pacific leatherbacks). In other words, how certain can we be that we have statistically significant differences in population growth rates with fishery mortalities and after removing fishery mortalities for 40 years (20 years for West Pacific leatherbacks). Details are provided on these analyses in Appendix C.

The upper limits of 95% confidence intervals, or the 95th percentile, are important because they are compatible with the data (although less so than the mean), represent greater risks for endangered and threatened species; although these upper limits will always be higher than the average values, they will still be exceeded in some years and these values are within the range of estimates considered reasonably likely to occur. Our assessments do not focus on the lower limits of 95% confidence intervals because they represent lesser risk to endangered and threatened species; however, values between the mean and the lower limit of 95% intervals are also compatible with the data and are not ignored in our analyses. However, for risk analyses over longer time intervals (for example, 40 years) we rely on mean estimates and 5-year maximum running averages because long time series are most likely to include values both above and below the mean.

4.3.1 Giant Manta Ray

Exposure Analysis

Giant manta ray have been incidentally captured in the DSLL fishery in 16 of the 19 years between 2004 and 2022. During this 19-year time interval fishery observers reported that 47 giant manta ray were incidentally captured in the DSLL fishery. An additional 105 unidentified *Mobulidae* and 29 unidentified rays were incidentally captured over that timeframe, some of which were likely to have been giant manta rays. To estimate how many rays from the *Mobulidae* spp. were likely to be giant manta rays, we calculated the proportions of giant manta ray and *Mobula* (devil rays) observed incidentally captured from 2004 to 2022 and used the Wilson Score method without continuity correction (Newcombe 1998) to estimate the 95% confidence interval around the proportion of giant manta ray (Table 14). To estimate how many rays from the unidentified ray category were likely to be giant manta rays, we calculated the proportions of giant manta ray, *Mobula* (devil rays), and pelagic stingray observed incidental captures from 2017 to 2021 (date range for which we have pelagic stingray capture data) and used the Wilson Score Method to estimate the 95% confidence interval around the proportion of giant manta ray (Table 15). We multiplied that upper 95% CI by the number of *Mobulidae* spp. and unidentified rays that McCracken (2019a) and McCracken and Cooper (2020a, 2020b, 2021b, 2022b) estimated were incidentally captured each year from 2004 to 2021 to estimate the proportions of each unidentified category that were likely to have been giant manta ray. For 2022 we used the expansion factor and the 20.22% observer coverage to estimate the number of *Mobulidae* spp. and unidentified rays. In total, from 2004-2022, the DSLL fishery had 61 observed interactions with giant manta rays (47 estimated giant manta rays plus 14 from the *Mobulidae* spp. and 0 from the unidentified ray categories).

Table 14. The proportion of estimated incidental captures identified as giant manta ray and *Mobula* (devil rays) from 2004 to 2022 and the 95% confidence interval for the proportion.

Species	**Original data**			**Proportions**		**Confidence Intervals for Proportions**	
	Number of Individuals Assigned to a Species	**Number of Individuals Not Assigned to a Species**	**Sample Size**	**p (Species Proportions)**	**1-p (Not Species)**	**Wilson LCI (Species)**	**Wilson UCI (Species)**
Giant Manta Ray	47	350	397	0.12	0.88	0.09	0.15
Mobula (Devil Rays)	350	47	397	0.88	0.12	0.85	0.91

Table 15. The proportion of estimated incidental captures identified as giant manta ray, *Mobula* (devil rays), and pelagic stingrays from 2004 to 2022 and the 95% confidence interval for the proportion.

Species	**Original data**			**Proportions**		**Confidence Intervals for Proportions**	
	Number of Individuals Assigned to a Species	**Number of Individuals Not Assigned to a Species**	**Sample Size**	**p (Species Proportions)**	**1-p (Not Species)**	**Wilson LCI (Species)**	**Wilson UCI (Species)**
Giant Manta Ray	47	23,775	23,822	0.00	1.00	0.00	0.00
Mobula (Devil Rays)	350	23,472	23,822	0.01	0.99	0.01	0.02
Pelagic Stingrays	23,435	397	23,822	0.98	0.02	0.98	0.98

We used the same ratios of confirmed giant manta rays, mobula (devil ray), and pelagic sting rays to estimate the number of giant manta rays incidentally captured based on the observed incidental captures. An estimated 239 giant manta ray and 576 unidentified *Mobulidae* spp. were incidentally captured from 2004 to 2022 (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021b, 2022b; and using the expansion factor and observer coverage of 20.22% for 2022). Using the upper 95th CI from Table 14, we estimate that 15% of the latter may have been giant manta ray, or 89 individuals. Therefore about 328 (=239+89) giant manta ray were likely to have been incidentally captured in the fishery over the 19-year period from 2004 to 2022 (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021b, 2022b).

Spatially, giant manta rays were incidentally captured across the range of the DSLL fishery. To visualize where incidental captures occur, we used ArcGIS/ArcMap kernel density tool to calculate a density, which is based on point features using interaction numbers (for captures) and set numbers (for effort). This allowed us to create a heat map, which illustrates the *relative* number of interactions or effort in an area corresponding to those particular values. According to ESRI, kernel density is based on the quartic kernel function described by Silverman (1986; ESRI 2018). By adjusting these ranges, the resulting maps appear smoother with edges that are more fluid. As the *relative* density increases, the color shifts from a cooler blue to a more dynamic warmer tone, also known as a color ramp. Therefore, blue areas have lower interaction or effort rates relative to areas with orange or reds.

The key was adjusted to illustrate low versus high-density values and is consistent with all other spatial maps presented. For giant manta ray, there were not enough observed incidental captures to conduct this analysis, hence we included the unidentified *Manta*/*Mobula* and unidentified ray observed incidental captures, increasing the sample size to 312 for Figure 37 and Figure 38. For observed giant manta ray incidental captures, 40% of them occurred in or near the Palmyra EEZ. Of the remaining 60%, 85% occurred south of the MHI. We provide a comparison of *Manta*/*Mobula* interactions versus the aggregate fishing effort to highlight that the incidental captures in or near the Palmyra EEZ are not simply a reflection of where effort occurs (Figure 38). While 2021 data is not shown in the heatmap, the 2 incidental interactions with giant manta rays occurred within the geographical range depicted in the heatmap and would not be expected to alter the density.

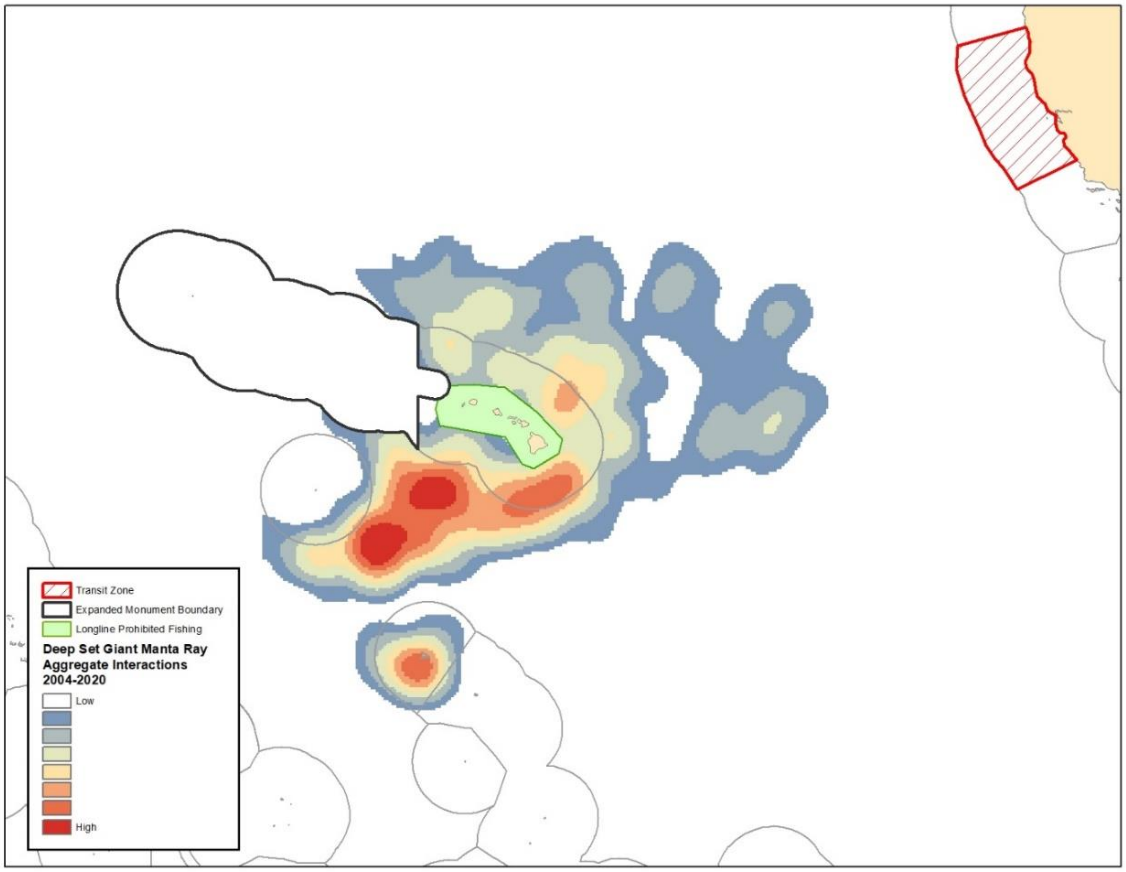


Figure 37. Geographical representation of observed aggregate giant manta ray and unidentified Manta/Mobula and ray interactions with the DSLL fishery between 2004 and 2020 (*n* = 312).

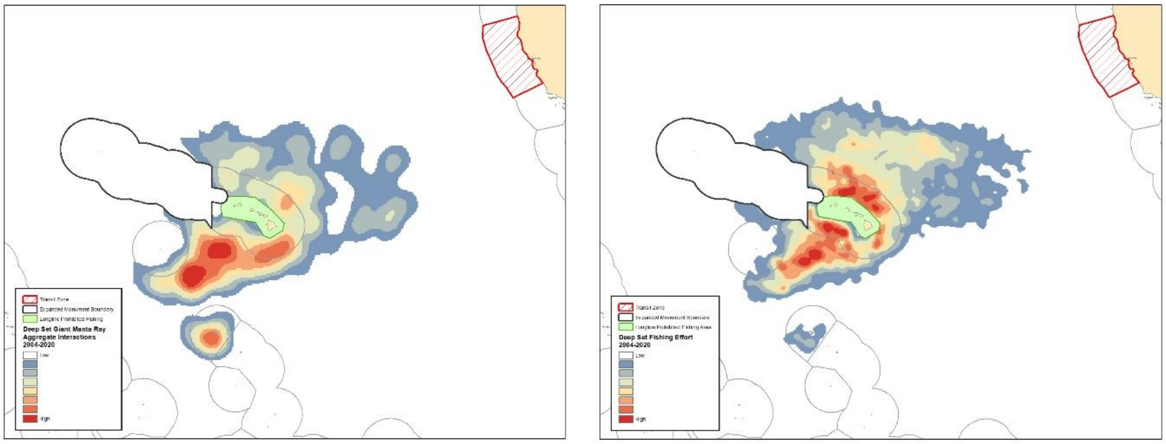


Figure 38. Heatmap comparison of observed giant manta ray and unidentified *Manta/Mobula* and interactions (left; *n* = 312) versus aggregated observed fishing effort during this same time frame (right; *n*= 65,245 from 2004 to 2020).

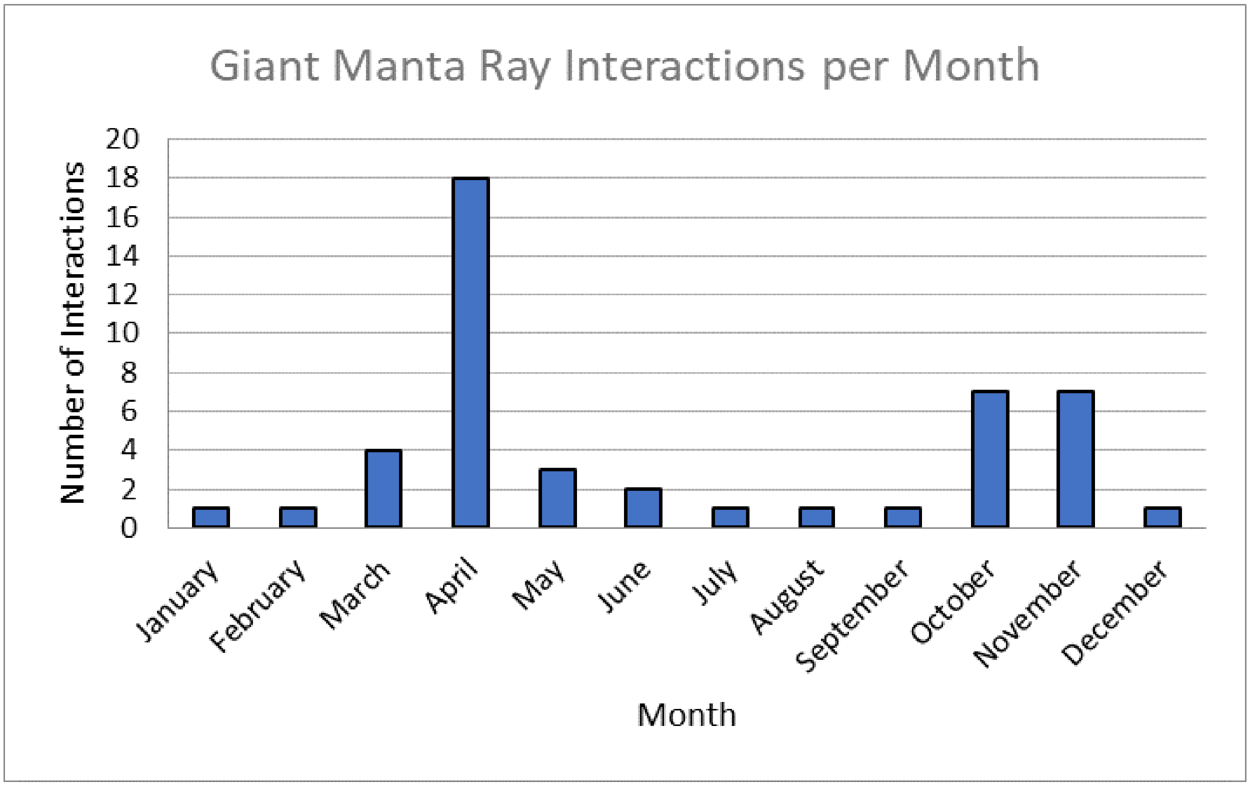


Figure 39. Numbers of observed giant manta ray incidentally captured per month in the DSLL fishery from 2004 to 2022 (*n* = 47).

Assessing aggregate annual observed interactions, 40% occurred in the month of April (Figure 39) and like oceanic whitetips, all interactions in this month occur south of the MHI. All interactions that occurred in Palmyra were also in this month. Observed and estimated giant manta ray interactions peaked in 2010 with 17 observed and 95 estimated (McCracken 2019a) incidental captures (Figure 40). The incidental captures in 2010 were more than 4 times higher than observed or estimated incidental captures in any other year.

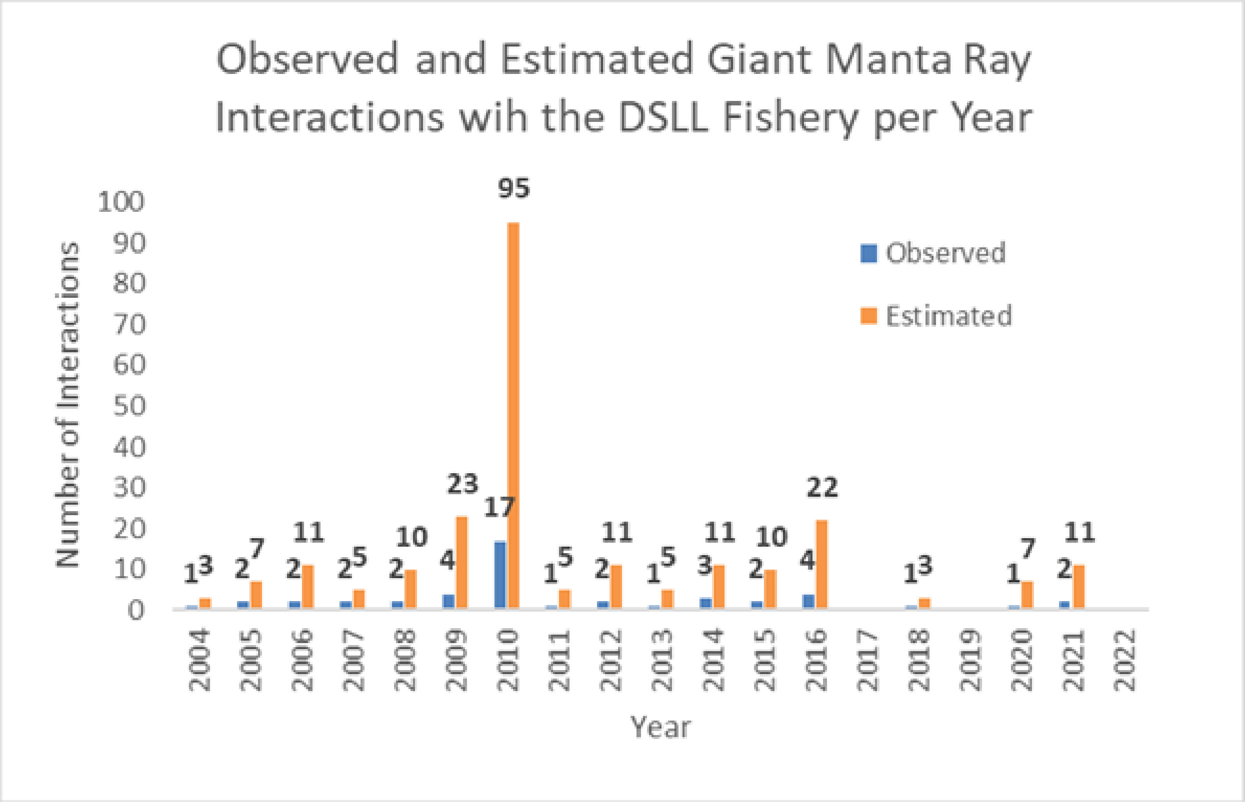


Figure 40. Giant manta ray interactions per year in the DSLL fishery between 2004 and 2022 for observed interactions (blue bars) and estimated interactions (orange bars; from McCracken 2019a and McCracken and Cooper 2020a, 2020b, 2021b, 2022b). There were no observed giant manta ray captured in 2022 and therefore we also assume zero for estimated interactions.

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interaction levels (McCracken 2019a) using Bayesian inference techniques as described in the Approach to the Assessment section of this biological opinion. The predictions were based on the recorded bycatch of giant manta ray, unidentified rays, and the *Mobulidae* spp. classification in the DSLL fishery (Table 16). We also report the maximum 5-yr running average and the maximum cumulative number of anticipated captures over 5 years that have been prorated for unidentified categories (Table 16).

Table 16. Anticipated annual and 95th percentile (from McCracken 2019a; 2002-2017), and maximum 5-year running average and maximum cumulative 5-year running sum anticipated exposures of giant manta ray in the DSLL fishery using estimated annual captures from McCracken 2019a and McCracken and Cooper 2020a, 2020b, 2021b, 2022b). Values in parentheses are the numbers anticipated inclusive of unidentified *Manta/Mobula.*

	DSLL Encounters	
Rate	**Mean**	**95th Percentile**
Annual	20 (23)	60 (68)
Maximum 5-Yr Running Average	28.8 (39.7)	N/A
5-Yr Running Sum	144 (198)	N/A

Response Analysis

One of the 47 giant manta rays observed captured in the DSLL fishery was dead at capture resulting in a mean at vessel mortality rate of 2.10% (95% CI: 0.40%, 11.1%). Such low at-vessel mortality rates for giant manta ray are consistent with other longline fisheries. Mas et al. (2015) looked at *Mobulidae* bycatch in the Uruguayan and Japanese longline fisheries operating in the Southwest Pacific. They found that 89.6% of all captured mobulids were released alive, 5.4% were released dead and 5% were lost during the haul and their fate was uncertain. Similarly, for the United States longline fisheries operating off the Southeast United States coast, Beerkircher et al. (2008) found that 0% of captured *Dasyatidae* and *Mobulidae* rays were dead at gear retrieval, although 3% were dead at release. Coelho et al. (2011) also found 0% mortality at haulback for *Mobulidae* rays captured in the Portuguese longline targeting swordfish in the Indian Ocean. From 2004 to 2018, the Hawaii SSLL interacted with 21 giant manta rays, of which 4 were dead at-vessel for a 19.1% mortality rate (NMFS 2019a).

As described above, there is available information regarding at-vessel mortality rates for giant manta rays captured in longline fisheries, however there is very limited information on post-release mortality. An extensive review of the literature for post-release survivorship for *Mobulidae* spp. has determined that there are no studies specific to longline fisheries that assess the effect of remaining gear on manta and mobulid species or the effect of stress and injuries that may be sustained during capture (Mas et al. 2015; Griffiths and Lezama-Ochoa 2021). When giant manta rays interact with longline gear, they are particularly prone to being entangled in fishing gear because of their body configuration and behavior. Giant manta rays tend to be more vulnerable to entanglement and foul hooking as opposed to being hooked in the mouth (Sales et al. 2010; Domingo et al. 2012). If entangled in a monofilament branch line or polypropylene float line, giant manta rays are at risk of severing their cephalic and pectoral fin. These are severe injuries that can reduce their feeding efficiency and result in their death. In general, trailing gear attached to animals after release from longline fisheries is likely to reduce survival by restricting swimming efficiency which may impact foraging efficiency and vulnerability to predation and by increasing vulnerability to disease and infection (Scott et al. 2022).

There is very little information on the physical and physiological effect of entanglement or trailing gear on the giant manta ray. Surveys of the reef manta from 2005 to 2009 at an aggregation site of Maui, Hawaii, revealed that 10% of the population had an amputated or non-functional cephalic fin (Deakos et al. 2011). Almost all injuries were attributed to entanglement in fishing line (most likely from recreational or nearshore fisheries) since the straight edge cut of all amputated cephalic fins resemble the severing effects of monofilament (Deakos et al. 2011). In fact, eight individuals had physical evidence of entanglement with fishing line; two individuals had hooks in the cephalic fin, two had monofilament wrapped around the cephalic fin and another two had scars from previous line entanglements, and two individuals had line that had begun to cut part way through the cephalic fin (Deakos et al. 2011). Deakos et al. (2011) observed that individuals in this population with an amputated cephalic fin appeared healthy; however, considering the function of the cephalic fin to guide food into the manta's mouth, feeding efficiency is most likely reduced, and the absence of this fin may negatively affect size, growth rate and reproductive success.

As with other marine species described in this section, even if a hook is external and removed, a captured giant manta ray is still at risk of post-release tissue and physiological trauma. However, due to their large size, giant manta rays are seldom boarded, so instead of removing the hook, fishermen tend to cut the branch line. For the 45 giant manta rays observed interacting with the DSLL fishery, one was reported as dead at-vessel, 15 were reported as entangled and there is no information regarding injury or hooking location for the remaining 29 rays. Therefore, these data do not allow us to estimate post-interaction mortality rates for giant manta ray incidentally captured in the fishery. The data available also do not allow us to reach any conclusions about the effects of capture on the longevity or reproductive success of giant manta ray that survive being captured in DSLL gear. We anticipate that at least some of the entangled individuals would likely have high post-interaction mortality rates based on the observer's descriptions, for example:

> "giant manta ray was tangled in long section of tangled mainline including many branchlines, therefore no specific hook number could be assigned", and "tangled with unknown number of lines starting at line 18."

Giant manta rays are obligate ram ventilators and must continually move to maintain oxygen requirements. Therefore, entanglement could limit their mobility and result in mortality. Even those animals not noted as entangled were likely released with hooks and an unknown amount of line attached which could result in tissue and/or physiological trauma (Scott et al. 2022). Because post-interaction mortalities are likely to occur based on our understanding of giant manta ray physiology, we need to incorporate these mortalities into our assessment and assign a post-interaction mortality to giant manta rays interacting with the fishery. In a recent assessment of spinetail devil ray population impacts from purse seine and longline fishery bycatch, Griffiths and Lezama-Ochoa (2021) assumed a baseline post-interaction mortality rate of 100% for both fisheries given the lack of information on this metric. They explored the impact of reductions in this mortality rate over 46 scenarios; the mean post-interaction mortality rate across these scenarios was 67.4%.

We do not have sufficient information to calculate a post-interaction mortality rate for giant manta ray, or any other mobulid species captured in the DSLL. Thus, we looked at the biology of other species for a surrogate. In our assessment of potential post-interaction mortality rates in the DSLL, we consider leatherback sea turtles as an appropriate surrogate species. Similar to the leatherback sea turtle, giant manta ray tend to be foul hooked externally or entangled in the branch line (Sales et al. 2010; Domingo et al. 2012). Since neither of these species are actively depredating the bait on longline gear but are attracted to marine life that collects on buoys and buoy lines at or near the surface, or some combination of these and/or other reasons, they tend to be foul hooked in the flipper and pectoral fin area as opposed to the mouth or in the esophagus. Moreover, both species are large in size; the giant manta ray is considered to be one of the largest fishes in the world whereas the leatherback sea turtle is the largest marine turtle and one of the largest reptiles in the world. Further, both animals propel through the ocean environment using similar movements; the leatherback sea turtle employs its long front flippers while the giant manta ray relies on its large pectoral fins as wings to "fly" through the water. There is a key difference between these species; leatherback sea turtles are air-breathers and must surface to breath, giant manta ray are ram ventilators and must continually move to maintain oxygen requirements. However, entanglements would restrict each species from gaining oxygen regardless of their method of breathing.

Therefore, we assume due to their similarities, and in the absence of species (or genus) specific mortality rates for giant manta ray, the leatherback sea turtle post-interaction mortality rates are the best available proxy. We follow the approach used in the SSLL biological opinion (NMFS 2019a) and use the Ryder et al. (2006) injury criteria for leatherback sea turtles as a proxy for determining post-hooking survival of giant manta rays for one method of assigning post-interaction mortality. As a second method, we assume 100% mortality for the 15 entangled individuals and 0% mortality for all other individuals. Assuming a 0% mortality rate for untangled manta ray and 100 % for entangled allow us to consider a range of mortality rates for this release condition as described below.

As we have no specific information on hook location or release condition for the 31 giant manta ray released alive but not recorded as entangled, we assumed post-interaction mortality rates of 0% (second method described above), 10% (consistent with a leatherback sea turtle released with all gear removed; Ryder et al. 2006), or 30% (consistent with a leatherback sea turtle released with an external hook and line greater than or equal to half of the carapace length; Ryder et al. 2006). For the entangled giant manta ray, we assumed post-interaction mortality rates of 60%
(consistent with a leatherback sea turtle released entangled; Ryder et al. 2006), 65% (consistent with a leatherback sea turtle released with hook and entangled; Ryder et al. 2006), and 100% (second method described above). Table 17 details the resulting post-interaction and total (inclusive of at-vessel mortality) mortality rates, the 95% CI for total mortality rates using these three methods ranged from 16.1% to 60.9%. To be precautionary, we used the highest estimate of mean total mortality (42.7%) in the risk analysis below. At this mortality rate, if 39.7 giant manta ray are captured in the fishery in any given year, we would expect an average of 16.9 of them to die as a result of being captured; using the 95th percentile value, if 68 giant manta ray are captured, we would expect and average of 29 of them to die as a result of their capture (Table 18).

As noted in the Status of Listed Resources section, the U.S. has issued a proposed rule that will implement CMM 2019-05 mobulid safe handling practice for U.S. fisheries in the WCPO including the DSLL. Specifically this will encourage that line cutters be used to remove line attached to mobulids. Therefore we consider it reasonable to assume a lower mortality rate for giant manta ray than that proposed by Griffiths and Lezama-Ochoa (2021) for spinetail devil rays.

Table 17. At-vessel, post-interaction, and total mortality rate estimates for giant manta ray interacting with the DSLL fishery from 2004 to 2021. Forty-four giant manta rays were observed interacting with the fishery, 1 was dead at-vessel, 15 were entangled and 29 were released alive but otherwise did not have any information on injuries or release condition.

	Method 1; assume 0.10 for not entangled and 0.60 for entangled	**Method 1; assume 0.30 for not entangled and 0.65 for entangled**	**Method 2; assume 0.0 for not entangled and 1.0 for entangled**
Post-Interaction Mortality Rate	0.263	0.414	0.326
L95%	0.158	0.283	0.209
U95%	0.405	0.557	0.470
At-vessel mortality rate	0.021	0.021	0.021
L95%	0.004	0.004	0.004
U95%	0.111	0.118	0.118
Total Mean Mortality Rate	0.279	0.427	0.340
L95%	0.161	0.286	0.212
U95%	0.471	0.609	0.533

Table 18. Based on the anticipated number of captures from Table 16 and the estimated mortality rate of 42.7%, these are the number of giant manta ray that are likely to die as a result of being captured in the DSLL each year. Values in parentheses are the numbers anticipated inclusive of unidentified *Manta/Mobula.*

	DSLL Encounters	
Rate	**Mean**	**95th Percentile**
Annual	9 (10)	26 (29)
Maximum 5-Yr Running Average	12.3 (16.9)	N/A
5-Yr Running Sum	61 (858)	N/A

Risk Analysis

On average, we expect that the Hawaii DSLL fishery will interact with 23 (95th percentile= 68) giant manta ray each year and of those 10 (95th percentile = 29) would be expected to die as a result of their interaction, accounting for unidentified *Mobulidae* that are likely giant manta ray. The maximum 5-yr running average is 39.7 giant manta ray and we anticipate that up to 198 giant manta ray may be incidentally captured over 5 years. Of those, 16.9 (5-year running average) and 85 (5-year running sum) would be expected to die. Some years may be higher or lower than the annual average, but the annual number of incidental captures should rarely exceed the 95th percentile of 70 and the cumulative captures over 5-years is not expected to exceed 198 giant manta ray (144 identified to species). Over a 40 year time horizon, if captures remain consistent with historical numbers we would anticipate a cumulative total of 1, 588 incidental captures and we would anticipate that 678 of those rays would die as a result of their interactions.

As discussed in the Status of Listed Resources section for giant manta rays, although listed globally, we do not know this species' underlying population structure. Nevertheless, the species' biogeography allows us to recognize that giant manta ray in the Atlantic and Indo-Pacific Oceans are independent of one another. Further, giant manta ray subpopulations appear to be regionally distinct (Lewis et al. 2015; Stewart et al. 2016a; Moazzam 2018; Beale et al. 2019) and may have distinct home ranges (Stewart et al. 2016a). Here, we use the term "subpopulation" distinct from DPS'. If groups of animals are not genetically or demographically interconnected, they are separate populations. When a set of individuals within a population is not spatially disjunct from other individuals, we refer to them as subpopulations (Wells and Richmond 1995). Whereas DPS' are vertebrate populations, or groups of populations, that are discrete from other populations of the species and significant in relation to the entire species (61 FR 4722). Subpopulations may exist within and may be significant in relation to populations and DPS'.

The degree to which subpopulations are connected by migration is unclear but is assumed to be low so, the decline of the small subpopulations may result in regional depletions or extirpations with the reduced possibility of successful recolonization (Marshall et al. 2018). For example, in Indonesia, gill net fisheries in the Lembeh Strait captured 1,424 manta rays (*Mobula* spp.) in a 10-month period (Cochrane 1997 as cited in Beale et al. 2019), resulting in an apparent local extirpation (D. Djalal and A. Doali, pers. comm. cited in Beale et al. 2019).

The number of giant manta ray subpopulations exposed to the DSLL fishery, and the size of those subpopulations is unknown; however, these are important attributes necessary for understanding the effect of fishery-related incidental captures and mortalities on giant manta rays and their risk of extinction. We used information from the distribution of observed giant manta ray incidental captures by the DSLL fishery to estimate the number of giant manta ray subpopulations that may be effected by the fishery, which resulted in a total of 3 subpopulations (Table 19). With the information we have collected from the literature about the limited movements of giant manta ray and their small home ranges (e.g., Graham et al. 2012; Stewart et al. 2016a; Beale et al. 2017) and the giant manta ray data from this fishery, we consider this the best approach to ensure we are evaluating the effect of the proposed action on the species appropriately, including relevant subpopulations.

Several studies have tracked individual giant manta rays and provide information on the spatial extent of giant manta ray subpopulations. Stewart et al. (2016a) studied four subpopulations of giant manta ray using genetics, stable isotopes, and satellite tags. They found that these subpopulations appeared to be discrete with no evidence of movement between them. The home ranges for three of these subpopulations (all of which are outside of the Action Area), defined as the areas where tagged animals were expected to spend 95% of their time encompassed areas of 79,293 km^2 (Raja Ampat, Indonesia), 70,926 km^2 (Revillagigedo Islands, Mexico), and 66,680 km^2 (Bahia de Banderas, Mexico; Figure 41A&B). They suggest that their findings indicate that giant manta rays form discrete subpopulations that exhibit a high degree of residency. Stewart et al. (2016a) state that this does not preclude occasional long-distance migrations, but that these migrations are likely rare and do not generate substantial gene flow or immigration of individuals into these subpopulations. One instance of a long-distance migration has been noted in the literature. Hearn et al. (2014) tracked nine giant manta rays at Isla de la Plata, Ecuador. Eight of the nine tagged giant manta rays remained in an area of 162,500 km^2, while the ninth traveled a straight-line distance of 1,500 km to the Galapagos Islands (Figure 41C).

We used ArcGIS to estimate the longest straight-line distance of the area of occupancy from maps presented in Hearn et al. (2014) and Stewart et al. (2016a). While these subpopulations are all outside of the Action Area, we do not have information on subpopulations within the Action Area and therefore these studies represent the best scientific information available. The resulting values ranged from 457 km in Bahia de Banderas, Mexico to 590 km in Isla de la Plata Ecuador (Figure 41), a circle with a minimum diameter of approximately 600 km or a radius of 300 km would encompass these subpopulations. However, as this is a small sample size of studied subpopulations, we increased the radius to 500 km to have a higher level of confidence that we would encompass all incidental fishery captures that are impacting specific subpopulations.

Between 5% (Stewart et al. 2016a) and 11% (1 out of 9 tagged animals; Hearn et al. 2014) of individuals can be expected to be outside of the core area of occupancy but would be encompassed within a 1,500 km radius. Again, given the low sample size of the studied subpopulations, we assumed that up to 15% of individuals belonging to a specific subpopulation would be between 500 and 1,500 km away from the center point of the aggregation. Therefore, to estimate the number of captures by the DSLL fishery that impact individual subpopulations, we created 500 km radius circles around the center points of the subpopulations identified in Table 19 (Figure 42) and assumed that all captures within that circle impacted that subpopulation (note that some captures are assigned to more than one subpopulation).

We then created 1,500 km radius circles around the center points of the subpopulations and assumed that 15% of all individuals between the 500 km circle and the 1,500 km circle impacted that subpopulation. The sum of these two values represents the observed 2004-2021 incidental captures impacting the individual subpopulations (Table 19). To estimate future incidental captures per subpopulation, we divided the number of captures per subpopulation by the total number of observed captures from 2004 to 2022 to get the proportion of total captures expected per subpopulation. We then multiplied these proportions by the future anticipated annual incidental captures (39.7, 95th percentile: 68) to get the number of annual incidental captures expected per subpopulations (Table 19).

Table 19. Giant manta ray subpopulations likely to be impacted by the DSLL fishery. The number of observed giant manta ray and the number of anticipated future exposures per year impacting individual subpopulations were calculated as described in the text.

Giant Manta Ray Subpopulation	**Center Point Latitude**	**Center Point Longitude**	**Observed Giant Manta Ray Captures**	**Anticipated Annual Exposures** **Maximum 5-yr RA (95th percentile)**
Hawaii	21.013	-157.635	14	12 (20)
Johnston Atoll	13.174	-165.825	13	11 (19)
Palmyra Atoll	6.446	-162.358	24	20 (35)

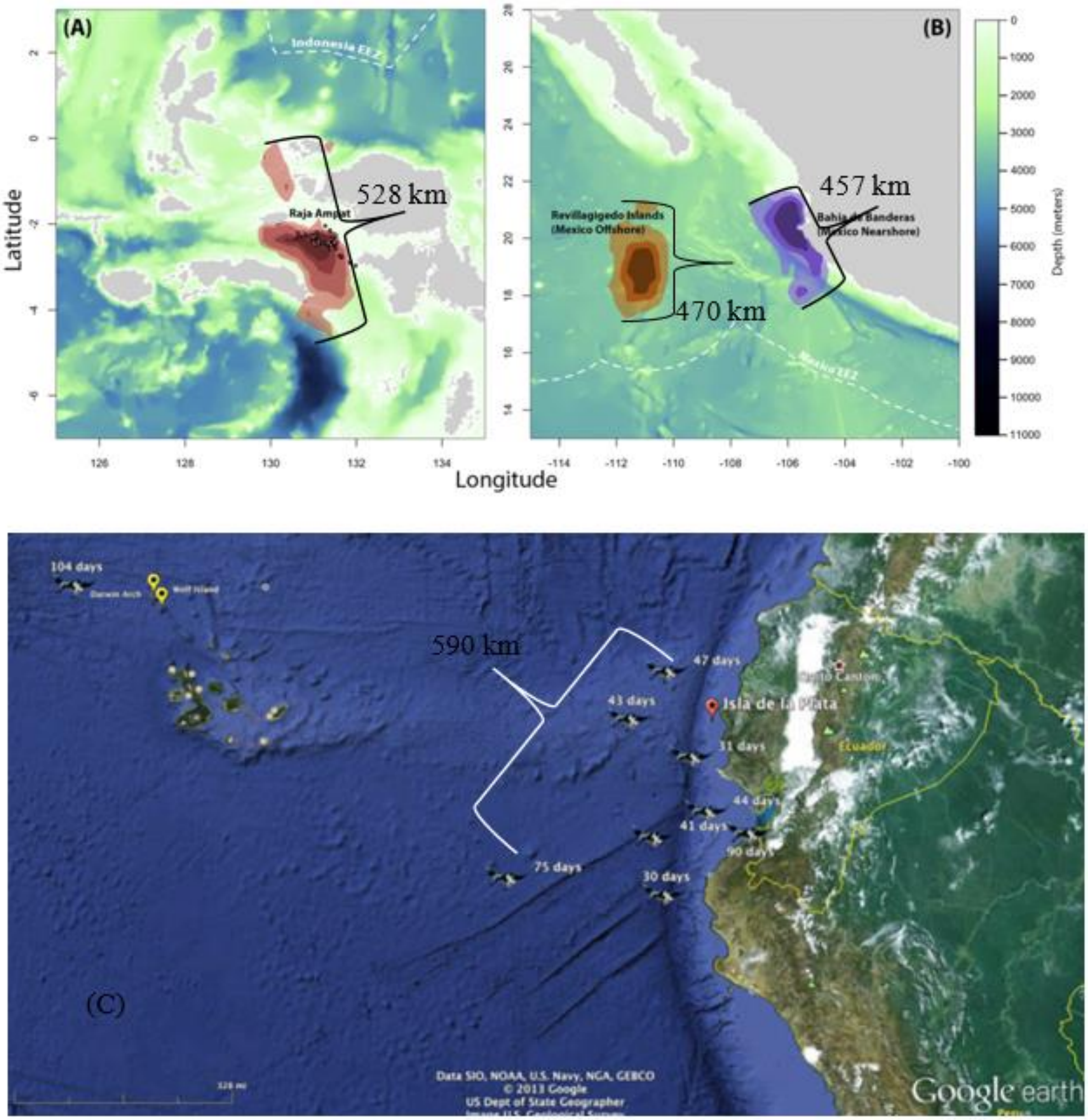


Figure 41. Extent of giant manta ray subpopulations based on tracking tagged individuals. Black brackets (white brackets in C) highlight the longest dimension of the area of occupancy with the length noted in kilometers (km). Locations: A) Raja Ampat, Indonesia (Stewart et al. 2016a); B) Revillagigedo Islands (left) and Bahia de Banderas (right), Mexico (Stewart et al. 2016a); C) Isla de la Plata, Ecuador (Hearn et al. 2014).

As described above, based on the distribution of observed incidental captures in the DSLL and the current understanding of giant manta ray movements and habitat use, we conclude that the action affects three giant manta ray subpopulations. Based on Stewart et al. (2016a), we would anticipate that these are discrete subpopulations with little or no movement between them. As noted above, studies indicate that giant manta rays likely have discrete subpopulation structure with limited movement between subpopulations (Lewis et al. 2015; Stewart et al. 2016a; Moazzam 2018; Beale et al. 2019), and we focus our analysis on the 3 subpopulations we conclude occur in the Action Area.

Scenarios to Understand the Potential Impacts of Fishery Interactions on Giant Manta Ray Population

To overcome the absence of information on the number of manta ray subpopulations that are likely to be exposed to the DSLL fishery and the size of those subpopulations we developed scenarios to assess the probable effects of the DSLL fishery on the giant manta ray. We used the exponential population growth model:

$$N_t = N_{t-1}e^r$$

Where N_t is the population size at time t, N_{t-1} is the population size one year earlier than N_t, and r is the intrinsic population growth rate.

Intrinsic population growth rates for giant manta rays have been estimated at 0.019 (reported mean, Rambahiniarison et al. 2018), 0.042 (Ward-Paige et al. 2013) and 0.116 (reported mean value, Dulvy et al. 2014). The differences in these values are assumptions regarding age at maturity, fecundity, longevity, and instantaneous natural mortality rate, as well as differences in the form of the Euler-Lotka equations used in each study. After a careful examination of the three published studies that present values of r_{max} for giant manta ray (Ward-Paige et al. 2013; Dulvy et al. 2014; Rambahiniarison et al. 2018), NMFS (2021a) concluded that the r_{max} values published by Rambahiniarison et al. (2018) likely represent the best r_{max} value available for this consultation ($r_{max} = 0.019$). NMFS (2021a) concluded that the r_{max} value published by Ward-Paige et al. (2013; $r_{max} = 0.042$) is also reasonable but left us with some concerns over the methodology because the density dependent assumptions used by Ward-Paige et al. (2013) to estimate survival to maturity result in somewhat illogical r_{max} values, whereby the lowest adult mortality rate and highest fecundity would combine to result in a low r_{max} value of 0.012 contrasted with the mean estimate of 0.116 estimated by Dulvy et al. (2014) using the same values. Lastly, NMFS (2021a) determined that the r_{max} estimated by Dulvy et al. (2014) is likely an overestimate because the Euler-Lotka equation that they used is not consistent with other derivations in the literature (i.e., it is missing a term for survival to maturity; Myers et al. 1997; Smith et al. 1998) and may be overestimating $r_{max,}$ which may underestimate fishery impacts to giant manta ray subpopulations[8]. Therefore, for this analysis, we follow the same methodology used in the United States WCPO purse seine biological opinion (NMFS 2021a) and focus on r_{max} values of 0.019 and 0.042.

Our scenarios considered a range of initial population abundances (abundances were either 60, 100, 200, 400, 600, 800, 1,000, 1,500, 1,875 or 2,464 individuals). The range of abundances we

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[8] Additional details on this analysis is provided in a Memo to the Record for NMFS (2021a): M. Snover to A. Garrett, February 2021, Review of studies that present values of rmax for giant manta ray.

consider in our scenarios was designed to capture the range of counts of individuals and population estimates from CITES (2013) and subsequent abundance estimates (Burgess 2017; Beale et al. 2019; Table 3). As noted in the Status of Listed Resources Section 3, we assume the viable giant manta ray subpopulations are likely to be larger in abundance, potentially greater than 1,000 individuals, and we focus our analysis on these population sizes but include the smaller population sizes since they may also represent exploited populations that may be at higher risk of extirpation. Within both scenarios, population abundance was projected over 10 years, 20, and 40 years. Population abundance in each year was estimated using the equation above and subtracting fishery mortalities, assuming constant fishing mortalities regardless of population size. The annual fishery mortalities for each population projection were calculated as the future exposures per year per subpopulation based either on the maximum 5-yr running average (Scenario 1) or the annual upper 95th percentile (Scenario 2) multiplied by the mean estimated post-interaction mortality rate of 0.427 (see the Response Analysis section above for details on this mortality rate). We consider the maximum 5-yr running average as the outcome that is reasonably likely to occur, as this level of interaction has been estimated (based on observed records) to have occurred. Exploration of the upper 95th percentile values represents the approximate most conservative reasonable number of interactions we anticipate will occur in any given year.

In other fisheries, misidentification of giant manta rays can be an issue, in that individuals identified as giant manta rays may be different species. Reef manta rays are the only species we would expect confusion on identification to occur in the DSLL. Given that there is no observer category for reef manta rays, it is possible individuals identified as giant manta rays may be reef manta rays. SFD has a high level of confidence that individual identified giant manta rays are manta rays and not likely to be other mobulids (B. Harman pers. comm. to A. Garrett, 11/5/2021), as is the case with the United States WCPO purse seine fishery where misidentification is likely a much greater problem (NMFS 2021a). For the DSLL, we assume all giant manta rays observed incidentally captured were correctly identified as the data are verified for species identification through the debriefing process, including examination of photographs required to be collected by observers, before data are finalized.

In summary, the scenarios capture our uncertainty in giant manta ray population structure and correct identification of giant manta rays.

> Scenario 1: three discrete populations; 44% post-release mortality rate; maximum 5-yr running average for anticipated future annual interactions; 10, 20, and 40 years (Table 20).
>
> Scenario 2: three discrete populations; 44% post-release mortality rate; 95th percentile of anticipated future annual interactions; 10, 20, and 40 years (Table 20).

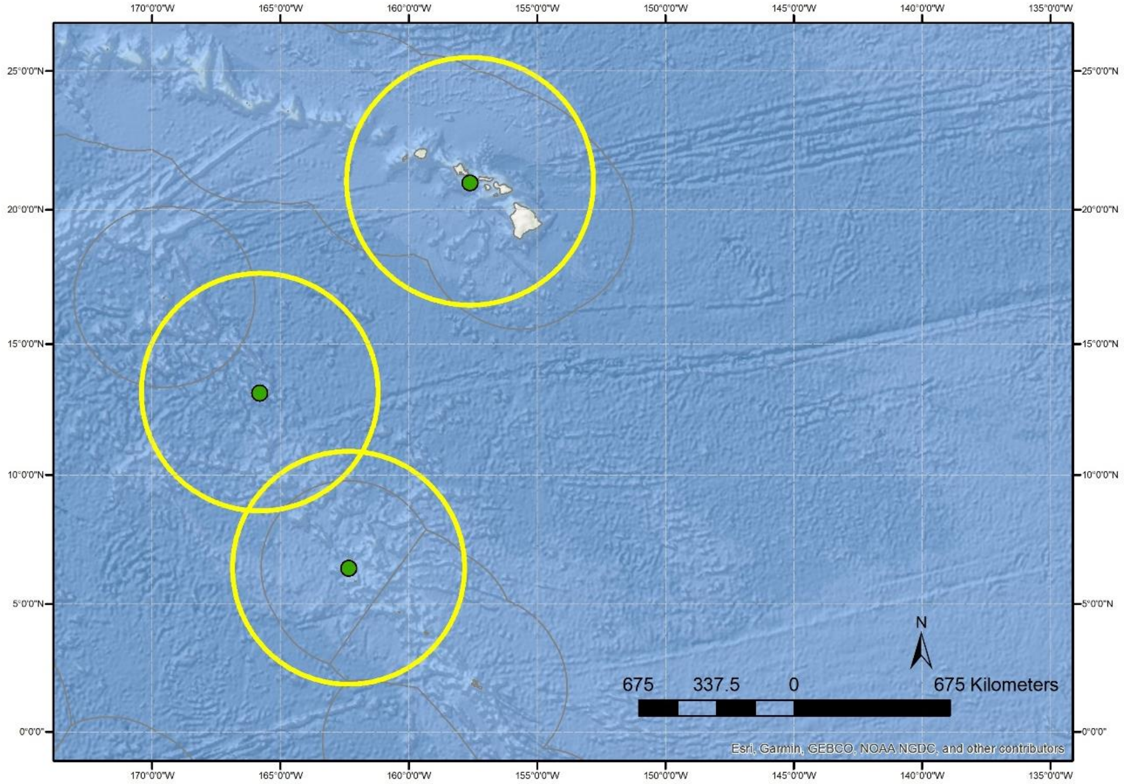

Figure 42. Map of fishing area for the DSLL fishery. Small green circles are the center points of the subpopulations in Table 19. Yellow circles represent an area around the subpopulation center points with a radius of 500 km.

Table 20. Parameters and values used for Scenarios 1 and 2 subpopulation projections.

Parameter	**Values**
Initial Subpopulation Abundance	60, 100, 200, 400, 600, 800, 1000, 1500, 1875, 2464
Post-Interaction Mortality	0.44
Maximum intrinsic Subpopulation Growth Rate (r_{max})	0.019, 0.042

In Scenario 1, if $r_{max} = 0.019$, the Hawaii and Johnston Atoll subpopulations have increasing trends if their starting population size is greater than 400; for Palmyra, there are increasing trends if the starting population size is greater than 600. If $r_{max} = 0.042$, the Hawaii and Johnston subpopulations have increasing trends if their starting population size is greater than 200; for Palmyra, there are increasing trends if the starting population size is greater than 400. (Figure 43; Appendix B). Figure 43 summarizes the impact of the intrinsic rate of subpopulation growth on the three subpopulations for Scenario 1.

In Scenario 2, if $r_{max} = 0.019$, the Hawaii and Johnston Atoll subpopulations have increasing population trajectories when starting population sizes are larger than 600; for Palmyra, increasing population trajectories occur when starting population sizes are greater than 800 (Figure 44; Appendix B). If $r_{max} = 0.042$, the Hawaii and Palmyra subpopulations have increasing population trajectories if starting population sizes are larger than 400; for Johnston Atoll, increasing population trajectories occur with starting population sizes greater than 200. Figure 44 summarizes the impact of the intrinsic rate of subpopulation growth on the three subpopulations for Scenario 2.

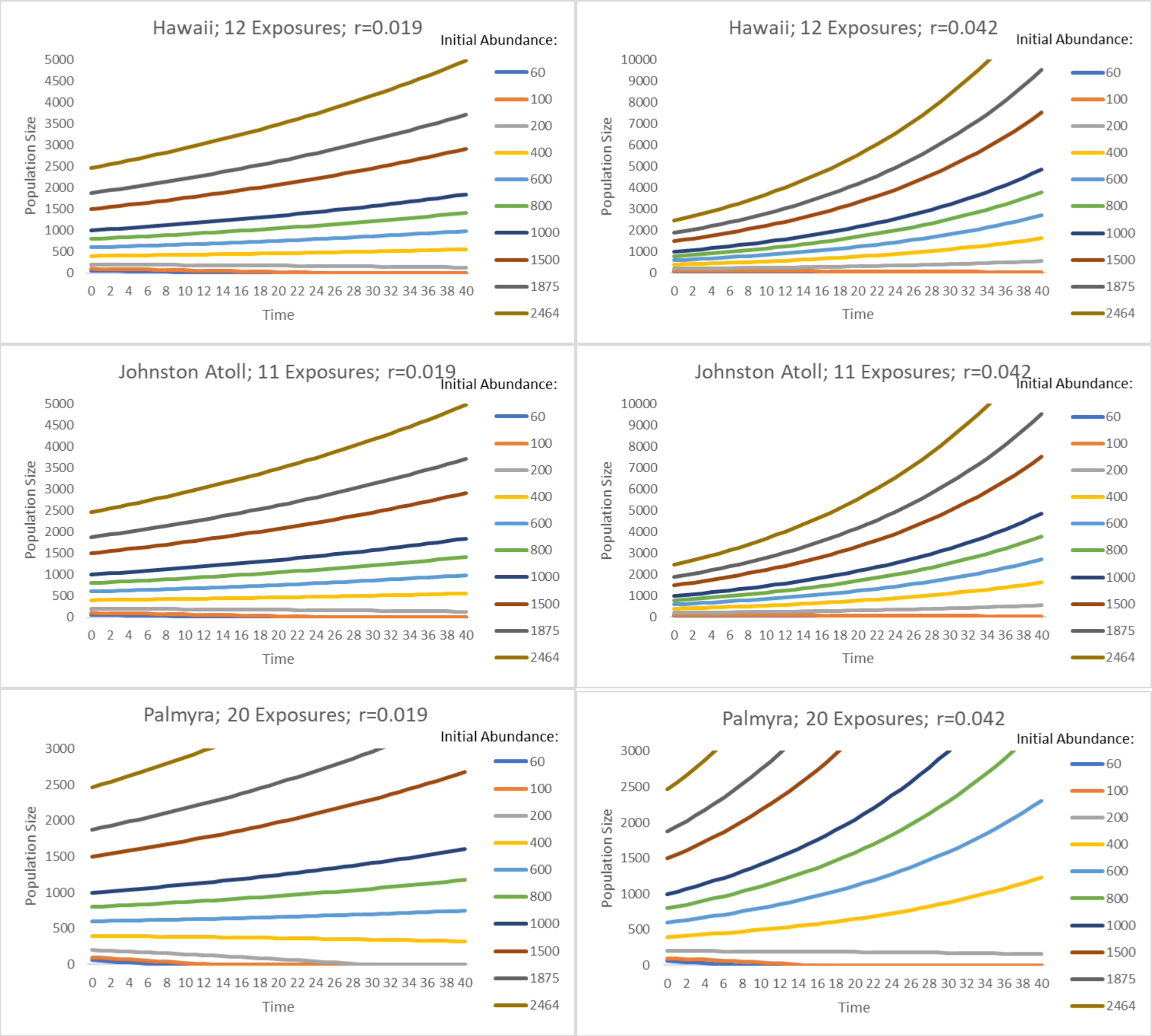

Figure 43. Results of Scenario 1 (maximum 5-year running average) for the Hawaii, Johnston Atoll and Palmyra Atoll giant manta ray subpopulations. Each graph shows population trajectories for the range of initial abundances (60 to 2,464).

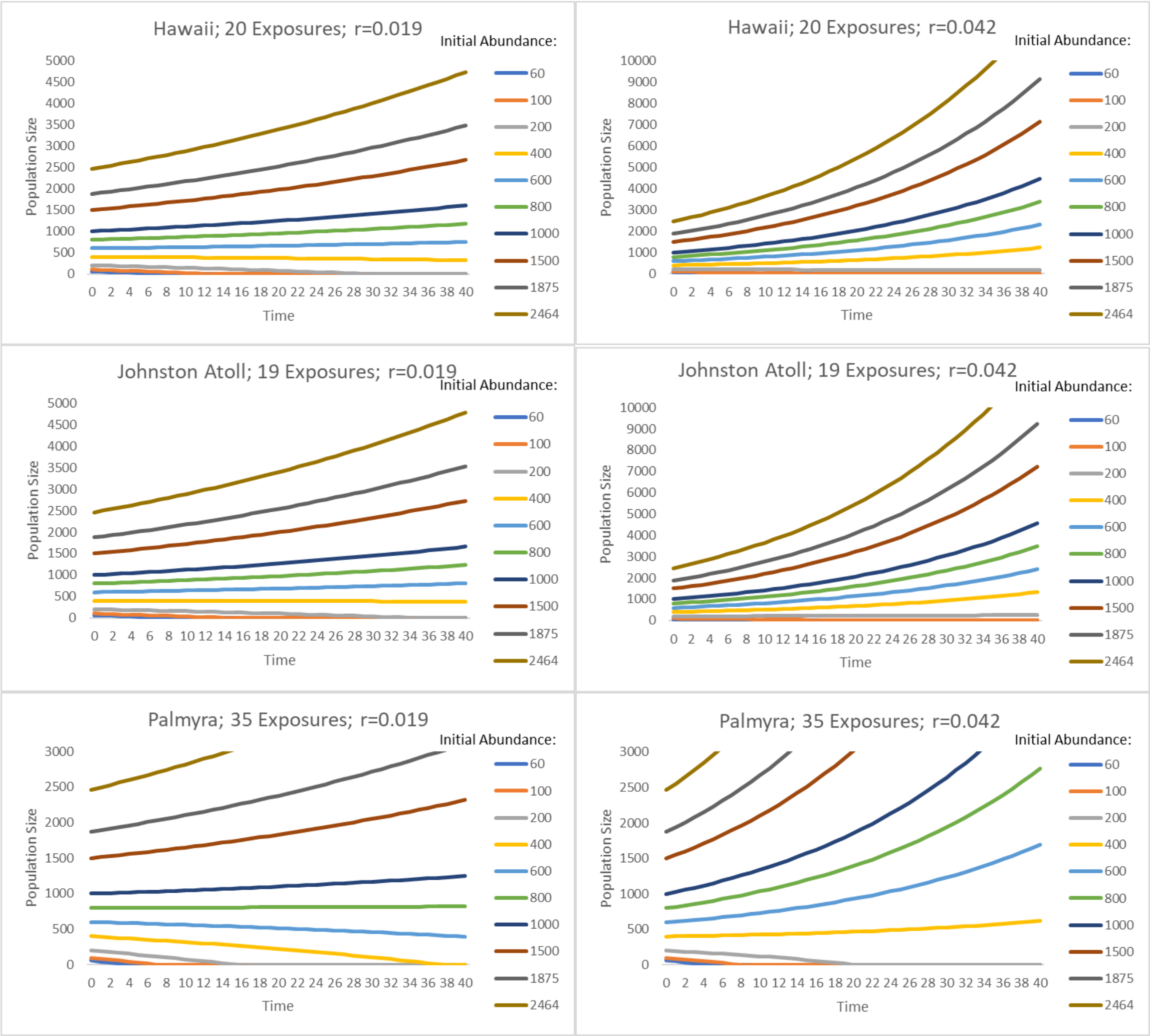

Figure 44. Results of Scenario 2 (95th percentile take levels) for the Hawaii, Johnston Atoll, and Palmyra Atoll giant manta ray subpopulations. Each graph shows population trajectories for the range of initial abundances (60 to 2,464).

The data available are not sufficient data to answer the question: "how many giant manta ray populations are likely to be exposed to the longline fisheries and what is their estimated abundance?" However, the data available suggest that giant manta rays for populations or subpopulations that are demographically and geographically independent (Lewis et al. 2015; Stewart et al. 2016a; Marshall et al. 2018; Beale et al. 2019); that those populations have probably declined by at least 30% globally and by up to 80% insignificant portions of its range (Marshall et al. 2018); and that most remaining populations are within the abundances we considered in our scenarios (CITES 2013; Burgess 2017; Beale et al. 2019; Nicholson-Jack 2020). We consider the implications of this effect on giant manta rays, as a species, in the Integration and Synthesis section of this opinion.

4.3.2 Indo-West Pacific Scalloped Hammerhead Shark

Exposure Analysis

Indo-West Pacific scalloped hammerhead sharks are rarely captured in the DSLL fishery primarily because their distribution only has slight overlap with the fishery. Most of the scalloped hammerhead sharks that interact with the DSLL are Central Pacific and not ESA-listed; we only consider those captures that are within the area of the DPS to be Indo-West Pacific (Figure 9), however we do rely on metrics from all captured scalloped hammerhead sharks for demographic information and temporal patterns. Scalloped hammerhead sharks within the range of the Indo-West Pacific scalloped hammerhead have been captured in the DSLL fishery in 2 of the 19 years between 2004 and 2022. During this 19-year time interval, fishery observers reported that three Indo-West Pacific scalloped hammerhead sharks were captured in the fishery. When these reports are adjusted to account for the percentage of observer coverage, 13 Indo-West Pacific scalloped hammerhead sharks are likely to have been captured in the fishery in the 15-year period from 2004 to the second quarter of 2018 (NMFS 2018b). Updated data received during this consultation revealed no interactions occurred with the DPS between 2018 and 2022 (Figure 45).

From 2004 to 2022, two unidentified hammerheads were observed captured by the DSLL (total of seven estimated by McCracken 2019a) within the range of the Indo-West Pacific scalloped hammerhead shark. These unidentified hammerhead sharks would be either smooth or scalloped hammerhead sharks. To estimate how many of these unidentified hammerhead sharks were likely to be Indo-West Pacific scalloped hammerhead sharks we used the same approach described above for giant manta rays and applied the Wilson Score method without continuity correction (Newcombe 1998) to estimate the 95% confidence interval around the proportion of unidentified hammerheads likely to be Indo-West Pacific scalloped hammerhead sharks. Given the limited numbers of observed hammerhead sharks within the range of the Indo-West Pacific scalloped hammerhead shark, we use all observed scalloped and smooth hammerhead shark captures in the DSLL to estimate the proportion of smooth to scalloped hammerhead sharks (Table 21). We multiplied the upper 95% CI of this proportion by the number of anticipated annual captures of Indo-West Pacific unidentified hammerhead sharks from McCracken (2019a).

Table 21. The proportion of observed incidental captures identified as scalloped hammerhead and smooth hammerhead from 2004 to 2022 and the 95% confidence interval for the proportion.

Species	Original data			Proportions		Confidence Intervals for Proportions	
	Number of Individuals Assigned to a Species	Number of Individuals Not Assigned to a Species	Sample Size	p (Species Proportions)	1-p (Not Species)	Wilson LCI (Species)	Wilson UCI (Species)
Scalloped Hammerhead Shark	32	344	376	0.09	0.91	0.06	0.12
Smooth Hammerhead Shark	344	32	376	0.91	0.09	0.88	0.94

Estimates of the number of Indo-West Pacific scalloped hammerhead sharks that are likely to be captured in the DSLL fishery each year are presented in Table 22 (estimates are from McCracken 2019a). A mean of 1 (95th percentile: 4) sharks are likely to be captured each year in the DSLL fishery. Multiplying these annual anticipated captured by the 95% CI proportion in Table 21 (0.11), we do not anticipate any unidentified hammerhead sharks to be Indo-West Pacific scalloped hammerhead sharks (i.e. a mean of 0.1 [95th percentile: 0.4] which round to zero). We also report the anticipated maximum 5-year running average and the maximum cumulative number of anticipated exposures over 5 years. We expect over any 5–year period, a maximum of 13 Indo-West scalloped hammerhead sharks will be caught in the fishery. We also expect a maximum 5 yr. running average of 1.4 unidentified hammerhead sharks (maximum 5 yr. running sum of 7) to be captured in the range of the Indo-West Pacific scalloped hammerhead shark. Based in the proportions in Table 21, we anticipate a maximum 5 yr. running average of 0.2 (maximum 5 yr. running sum of 0.8) unidentified hammerhead sharks to be Indo-West Pacific hammerhead sharks. As with all the species discussed thus far, only a portion of the scalloped hammerhead sharks expected to be captured would be reported by observers because the DSLL fishery currently has only 20% observer coverage. Lastly, it is expected that one unidentified hammerhead shark per year is likely an Indo-West Pacific scalloped hammerhead shark (NMFS 2018b; McCracken 2019a).

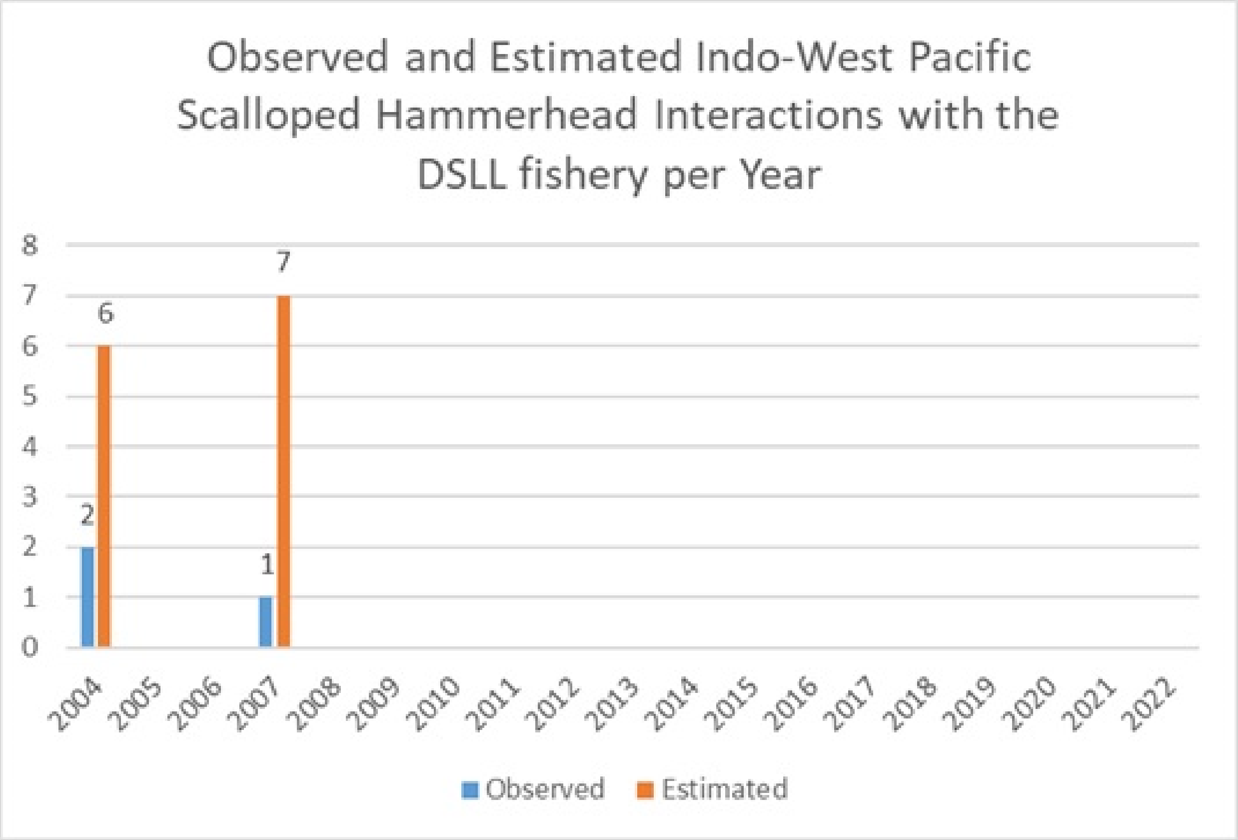

Figure 45. Indo-West Pacific scalloped hammerhead shark interactions per year in the DSLL fishery between 2004 and 2021 for observed interactions (blue bars) and estimated interactions (orange bars; McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021b, 2022b).

Table 22. Anticipated mean, 95th percentile (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021b, 2022b), maximum 5-year running average and maximum 5-year running sum exposures of Indo-West Pacific scalloped hammerhead sharks to fishing gear in the DSLL fishery. Values in parentheses are proportioned with unidentified hammerhead sharks.

Number of Years	**Mean**	**95th Percentile**
1	1 (1)	4 (4)
5-Yr Running Annual Average	2.6 (2.8)	N/A
5-Yr Cumulative Sum	13 (14)	N/A

Considering the lack of demographic data on this DPS, we reviewed all records of scalloped hammerhead sharks (inclusive of the Central Pacific scalloped hammerheads) available in the DSLL observer data. Of those, only two sharks were measured, 162 cm and 155 cm TL respectively. Neither measured animal contained any information to clarify whether claspers were present. Therefore, the gender of these sharks is unknown. This data suggests they were either juvenile female sharks, or juvenile to adult male sharks based on the evidence compared to the values presented by Miller et al. (2014). Twenty additional estimates were provided by observers ranging from 5 to 12 ft. between 2004 and 2021. Of these estimates, only two sharks are potentially adults based on size estimates, and the remainder, we suspect were juveniles. However, without additional demographic information on these interactions, we cannot be certain.

The number of scalloped hammerhead sharks captured in the DSLL fishery is too small to form a spatial pattern or a heatmap product in ArcGIS. This DPS has been caught across the latitudinal and longitudinal range of the fishery with spatial discreteness between interaction locations. However, 25% of scalloped hammerhead sharks captured in this fishery were captured in October; another 12.5% were captured in November and another 25% were captured in the months of January and February combined.

As of this writing the data for 2022 are not yet complete but to place them in context, through the end of the 3rd quarter (i.e. January to September 2022) no scalloped hammerhead sharks were observed captured and no unidentified hammerhead sharks were observed captured within the range of the Indo-West Pacific scalloped hammerhead shark. Therefore we estimate no Indo-West Pacific scalloped hammerhead sharks were captured in 2022.

Response Analysis

Between 2004 and 2021, 32 scalloped hammerhead sharks were captured with 23 released alive (71.88%) in the DSLL fishery with 3 records being attributed to Indo-West Pacific scalloped hammerheads based on capture location and expected DPS range. Again we use all records of scalloped hammerhead sharks captured in the DSLL to assess at-vessel mortality. Our observed at vessel mortality for all scalloped hammerhead sharks in this fishery is 28.13% (9/32*100; 95% CI: 15.82, 32.58) during this 18-year period. As the fishery only has 20% observer coverage, we calculated confidence intervals to reduce the uncertainty in this metric and to capture any variation due to the fact that with a small number of observed interactions, any new data will likely change the mortality rate.

Additionally, we looked at the number of unidentified hammerhead records as some could be Indo-West Pacific scalloped hammerheads and they would share the same biological characteristics and respiratory modes. When we consider all unidentified hammerhead sharks (*n*= 119) caught in the fishery during this time period, 28 were dead at the vessel and 91 were released alive. Therefore, the at vessel mortality for all unidentified hammerhead sharks in addition to the observed Indo-West Pacific scalloped hammerheads, was 23.53%. We will use the estimated at vessel mortality rate of 28.13% because it was calculated using data specific to scalloped hammerhead sharks.

Currently, we do not have a post release mortality metrics for this DPS in this fishery. However, this DPS is known to suffer high mortality based on their requirement for obligate ram ventilation and poor resiliency when compared to other shark species. Meaning scalloped hammerhead sharks need to keep moving forward to flush water across their gills to respire. At vessel mortality ranges in the literature upwards of 90% depending on the fishery (see Ellis et al. 2017 for a synopsis) and gear type. Longline vessels appear to have lower mortality when compared to gears with static nets or those that do not allow the animal to move. Furthermore, we did not find any evidence in the literature specific to scalloped hammerhead sharks comparing mortality using wire leaders versus monofilament. However, studies and modeling are present for other species that show mortality can be reduced when using monofilament line as sharks can bite the line and self-release reducing their time and effort expended while on the line (for instance see Harley et al. 2015).

Reviewing the scientific literature for scalloped hammerhead sharks and potential surrogate species in the fisheries that best exemplify the same operational and handling methods practiced in the Hawaii DSLL fishery; we found that Dapp et al. (2016) modeled a post release mortality of 19.5% for commercial pelagic longline fisheries specific to scalloped hammerhead sharks (see Table 2 in Dapp et al. 2016). We note there are limitations, specifically that this metric was derived from small sample sizes and that a confidence interval was not provided. However, currently, this is the best available scientific and commercial data available in the literature as we did not find other applicable post release mortality values in our systematic review. Therefore, the post release mortality rate of 19.5% will be used in our analyses for this DPS and note this is a data gap for the DPS. Thus, our total or overall mortality rate is 42.14% (0.2813+0.195*[1-0.2813]) using the known at vessel mortality rate and this modeled post release mortality estimate.

When we apply the total mortality rates to the number of Indo-West Pacific scalloped hammerhead sharks that are likely to be captured in the DSLL fishery each year (Table 16), the number of Indo-West Pacific scalloped hammerhead shark mortalities we expect are presented in Table 23.

Table 23. The estimated number of Indo-West Pacific scalloped hammerhead sharks that are likely to die as a result of being captured in the DSLL, assuming a total or overall mortality estimate of 43.75% (95% CI: 35.79, 48.42).

	Estimated Indo West Pacific scalloped hammerhead shark Mortalities in the DSLL	
Rate	**Mean with Estimated Total Mortality of 42.14%**	**95th Percentile with Estimated Total Mortality of 42.14%**
Annual	1	2
5-Yr Running Annual Average	1.2	N/A
5-Yr Cumulative Sum	6	N/A

On average we expect a mean of 1 (95th Percentile: 2) Indo-West Pacific scalloped hammerhead sharks to die as a result of interaction with the DSLL in a given year. Over 5 years we anticipate a maximum of 6 sharks to die. Again, we note that due to a small sample size, mortality metrics may shift as data becomes available. However, currently, this is the best assessment for mortality in the DSLL fishery given the uncertainty in post release mortality metrics. We have accounted for these effects in our analysis by rounding mortalities up to the nearest whole number.

Risk Analysis

Scalloped hammerhead sharks are commonly caught in artisanal fisheries, recreational fisheries, gillnet fisheries, and purse seine fisheries, but are not considered to be exceptionally vulnerable to longline fisheries (IOTC 2016). In the DSLL, most scalloped hammerhead sharks that interact with the fishery are Central Pacific scalloped hammerheads and not listed under the ESA. Only when the fishery fishes south of 10°N (i.e. the area around Palmyra in Figure 6) would an Indo-West Pacific scalloped hammerhead be captured and we note that the fishery has not fished in this area since 2016. Nevertheless, based on historical captures we estimate that the fishery would capture a maximum 5-year running average of 2.8 sharks with an average of 1.2 of them dying each year if it were to operate within the boundaries of the Indo-West Pacific scalloped hammerhead shark (Figure 9). Over 5 years, the maximum 5-year running sum is 14 captures with 6 of them dying as a result of their capture. Over a 40-year time horizon, if captures remain consistent with historical numbers we would anticipate a cumulative total of 112 captures and we would anticipate that 48 would die as a result of their interactions.

Although Indo-West Pacific scalloped hammerhead sharks occur across a large geographic area, we do not know if this DPS forms discrete populations and, if so, which of these populations is likely to be exposed to the DSLL fishery. The information available suggests that these scalloped hammerhead sharks represent a single population but there are no abundance estimates specific to this DPS (NMFS 2014, 2018b). The global abundance of adult scalloped hammerhead sharks based on effective population size is 280,000 (Miller et al. 2014)

As discussed previously in the Status of Listed Resources, Pacoureau et al. (2020) indicates a 67% decline from 1970 to 2018 equating to a 2.31% decline per year for the global population and we estimated a 2.28% decline for the Indo-West Pacific scalloped hammerhead shark based on ocean basin trends reported in Pacoureau et al. (2020). We simulated populations declining at a rate of 2.28% per year for a range of adult abundances from 28,000 (i.e. 10% of the global abundance) to 280,000 (i.e. 100% of the global abundance). For each starting population size, we projected the populations out 40 years at 2.28% per year declines. We then removed one mortality that might be attributable to the DSLL each year and recalculated the population trajectories. Regardless of the starting population size, adding back one mortality did not change the population trajectories, the new trajectories were -2.28% per year. We consider the implications of this effect on Indo-West Pacific scalloped hammerhead sharks in the Integration and Synthesis section of this opinion.

4.3.3 Oceanic Whitetip Shark

Exposure Analysis

Oceanic whitetip sharks have been incidentally captured in the Hawaii DSLL fishery every year since 1994. However, it was not until 2004 that observations of the species in this fishery were separated from the Hawaii SSLL fishery. From 2004 to 2022, there have been 6,139 *observed* interactions in the DSLL fishery. When we adjusted these numbers to account for the percentage of observer coverage, approximately 31,467 oceanic whitetip sharks are likely to have been captured in the Hawaii DSLL fishery between 2004 and 2022 (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021b, 2022b).

We expect that the majority of sharks will be hooked rather than entangled (NMFS 2018a). Table 25 contains our estimates of the number of oceanic whitetip sharks that are likely to be incidentally captured in the Hawaii DSLL fishery each year (values from 2004 to 2021are from McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021b, 2022b; NMFS 2021b). The value for 2022 was estimated based on the expansion factor with 471 observed captures and observer coverage of 20.22% (471/[20.22/100]=2,329.4 rounded to 2,329). On average, based on data from this fishery we would expect 1,708 oceanic whitetips to be incidentally captured each year in the Hawaii DSLL fishery (95th percentile = 3,185). Only a portion of these sharks would be reported by observers because the Hawaii DSLL fishery maintains about 20% observer coverage each year.

As noted in the Description of the Proposed Action section of this opinion, the Hawaii DSLL fishery transitioned from wire leaders to monofilament leaders during this consultation with voluntary transition occurring in 2021 and full transition to monofilament leaders required by May 31, 2022 (87 FR 25153; 04/28/2022). This gear change is expected to decrease the number of oceanic whitetip sharks that are hauled to the vessel by 32% (95% CI: 24% to 40%; Bigelow and Carvalho 2021). Sharks that bite through monofilament leader lines would not be hauled in and would not likely be recorded as animals that are incidentally captured despite that they may still experience some amount of harm as a result of their interaction. In this analysis, we refer to these animals as *unobservable* due to self-releasing via biting through the monofilament leader, while the remaining 68% (95% CI: 60% to 76%) of interactions are *observable*. If the assessment of Bigelow and Carvalho (2021) is accurate we anticipate declines in observable oceanic whitetip shark captures of 24% to 40%, however, as the regulation was implemented in May of 2022, not enough time has elapsed to ascertain if this decline will be realized.

As noted previously, we estimate that 2,329 oceanic whitetip sharks were captured in 2022. Looking only at the 3rd and 4th quarters, after implementation of the regulation restricting the use of wire leaders, there were 293 observed oceanic whitetip shark captures. We also compared 3rd and 4th quarter observed and estimated captures from 2017 to2022 and found that both observed and estimated captures in 2022 and the rate of captures (observed sharks per observed hooks) were higher than the previous five years (Table 24).

Table 24. Observed and estimated 3rd and 4th quarter (July 1 through December 31 2022) oceanic whitetip shark captures in the DSLL. Estimated captures were calculated using the expansion factor ([Observed Captures/Observer Coverage]*100).

Year	3rd and 4th Quarter Observed Oceanic Whitetip Sharks	3rd and 4th Quarter Observer Coverage	3rd and 4th Quarter Observed Hooks	3rd and 4th Quarter Estimated Oceanic Whitetip Sharks (observed captures/ observer coverage)	3rd and 4th Quarter CPUE (observed sharks/observed hooks)
2017	140	21.28%	5,573,185	658	0.000025
2018	158	20.81%	6,189,951	759	0.000026
2019	265	20.05%	6,826,267	1,322	0.000039
2020	189	18.05%	5,369,269	1,047	0.000035
2021	200	19.85%	6,100,798	1,008	0.000033
2022	293	22.08%	6,487,382	1,327	0.000045

Since the fishery was required to switch to monofilament (87 FR 25153) in May of 2022, we only have two quarter's worth of data available on the capture of this species at the time of this writing. The data from the 3rd and 4th quarters of 2022 gives us caution in applying a reduction of observable captures due to the switch to monofilament. Bigelow and Carvalho (2021) found a mean reduction of 32%, however the 3rd and 4th quarters of 2022 had the highest reported number of interactions for those quarters in the fisheries recent history. Because they are collected directly from the fishery under conditions the same as the proposed action, the 3rd and 4th quarter data constitute the best available information. Therefore, for this biological opinion we focus our analysis on the full estimate of observed captures. In the recent supplemental biological opinion for oceanic whitetip sharks in the DSLL (NMFS 2022b), we focused our analysis on a 32% reduction in observable captures; however, at that time, the data from 2022 were not yet available. We note, the range considered in our analysis would capture a 32% reduction in captures if the assumptions of Bigelow and Carvalho (2021) are correct.

Table 25. Numbers of oceanic whitetip sharks that are likely to be incidentally captured in the Hawaii DSLL fishery based on existing data collected under a mixed (mostly wire) leader fishery each year (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021b, 2022b; NMFS 2021b) and the maximum 5-year running average and cumulative sum.

<table>
<tr><th colspan="3">Estimated number of oceanic whitetip sharks incidentally captured in the Hawaii DSLL fishery based on the fishery’s previous gear configuration using wire leaders (32% reduction with monofilament leaders)</th></tr>
<tr><th>Rate</th><th>Mean</th><th>95th Percentile</th></tr>
<tr><td>Annual</td><td>1,708</td><td>3,185</td></tr>
<tr><td>5-Year Running Average</td><td>2,117.8</td><td>N/A</td></tr>
<tr><td>5-Year Cumulative Sum</td><td>10,589</td><td>N/A</td></tr>
</table>

To examine seasonality of incidental captures, we aggregated data of observed interactions with oceanic whitetip sharks in this fishery by month for the all years from 2004 through 2022 (Figure 46). Oceanic whitetip sharks are caught all months of the year with interactions peaking in April and May, and again later in the calendar year, October through December. Based on the data, about 52% of observed oceanic whitetip sharks were incidentally captured in April and May (~21%) and October through December (~31%) from 2004 to 2022 fishery wide.

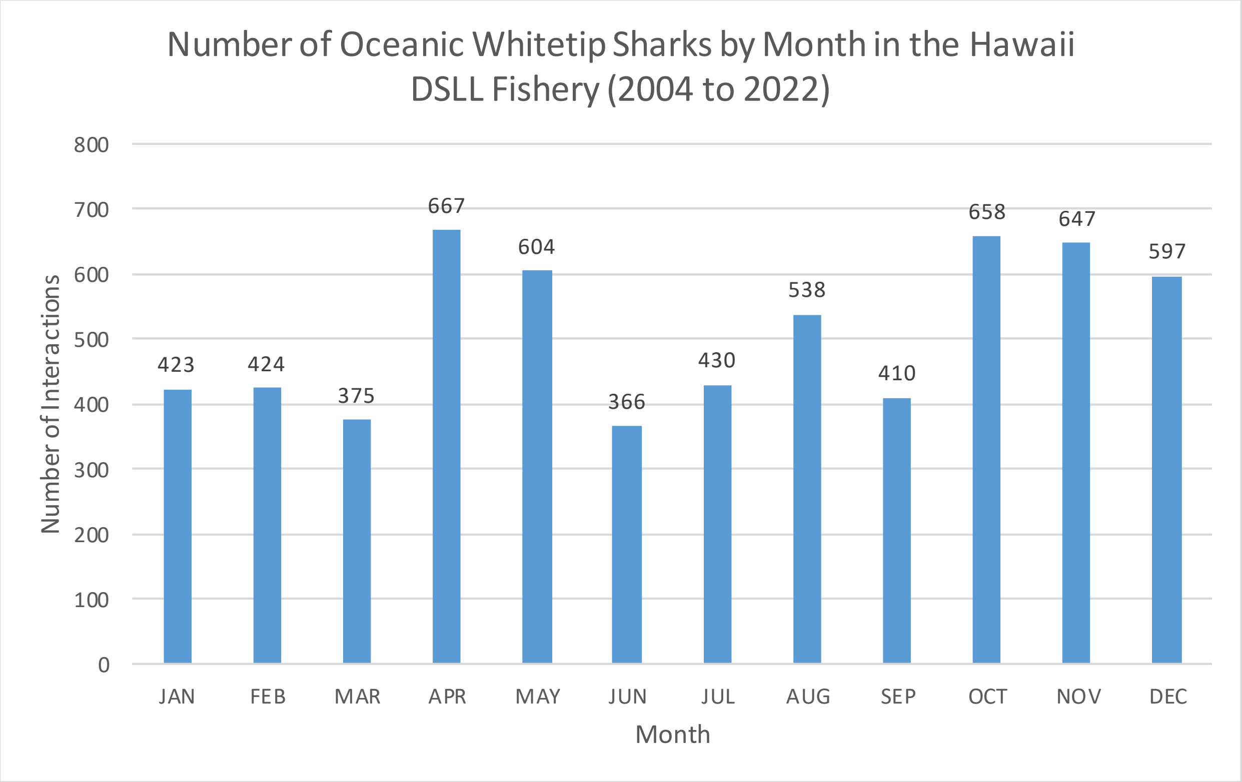

Figure 46. Number of observed oceanic whitetip sharks incidentally captured by month in the Hawaii DSLL fishery from 2004 to 2022 (*n* = 6,139).

We also examined the annual number of observed oceanic whitetip sharks incidentally captured in this fishery and compared that against the estimated number of total interactions from McCracken and others in Figure 47. Based on the raw data, the highest number of observed interactions occurred in 2015 and 2021, with corresponding high estimated captures (Figure 47). Notably the highest level of estimated captures occurred in 2021.

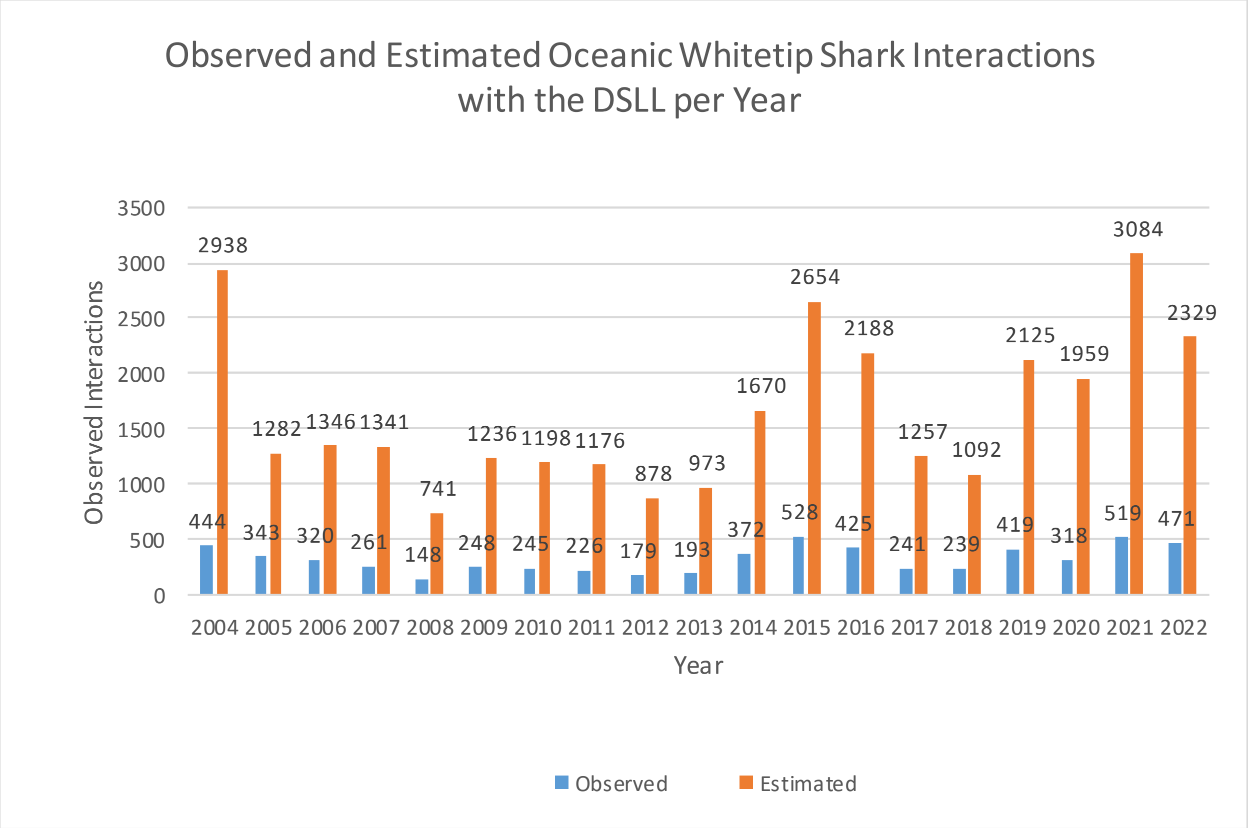

Figure 47. Oceanic whitetip shark interactions per year in the DSLL fishery between 2004 and 2022 for observed interactions (blue bars) and estimated interactions adjusted for observer coverage (orange bars; McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021b, 2022b). Estimated interactions for 2022 were calculated using the expansion factor based on the observed captures and the observer coverage.

As we did with the other species in this biological opinion, to visualize where incidental captures occur, we used ArcGIS/ArcMap kernel density tool to create a heat map illustrating the *relative* number of interactions or effort in an area (Figure 48).

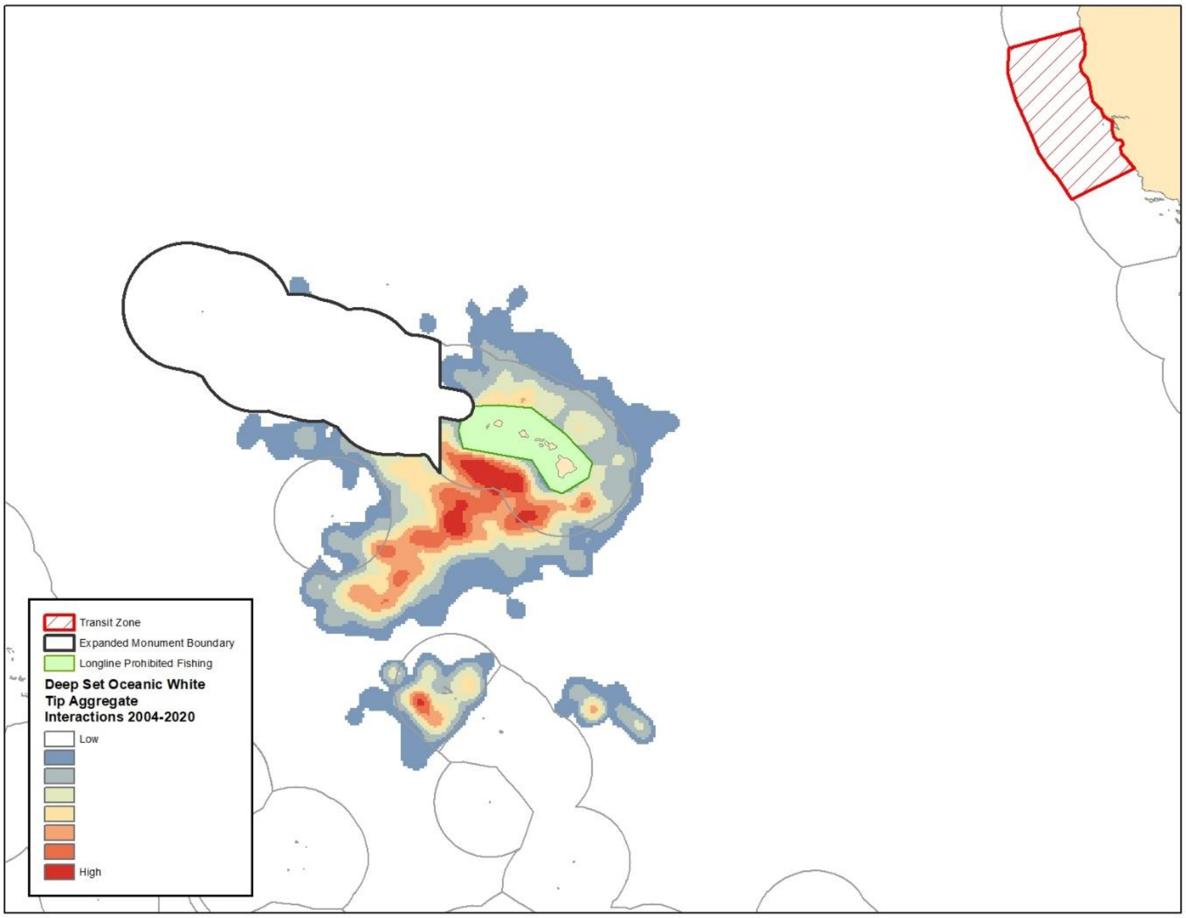

Figure 48. Geographical representation of observed aggregate oceanic whitetip shark interactions with the DSLL fishery between 2004 and 2020 (*n*= 5,149).

Based on observed data, interactions with oceanic whitetip sharks do not occur throughout the operational range of the fishery. Rather, higher relative densities of interactions occur within specific areas, and most notably to the south and southwest of the MHI and further south by Palmyra. To highlight this point, we compare the interactions with aggregated fishing effort (Figure 49). Both maps are scaled the same; however, due to the vast number of sets depicted in the effort data, the blue areas look different as lower effort is present throughout an extensive geographical range. The blue shown in the interactions heatmap has a higher density than those blues depicted in the effort data. While 2021 and 2022 data is not shown in the heatmap, all incidental interactions that occurred with the species are within the geographical range shown. Almost all of the interactions are within the area southwest of the MHI as depicted in the heatmap. Therefore, we would not expect any significant changes in the data, but only strengthen the density values within this geographical area.

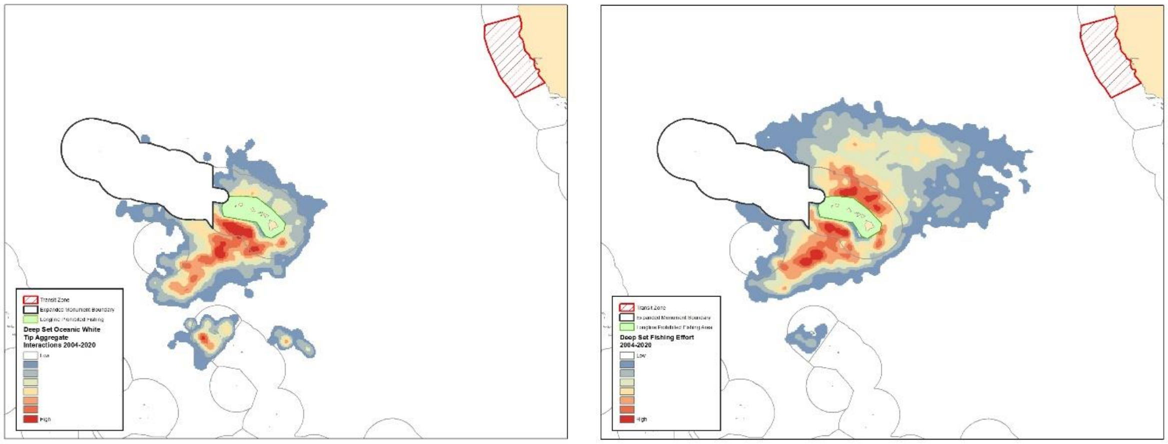

Figure 49. Heatmap comparison of observed oceanic whitetip shark interactions (left; *n* = 5,149) versus aggregated observed fishing effort during this same time frame (right; *n*= 65,245 from 2004 to 2020).

These heatmaps suggest that while there is a significant amount of fishing occurring on the north side of the MHI, there does not appear to be a corresponding high number of interactions occurring with this species in this area. The higher density areas of interactions with oceanic whitetip sharks are to the southwest of the MHI, and are similar between the two maps in shape and density. Based on our review of the data on interactions, in April and May very few interactions occur north of the Hawaiian Islands. This is consistent with the range of the species that is primarily south of the MHI (Figure 10). In October and November, interactions appear to be closer to the MHI on the south side of the islands, with more interactions occurring north of the islands. We discussed these fishing trends with the fishing industry at the oceanic whitetip shark recovery workshop held in Honolulu Hawaii on April 23-24, 2019. We were informed that the industry fishes closer into the islands due to ocean conditions during that time of year, to reduce financial costs, and to reduce the time spent at sea. Therefore this pattern may be more indicative of where fishing is occurring than the biology of oceanic whitetip sharks. Without further research, we cannot be certain if this pattern is important to the species.

The heatmap also shows a distinct aggregation of interactions around Palmyra Atoll. Overall, 470 interactions have occurred with oceanic whitetip sharks at this location from 2004-2020 and this area is spatially separated by approximately 190 nm from the northern hotspot identified by the interactions heatmap (south of Johnston Atoll EEZ as depicted on the interactions map). The industry informed NMFS biologists at the oceanic whitetip shark recovery workshop held in Honolulu Hawaii on April 23-24, 2019, that they (the industry) no longer fish near Palmyra due to the logistical difficulties of reaching the area. Namely, it was more costly, took more time to reach the destination, and resulted in more days at sea. However, there are no regulations in place to prevent the fishery from resuming activities here and we include the incidental captures that occurred in this area in our assessment. When comparing interactions to effort, the interaction map shows a higher density of interactions compared to a lower effort.

Observer data confirm the last time vessels fished and interacted with oceanic whitetip sharks in this area was in 2015. Furthermore, in the BE for this opinion (NMFS 2018b), NMFS SFD does not believe this fishery overlaps with United States WCPO purse seine fishery now that the DSLL no longer fishes this geographical area. Interactions that occurred in the Palmyra hotspot peaked in April. Only two vessels accounted for all interactions (*n*= 51) in 2015 at this broad location even though 43 vessels have incidentally caught oceanic whitetip sharks in Palmyra's EEZ over the time period assessed. We highlight that the heatmap projects all sets, by all vessels, in this geographic area to protect data confidentiality. However, these data show how a small number of vessels can have a disproportionate effect on a population by fishing in an area where larger concentrations of sharks may be present at a certain time of year.

Response Analysis

Oceanic whitetip sharks in the Hawaii DSLL fishery likely experience a range of responses to being incidentally captured in this fishery, from altering behaviors to injury and death. In this analysis, while we focus on the mortality and injury that leads to latent mortality of oceanic whitetip sharks, we have accounted for sublethal responses in our analysis to the best of our ability using the best available information.

The total mortality of oceanic whitetip sharks is the combination of the total number of *observable* individuals that we expect to die with the number of *unobservable* individuals that are expected to die despite biting through monofilament leaders and self-releasing from the fishing gear. This calculation is represented as follows:

$$\mathrm{TMO} + \mathrm{TMU} = (\mathrm{TO}*\mathrm{TMV}) + (\mathrm{TU}*\mathrm{TMB})$$

In the above equation TMO equals total mortality of *observable* oceanic whitetip sharks, which is calculated by multiplying the estimated total number of observable oceanic whitetip sharks (TO) against a total mortality value that includes at-vessel and post-interaction mortality for sharks that are brought to the vessel (TMV). TMU is equal to the total mortality of *unobservable* oceanic whitetip sharks given the fishery's gear change. TMU is calculated by multiplying the estimated total number of *unobservable* oceanic white tip sharks (TU) multiplied by the post-interaction mortality of sharks that are hooked but free themselves by biting off the line (TMB). Combined, this gives us our best estimate of the total number of oceanic whitetip sharks that we expect will die as a result of interacting with the Hawaii DSLL fishery. For this biological opinion, the total mortality of oceanic whitetip sharks is estimated as:

$$\mathrm{TMO} = (\mathrm{TO}*\mathrm{TMV})$$

We present our best estimates of future mortality of oceanic whitetip sharks resulting from the Hawaii DSLL fishery in Table 26. In the sections that follow, we explain these values in greater detail, including how they are calculated and why they are the best available estimates for this analysis.

Table 26. Estimates of number of oceanic whitetip sharks that will be incidentally captured and killed within the Hawaii DSLL fishery.

Metric	**Total Interactions**	**Total Mortality**
Annual Mean	1,708	557
Annual 95th Percentile	3,185	1,038
Max. 5-Year Running Average	2,117.8	690.4
5-Year Cumulative Sum	10,589	3, 452

Estimated Total Mortality of Oceanic Whitetip Sharks

Our estimate of the total number of oceanic whitetip sharks expected to die as a result of the Hawaii DSLL relies upon our calculated at-vessel mortality for this fishery for observed oceanic whitetip sharks using historical data from 2004 to 2021, and our best estimate of post-interaction mortality for the species. This calculation is as follows:

$$TMO= (AVM+PIM(1-AVM))TO$$

Here, the total mortality of oceanic whitetip sharks is represented by TMO and is calculated by combining our calculated at-vessel mortality (AVM) with the mean post-interaction mortality (PIM) rate of Hutchinson et al. (2021) for oceanic whitetip sharks with monofilament leaders and trailing gear, multiplied by the estimated total number of oceanic whitetip sharks (TO) that we expect to interact with this fishery.

Our calculated best estimate of at-vessel mortality for this fishery is 22. 52%. Between 2004 and 2022, a total of 6,052 oceanic whitetip sharks were incidentally captured by the Hawaii DSLL fishery with release conditions of alive or dead and of those, 1,363 were dead for an at-vessel mortality of 22.52 % ([1,363/6,052]x100 = 22.52) with 77.48 % released alive. Note that we did not include sharks with release conditions of "retained" or "unknown" in this calculation.

Our analysis also relies upon the mean post-release mortality for oceanic whitetip sharks from Hutchinson et al. (2021) and is based on the 30-day post-interaction survival rates for oceanic whitetip sharks on monofilament leaders with trailing gear (0.13). Since this is a new gear requirement for the Hawaii DSLL fishery, which encourages removal of trailing gear from sharks but allows for some exceptions to the removal of trailing gear, we do not know what proportion of sharks would be released with trailing gear in this fishery, or the average length of that trailing gear. Therefore, we consider the post release mortality rate of 0.13, from Hutchinson et al. (2021) as the best estimate for post release mortality in this fishery for oceanic whitetip sharks, because it is developed with fishery specific data, and because this value is consistent with published studies for pelagic longlines and *Carcharhinus* spp. in general, and because we expect some unknown portion of the sharks will be released with trailing gear.

In other words, we calculated total mortality of observable oceanic whitetip sharks as follows:

$$TMO= (0.2252+0.13(1-0.2252)) \quad TO$$

Hutchinson et al. (2021) tagged and released 27 oceanic whitetip sharks incidentally captured in the Hawaii DSLL fishery with satellite tags. Of these, one tag didn't report and one tag had to be removed due to concerns with handling at release. Of the remaining 25 tags, 4 sharks died and 21 survived for a mean post-interaction mortality rate for animals brought to the vessel of 0.16 (95% CI: 0.064 to 0.35; inferred from data in Hutchinson et al. 2021 and using the Wilson Score method to construct confidence intervals). An additional 35 oceanic whitetip sharks were tagged in the ASLL with two that didn't report and two that had to be removed due to handling concerns. Of the remaining 31 tags, 4 died and 27 survived for a mean post-interaction mortality rate of 0.13 (95% CI 0.051 to 0.29; inferred from data in Hutchinson et al. 2021 and using the Wilson Score method to construct confidence intervals). Combining these data in a random effects model (Borenstein et al. 2009), we estimate a mean mortality rate of 0.14 (95% CI: 0.11 to 0.17; inferred from data in Hutchinson et al. 2021). Most of the mortalities occurred within 4 days of release, with one delayed mortality occurring at 22 days post release (Hutchinson et al. 2021); therefore the mortalities can be attributed to the shark's interaction with the fishery with a high degree of confidence.

Of the 56 tags used to estimate post-interaction mortality rates in the DSLL and ASLL, 16.1% were on sharks released alive but condition unknown (A), 73.2% were on sharks released alive in good condition (AG), and 10.7% were on sharks released with injuries (I; Table 27; based on data in Table S1 of Hutchinson et al. 2021). In the same tagging study, information was collected on condition and handling data on 19,572 elasmobranchs for the two fisheries (Hutchinson and Bigelow 2019). The proportion of those 19,572 elasmobranchs that were oceanic whitetip sharks was not reported, but the proportions of condition at release was reported for oceanic whitetip sharks (see Table 6 in Hutchinson and Bigelow 2019). Of the oceanic whitetip sharks released alive, 82.2% were condition code AG, 9.9% were condition code A, and 7.8% were condition code I. Of the 56 tags, 14 were MiniPAT tags designed for collecting long-term data (6 in the DSLL and 8 in the ASLL), and all of these were placed on AG condition sharks to maximize the probability of longer-term data reporting (Hutchinson et al. 2021).

Table 27. Capture and release conditions for oceanic whitetip sharks tagged (reporting tags retained in the study only) in the Hawaii DSLL and ASLL fisheries (Hutchinson et al. 2021). A= Alive; AG= Alive in Good Condition; I = Injured.

Capture Condition	Release Condition	Number	Percent
A	A	8	14.3
AG	A	1	1.8
AG	AG	41	73.2
AG	I	3	5.35
I	I	3	5.35
	Totals:	56	100

Hutchinson et al. (2021) conducted Bayesian survival analyses on the tag data for both fisheries. Overall they found that condition at release, branchline leader material, and the amount of trailing gear left at release had the largest impact on post-release mortality. They present post-interaction survival rates at 1, 30, 60, 180, and 360 days post-interaction. We focus on the 30 day survival rates as being the most indicative of mortalities resulting from interaction with the fishery. Hutchinson et al. (2021) report 30 day survival rates for oceanic whitetip sharks interacting with the DSLL to be 0.84 (90% Credible Interval: 0.34-0.98; see Table 7 in Hutchinson et al. 2021), suggesting a slightly higher mortality rate of 0.16 (1-0.84) than estimated from the raw tag results noted above (0.13). Hutchinson et al. (2021) post-interaction survival rates for oceanic whitetip sharks on monofilament leaders (with and without trailing gear, based on the 30-day survival rate) are summarized in Table 28. Mean post-interaction mortality rates for oceanic whitetip sharks in the DSLL are estimated 0.09 (no trailing line) to 0.13 (trailing line remaining on shark) using monofilament leaders.

Table 28. Post-interaction and total mortality rates based on monofilament leader and trailing line for the DSLL (based on Table 9 in Hutchinson et al. 2021).

Mortality Metric	**Trailing Line Left**	**Mean Mortality Rate**	**90% Credible Interval**
Post-Interaction	Yes	0.13	0.01 – 0.62
Post-Interaction	No	0.09	0.01 – 0.52
Total Mortality	Yes	0.30	0.2 – 0.69
Total Mortality	No	0.27	0.2 – 0.61

We also conducted a systematic literature review to place the mortality rates from Hutchinson et al. (2021) in context with results of other studies on other similar fisheries[9]. The mean post-interaction mortality rates from our meta-analysis, especially those for fisheries using monofilament leaders and for sharks in the same genus as oceanic whitetip sharks are consistent with the post-interaction mortalities rate reported for oceanic whitetip sharks with trailing gear as reported by Hutchinson et al. (2021).

Risk Analysis

In this section of this biological opinion, we examine the probable consequences of capturing over 84,000 oceanic whitetip sharks in the Hawaii DSLL fishery for the next 40 years (N=2117.8*40), and killing nearly 28,000 individuals during the same time period (N=27,616=690.4*40). As noted previously, most of our analysis focuses on capture, mortality, and injury that can be associated with latent mortality.

Based on a mean number of interactions of 1,708 sharks, we estimate that about 557 oceanic whitetip sharks would die each year in this fishery. The 95th percentile represents our upper estimate of interactions in any given year, which is not to say this number cannot be exceeded but that, given the data, there is a 95% probability that the true number of interactions that this fishery will have with oceanic whitetip sharks will be lower than 3,185 individual interactions. Based on the maximum 5-year running average, we estimate that about 2,118 oceanic whitetip

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[9] A total of 325 studies were reviewed, of those 46 references provided information on at-vessel and/or post-interaction mortality rates for sharks. References are available upon request.

sharks will be incidentally captured each year (N=10,589 individuals every five years) in the Hawaii DSLL fishery. From these animals, we estimate that 690 individuals will die as a result of the interaction (N=3,452 mortalities in 5 years).

The best available genetic information for this species suggests that one population occurs in the Pacific Ocean. Although distinctions are often made between the East Pacific and the West Pacific, this distinction may be one of convenience based on management and is potentially biologically arbitrary. The only formal stock assessment for the Pacific only represents part of the population that likely occurs within the Pacific Ocean–the West and Central Pacific portion of the population's range (aka. the West and Central Pacific stock). To assess the risk that the Hawaii DSLL fishery has on the Pacific population of oceanic whitetip sharks, we first started by examining the effect of the action on the West Pacific stock because we have an abundance estimate for that portion of the Pacific population. Given that this is a partial population estimate this comparison would overestimate the effect of the Hawaii DSLL fishery on the Pacific population of oceanic whitetip sharks; therefore, we included one other scenario in our analysis.

In the first scenario, we treat the West Pacific stock estimate of 775,214 oceanic whitetip sharks in 2019 as a *minimum* population estimate for oceanic whitetip sharks in the Pacific Ocean. In our second scenario, we assume the West Pacific stock estimate of 775,214 oceanic whitetip sharks represents 60% of the total Pacific Ocean population (N= 1,292,023). In both scenarios, we assumed that the population is currently either declining at a rate of 0.13% per year (Rice et al. 2021 with a 10% reduction in fishery mortality) or increasing at a rate of 0.36% annually (Rice et al. 2021 with a 20% reduction in fishery mortality) with the DSLL as currently managed (Tables 26 and 27). As noted in the Status of the Listed Species section of this biological opinion, we believe the actual population trend of oceanic whitetip sharks in the Pacific Ocean is between the median values of -0.13% to +0.36% provided by Rice et al. (2021) under mortality decreases of 10 to 20% in WCPO longline fisheries (with the DSLL as currently managed). Though, the subsequent analysis of Bigelow et al. (2022) indicates the rate is likely positive with the implementation of conservation measures.

Under the first scenario, the Hawaii DSLL fishery interacts with about 0.22% of the Pacific population, and 0.07% of the population dies as a result of their interaction with the Hawaii DSLL fishery in the first years based on mean estimated interactions (Table 29, Table 30). If the population declines at a rate of 0.13% per year, by 2062 the population will have declined by 5% and the fishery will interact annually with about 0.29% of the population with 0.09% being killed. If the population increases at a rate of 0.36% per year, by 2062 the population will have increased by 15.5% and the fishery will interact annually with about 0.23% of the population with 0.08% being killed.

Accumulating incidental captures over the initial 10 years suggests that up to 0.85% of the population would be killed over 10 years if the population is increasing; 0.94% if the population is declining. Although this scenario is based on what may be described as a minimum population estimate, we believe that using this stock assessment to represent the entire population in the Pacific Ocean presents an unrealistic picture of the actual size of the Pacific population and results in overestimating the effect of the fishery on the species.

Therefore, we also examined the effect of the action on the species, assuming that the West Pacific stock estimate of 775,214 oceanic whitetip sharks represents about 60% of the total Pacific Ocean population. We developed this proportion by using ArcGIS to estimate the portion of the species expected range (i.e. 30N and 35S) in the Pacific Ocean that is covered by the WCPO and the species' only formal stock estimate in the Pacific Ocean. Assuming the West Pacific stock represents about 60% of the total population within the Pacific Ocean, then we expect that the total Pacific Ocean population is comprised of over a million oceanic whitetip sharks. Under this scenario, during the initial years, the Hawaii DSLL fishery interacts with approximately 0.13% of the total population annually, and kills approximately 0.04% of the population each year across the range of population trends considered.

If the population declines at a rate of 0.13% per year, by 2062 the population will have declined by 5% and the fishery will interact annually with about 0.16% of the population with 0.06% being killed. If the population increases at a rate of 0.36% per year, by 2062 the population will have increased by 15.5% and the fishery will interact annually with about 0.14% of the population with 0.05% being killed.

Accumulating incidental captures over the initial 10 years suggests that up to 0.51% of the population would be killed over 10 years if the population is increasing; 0.57% if the population is declining (Table 29, Table 30). Our estimates assume that the estimated population trends in the WCPO is indicative of the trend for the entire Pacific population.

Table 29. Estimated effect of Hawaii DSLL fishery on the Pacific population of oceanic whitetip sharks incidentally captured and killed by the proposed action projected out 40 years for scenarios 1 and 2 and the assumption of a 10% reduction in fishery mortalities (-0.13% per year population decline).

	Year	**Oceanic Whitetip Sharks in the Pacific Ocean**					
		Estimated Population Abundance		**Estimated of percent captured**		**Estimated Percent Killed**	
		WCPO	Pacific	WCPO	Pacific	WCPO	Pacific
Annual Mean	2022	772,196	1,286,995	0.22	0.13	0.07	0.04
Max. 5-Year Running Average	2022	772,196	1,286,995	0.27	0.15	0.09	0.05
Max. 5-Year Running Average	2032	762,222	1,270,372	0.28	0.15	0.09	0.05
10-year based on 5-yr running average	2032	762,222	1,270,372	2.78	1.50	0.91	0.54
Max. 5-Year Running Average	2042	752,377	1,253,965	0.28	0.15	0.09	0.06
10-year Mean Projected 20 years	2042	752,377	1,253,965	2.81	1.52	0.92	0.55

	Year	Oceanic Whitetip Sharks in the Pacific Ocean					
		Estimated Population Abundance		Estimated of percent captured		Estimated Percent Killed	
Max. 5-Year Running Average	2052	742,660	1,237,769	0.29	0.15	0.09	0.06
10-year Mean projected 30 years	2052	742,660	1,237,769	2.85	1.54	0.93	0.56
Max. 5-Year Running Average	2062	733,067	1,221,782	0.29	0.16	0.09	0.06
10-year Mean projected 40 years	2062	733,067	1,221,782	2.89	1.56	0.94	0.57
95th Percentile	2062	733,067	1,221,782	0.43	0.26	0.14	0.08

Table 30. Estimated effect of Hawaii DSLL fishery on the Pacific population of oceanic whitetip sharks incidentally captured and killed by the proposed action projected out 40 years for scenarios 1 and 2 with the assumption of a 20% reduction in fishery mortalities (0.36% per year population increase).

	Year	Oceanic Whitetip Sharks in the Pacific Ocean					
		Estimated Population Abundance		Estimated of percent captured		Estimated Percent Killed	
		WCPO	Pacific	WCPO	Pacific	WCPO	Pacific
Annual Mean	2022	783,632	1,306,052	0.22	0.13	0.07	0.04
Max. 5-Year Running Average	2022	783,632	1,306,052	0.27	0.16	0.09	0.05
Max. 5-Year Running Average	2032	812,356	1,353,925	0.26	0.16	0.08	0.05
10-year based on 5-yr running average	2032	812,356	1,353,925	2.61	1.56	0.85	0.51
Max. 5-Year Running Average	2042	842,134	1,403,557	0.25	0.15	0.08	0.05
10-year Mean Projected 20 years	2042	842,134	1,403,557	2.51	1.51	0.82	0.49
Max. 5-Year Running Average	2052	873,002	1,455,005	0.24	0.15	0.08	0.05

	Year	Oceanic Whitetip Sharks in the Pacific Ocean					
		Estimated Population Abundance		Estimated of percent captured		Estimated Percent Killed	
10-year Mean projected 30 years	2052	873,002	1,455,005	2.43	1.46	0.79	0.47
Max. 5-Year Running Average	2062	905,002	1,508,338	0.23	0.14	0.08	0.05
10-year Mean projected 40 years	2062	905,002	1,508,338	2.34	1.40	0.76	0.46
95th Percentile	2062	905,002	1,508,338	0.35	0.21	0.11	0.07

Because they are based on models with multiple assumptions, we expect that the population sizes used in both of the scenarios are imperfect. This is because we lack reliable evidence regarding the true size of the Pacific Ocean population of oceanic whitetip sharks. We have reason to believe that the one formal stock assessment on the species that was conducted in the WCPO by Tremblay-Boyer et al. (2019), is not representative of the entire population of oceanic whitetip sharks in the Pacific Ocean. We used the best data, both on the range, the portion of the range, and the available stock assessment, to develop a reasoned scenario that we believe likely paints a more realistic picture of the total effect of the Hawaii DSLL fishery on this species (our second scenario, which starts with an initial population of 1,292,023 individual oceanic whitetip sharks). Even when we ignored potential reproduction in our simulation, which would overestimate the potential effects of these mortalities on the population, the mortalities reduced the initial abundance by up to 0.57% over 10 years.

As noted earlier, we also understand that the fishery has been increasing the number of hooks in the water overtime, and yet we could not detect a meaningful relationship between fishing effort (e.g., number of hooks in the water) and oceanic whitetip shark incidental captures. This was not particularly surprising because our time series is rather short (17 samples) and we lacked a number of relevant covariates (variables other than number of hooks). In addition, the lack of statistical significance should not be interpreted to mean that an increasing number of hooks in the water is not meaningful either, in fact as hooks are increased it would seem reasonable that the probability of capture should go up. However, the relationship is multifaceted and we lack information to incorporate future growth in the fishery into our analysis. Our best estimates of the effect of the Hawaii DSLL fishery are based on historical data from the observer program.

Our assessment illustrates that the Hawaii DSLL fishery interacts with (captures) a very small portion of the Pacific population of oceanic whitetip sharks. At projected abundance levels, (1,292,023 individuals), capturing an average of 1,708 and killing 557 oceanic whitetip sharks from the Pacific Ocean population per year. Even when we treat the WCPO stock estimate as if it was a reasonable minimum estimate for the Pacific population, (with an initiation abundance of 775,000 individuals), and assume that the population is declining at 0.13% per year, the Hawaii

DSLL fishery only kills 0.09% annually. We consider the implications of this effect on OWT, as a species, in the Integration and Synthesis section of this opinion.

4.3.4 Green Sea Turtle

Exposure Analysis

Six DPSs of green sea turtles occur in the Action Area for this consultation. Although the nesting distribution of these DPSs is discrete, their distribution overlaps in the open ocean and in some coastal areas. As a result, genetic analyses from tissue samples collected from green turtles captured in the fishery are required to determine which DPS of green turtle an individual turtle represents and, in some cases, the genetic data does not allow us to assign a particular tissue sample to one green turtle DPS. As a result, assigning green turtles that are expected to be captured in the DSLL fishery to a particular DPS of green turtle requires several steps and the results are probabilistic. The exposure and response analyses that follow begin by treating green turtles as a group, then work through the process of identifying which of the 6 DPSs of green turtle are likely to be exposed to the fishery.

Green sea turtles have been captured in the DSLL fishery in 14 of the 19 years between 2004 and 2022. During this 19-year time interval fishery observers reported that 25 green sea turtles were captured in the DSLL fishery. When these reports are adjusted to account for the percentage of observer coverage, about 128 green sea turtles are likely to have been captured in the fishery in the 19-year period from 2004 to 2022 (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a). The value for 2022 was estimated based on the expansion factor with 1 observed capture and observer coverage of 20.22% (1/[20.22/100]=4.95 rounded to 5).

Based on genetic analyses of 14 tissue samples collected from green turtles captured in the DSLL fishery (Dutton, personal communication, 2018), the green turtles captured in the fishery represent the following DPSs of green turtle that have been listed as endangered or threatened: (1) Central North Pacific; (2) Central West Pacific; (3) Central South Pacific; (4) East Pacific; (5) East Indian-West Pacific; (6) Southwest Pacific green turtles. NMFS conducted mixed stock analyses of 14 genetic samples of green turtles captured in the DSLL fishery since 2002 (P. Dutton, PIFSC, Jun 29, 2018). These analyses produced the proportions and the associated 95% CI of each of the 6 green turtle DPSs that are likely to be exposed to the DSLL fishery. However, the proportions assigned to each green sea turtle DPS have a high level of uncertainty with large confidence intervals due to the low sample size.

Because the reliability of the genetic data is so low, we cannot be reasonably certain that all captures will not come from one green turtle DPS. Therefore, we focus our Risk Analysis on the possibility that all green sea turtle captures may come from any one greens turtle DPS by analyzing as if 100% of the captures comes from each affected DPS. While this is a conservative assumption, the best scientific and commercial data available do not allow us to make more reliable estimates. Furthermore, it is impossible to know the source DPS for any individual green sea turtle at time of capture and we are recognizing that any captured green sea turtle may be from any one of the six affected DPSs. Sizes of green sea turtles captured by the DSLL fishery ranged from 35 to 73 cm SCL (Figure 50). The minimum size of nesting green sea turtles in

Hawaii is 75 cm SCL (Balazs et al. 2015), therefore green sea turtles interacting with the fishery are primarily juveniles, although the largest captured green sea turtles (73 cm SCL) may be a subadult or adult; the next largest turtle was 68 cm SCL and likely a large juvenile.

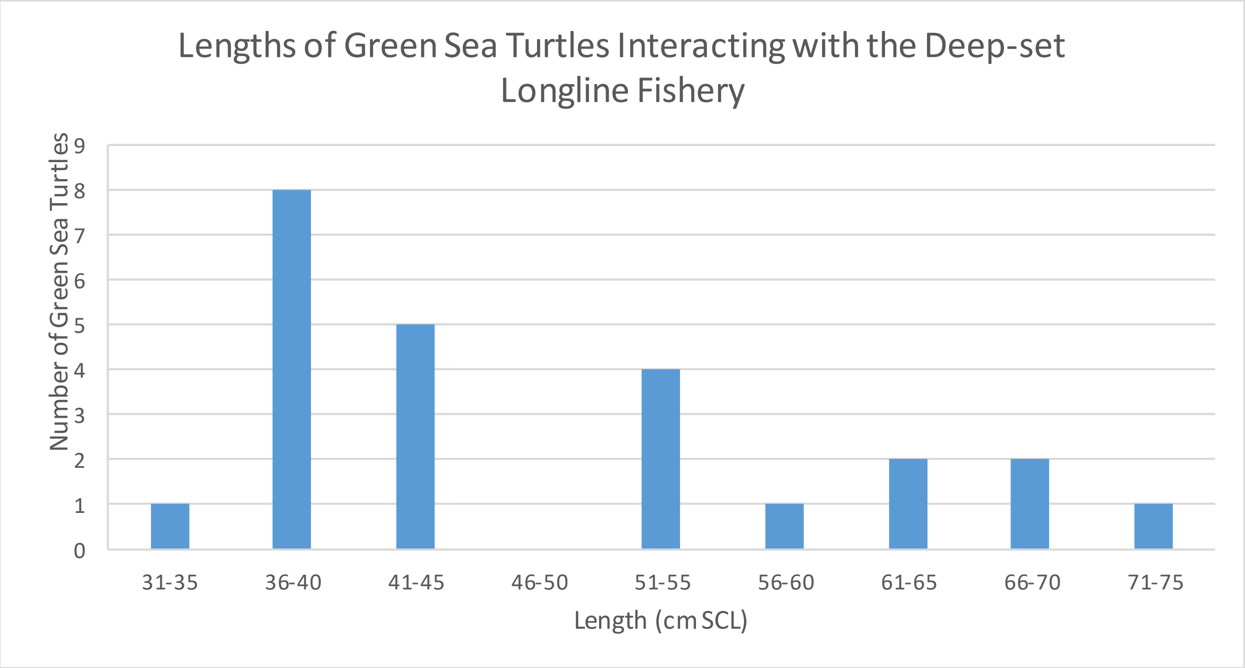

Figure 50. Lengths of observed green sea turtles interacting with the DSLL fishery from 2004 to 2022 (*n*= 24).

There was no spatial pattern to green turtle captures: green turtles were captured across the entire Action Area. Additionally, the data points were too sparse to allow sufficient density to create a heatmap including the addition of the 4 interactions in 2021 and 2022. Temporally, about 48% of all green sea turtles were captured in March, April, and May, with 24% captured in April alone (Figure 51). A secondary peak occurred in November and December.

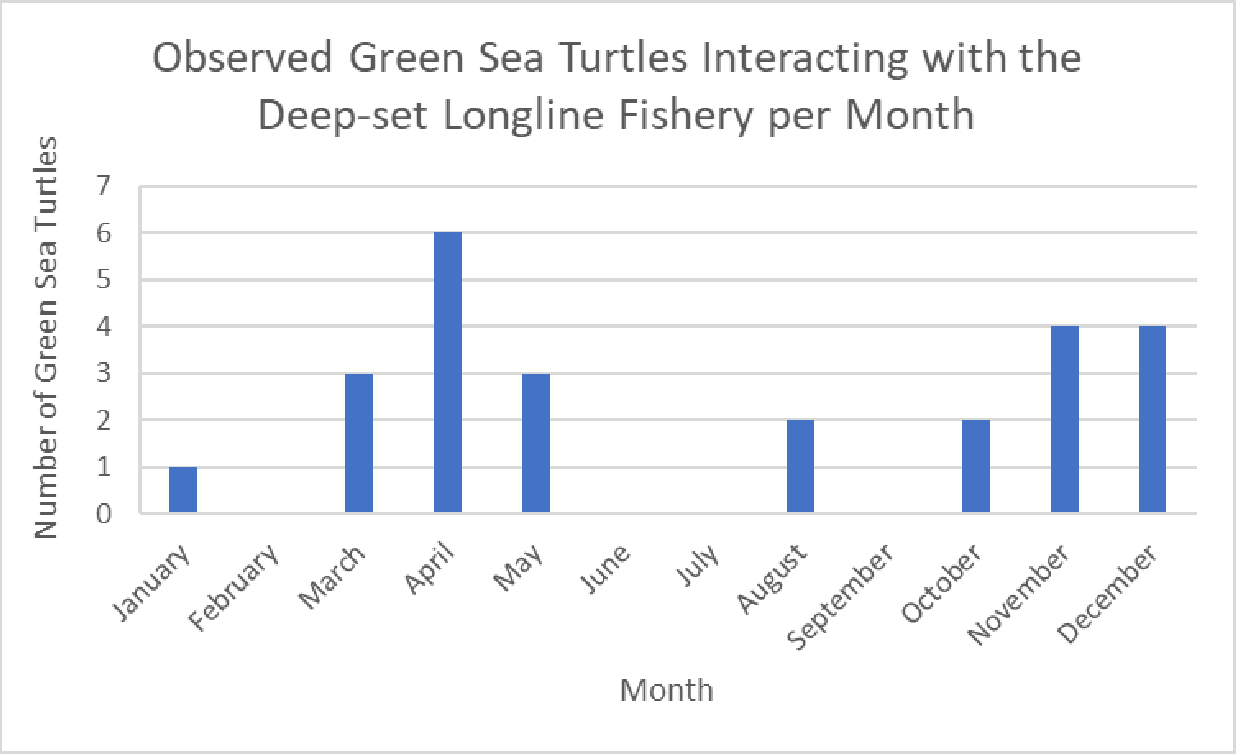

Figure 51. Numbers of observed green sea turtles captured per month in the DSLL fishery from 2004 to 2022 (*n*= 25).

Annual observed green sea turtle interactions were the highest in 2014, 2017, 2018, and 2021 with 3 observed in each of those years. Estimated interactions peaked in those same years with 16, 18, 17 and 17 estimated (McCracken 2019a) captures respectively (Figure 52).

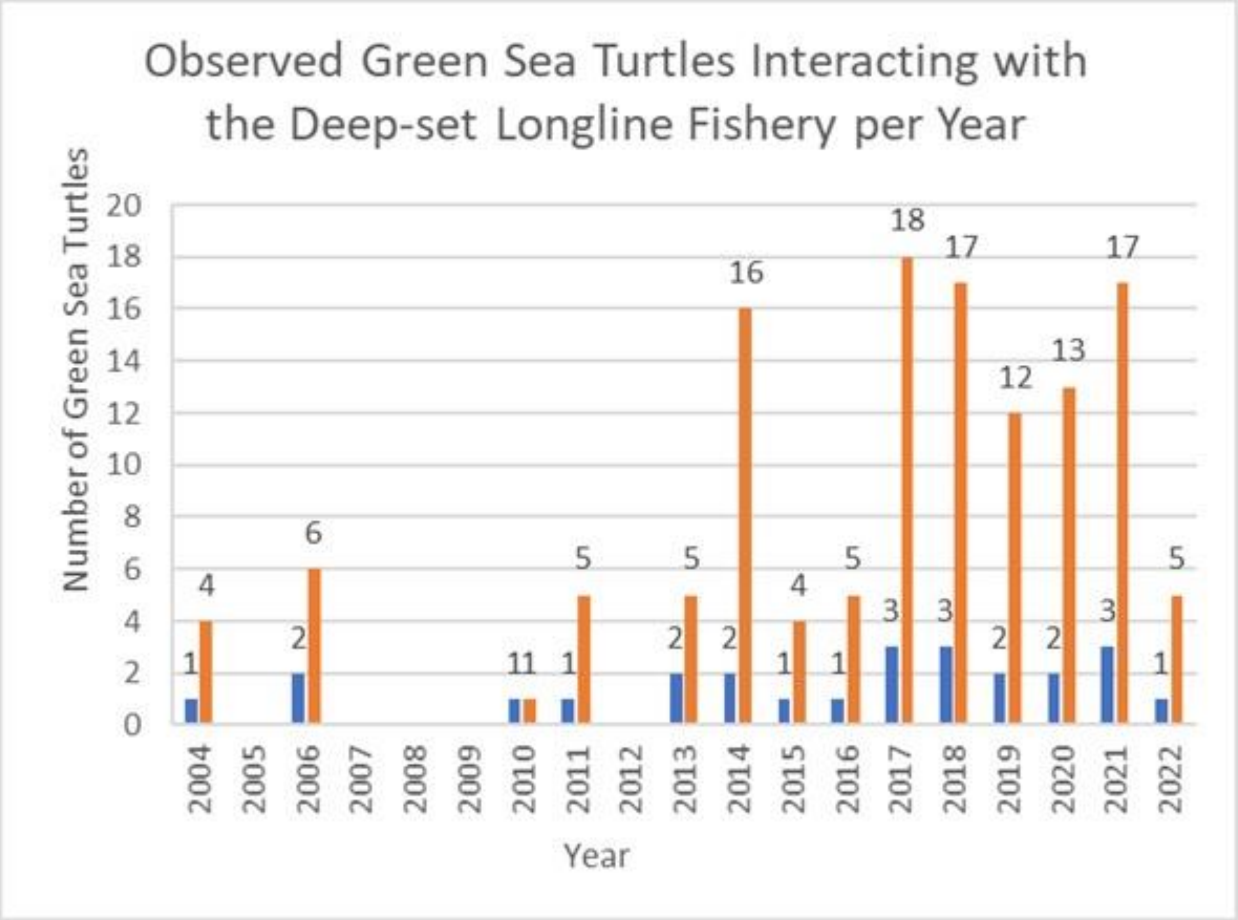

Figure 52. Green sea turtle interactions per year in the DSLL fishery between 2004 and 2022 for observed interactions (blue bars) estimated interactions (orange bars; McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a). Estimated interactions for 2022 were calculated using the expansion factor based on the observed captures and the observer coverage.

Estimates of the number of green turtles that are likely to be captured in the DSLL fishery each year are presented in Table 31 (estimates are from McCracken 2019a). We interpret the data in the table as follows: on average, 13 green turtles are likely to be captured each year in the DSLL fishery; however, about 5% of the time, 39 or more green turtles could be captured in any particular year. Because of the 20% observer coverage for this fishery, only a portion of these turtle captures would be reported by observers. We also report the maximum 5-yr running average and the maximum cumulative number of captures over 5 years (Table 31). There is no 95th percentile metric associated with the running average.

As noted in the introduction to Section 4.3, hooks per year in the DSLL have been increasing, specifically at a rate of ~1,802,000 hooks per year (95% CI: 1,557,699 to 2,046,303; Figure 35) and this rate of increase is expected to continue into the near future. We found a statistically significant relationship between observed hooks per year and observed green sea turtles captured per year with $R^2 = 0.23$ and $P = 0.02$ (Figure 53a). The Central North Pacific and East Pacific green sea turtle are likely the most common DPSs captured by the DSLL (NMFS unpublished data), especially given the proximity of their nesting beaches to the *Action Area*. Both of these DPSs have increasing population trends and the increasing trend in captures may be related to increasing densities of green sea turtles from both DPSs in the Action Area and not just increasing numbers of hooks set by the fishery. We examine this by looking at the trend in CPUE for observed green sea turtles where we find a similar increasing trend to our analysis of observed captures and observed hooks (Figure 53b; $R^2 = 0.18$; $P = 0.04$). This relationship is the number of captures per-1,000 hooks and is therefore not influenced by the increasing numbers of hooks in the fishery. This suggests the trend is due to either increasing abundances of green sea turtles or other environmental factors that are increasing the probability of capture.

Our CPUE regression indicates that we anticipate an additional 9.4×10^{-6} (or 0.0000094) observed green sea turtle captures per 1,000 hooks per year (CI: 4.3×10^{-7} to 1.8×10^{-5}). Given the anticipated increase in hooks per year for the DSLL (using the mean of 1,802,000 additional hooks per year), this equates to an additional 0.017 (CI: 0.015 to 0.019) observed green sea turtles per year from the anticipated annual increase in hooks. Over 10 years this would equate to an additional 0.17 (CI: 0.15 to 0.19) observed green sea turtle captures. Given all of the uncertainties associated with factors that may be driving the trend in CPUE for green sea turtles, and the low number of additional observed green sea turtles expected to be captured as a result of the fishery's potential increase in effort (less than 1 over 10 years), at this time we are not adjusting our anticipated future captures for the fishery based on the expected increase in hooks per year but acknowledge that captures will need to be closely monitored to assess if the fishery's impact to the population is greater than we have analyzed here.

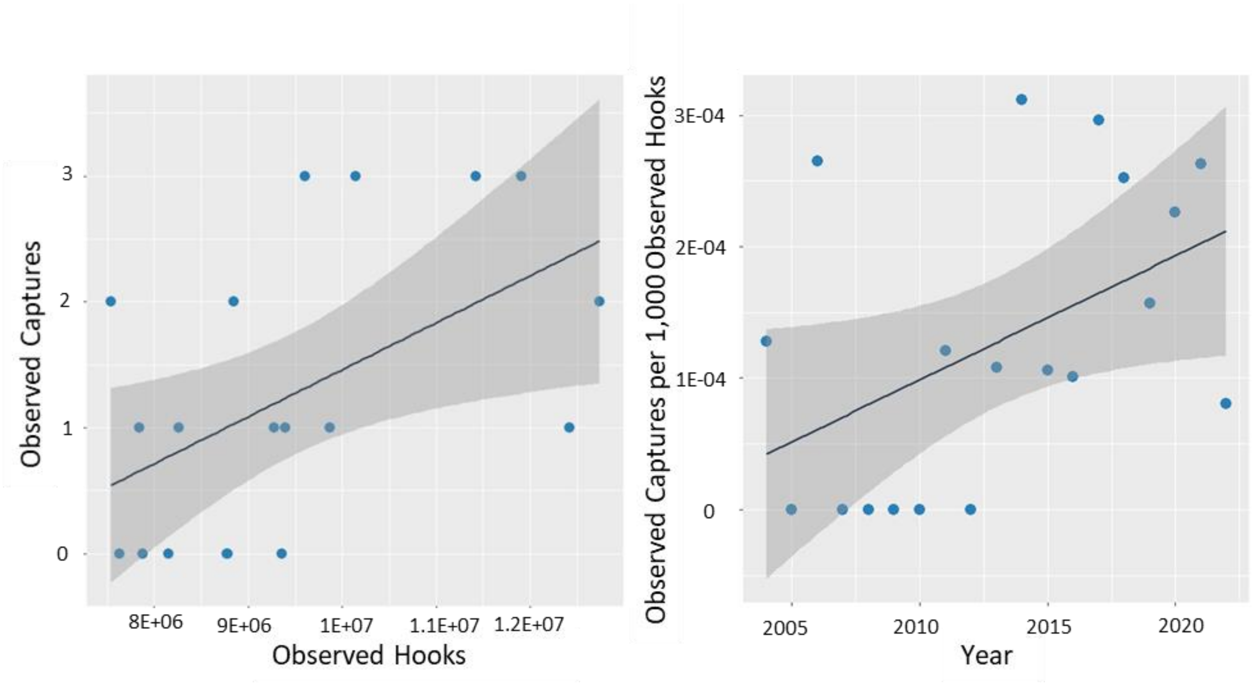

Figure 53. Relationships between observed hooks and observed green sea turtle captures in the DSLL. Left is the relationship between the number of observed green sea turtle captures per year and the number of observed hooks per year (blue dots). Right is the relationship between the annual number of green sea turtle captures per 1,000 hooks (CPUE) and year (blue dots). In both plots, the solid black line is a linear regression and the shaded area represents the 95% CI. See the text for the regression statistics.

Table 31. Number of green sea turtles expected to be incidentally captured in the DSLL per year (McCracken 2019a) and the maximum 5-yr running average and maximum 5-yr running sum calculated using estimated captures from 2004 to 2022 (data from McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a).

	DSLL Encounters	
Rate	**Mean**	**95th Percentile**
Annual	13	39
Max 5-Year Running Average	15.4	NA
5-Year Running Sum	77	NA

Response Analysis

If the patterns found in the observer data from 2004 to 2022 are representative, all the green turtles captured in the fishery would be hooked rather than entangled. About 96% (95% CI = 80 to 99%) of green turtles captured in the DSLL fishery have been dead when they were hauled back to the fishing vessel; this assessment assumes these percentages are applicable to future captures.

We used the post-interaction mortality rates of Ryder et al. (2006) for hard shell sea turtles to estimated post -interaction mortality of green sea turtles that were alive when they were captured. Only one green sea turtle was captured and released alive; this turtle was jaw hooked and released with no gear attached and using Ryder et al. (2006) was estimated as having a low chance of mortality as a result of the hooking (post-interaction mortality of 0.10). When at-vessel mortality and the post-release mortality of this one turtle are combined, the effective mortality rate for green sea turtles captured in this fishery is 96% (95% CI = 81 to 99%).

At this mortality rate, if 15.4 green turtles are captured in the fishery in any given year, we would expect an average of 14.8 of them to die as a result of being captured; using the 95th percentile value, if 39 of these turtles are captured, we would expect and average of 37 of them to die as a result of their capture (Table 32).

Table 32. Based on the anticipated number of captures from Table 31 and the mean mortality rate, these are the number of green sea turtles that are likely to die as a result of being captured in the DSLL each year.

	Anticipated mortalities based on the mean (96%) mortality rates	
Rate	**Mean**	**95th Percentile**
Annual	12	37
Max 5-Year Running Average	14.8	NA
5-Year Running Sum	74	NA

Risk Analyses

The analyses we presented thus far allowed us to estimate the total number of green turtles we expect to be captured and killed in the DSLL fishery. Because these six DPSs are listed separately and are genetically, ecologically, and demographically independent, the analyses that follow treat these six DPSs separately.

NMFS predicts that the DSLL fishery will interact with an annual mean of 13 (95th percentile: 39) green sea turtles over a one-year time period. The maximum 5-yr running average is 15.4 green sea turtles and we anticipate that up to 77 green sea turtles may be captured in any 5-year period. Some years will be higher or lower than the annual average, but the annual number of captures is unlikely to exceed the 95th percentile of 39 and the cumulative captures over any 5-year period is not expected to exceed 77 turtles. Over a 40 year time horizon, if the number of individual turtles captured remains consistent with historical numbers we would anticipate a cumulative total of 616 captures and we would anticipate that 591 of those turtles would die as a result of their interactions. As noted previously, because the genetic data do not allow us to reliably predict how many individuals will be captured from each DPS, we conduct our analysis as if all green turtle captures are from each of the affected DPS.

As discussed in the Status of the Listed Species for each green sea turtle DPS, we parameterized a post-breeding, age-based Leslie matrix model using the fecundity values in Balazs et al. (2015) and the survival rates and age at maturity of Piacenza et al. (2016); and conducted 10,000 simulations drawing from triangular distributions for the survival rates and age to maturity ranges specified in Table 4. All other parameters were held stable as specified in Table 4 across the simulations. Based on this model we were able to estimate the abundance for all green sea turtle DPSs.

Central North Pacific

As discussed in the Status of Listed Species, the abundance of Central North Pacific green turtles had increased by about 5.4% per year over the past 40 years (Balazs et al. 2015). Between 1973 and 2015, nesting activity has ranged from a low of 67 in 1973 to a high of 808 nesting females in 2011 at East Island (NMFS unpublished data). These increases have been attributed to increased survivorship (since harvesting of turtles in foraging areas was prohibited in the mid-1970s) and cessation of habitat damage at the French Frigate Shoals rookeries beginning in the early 1950s (Balazs and Chaloupka 2004). Nevertheless, more than 96% of nesting of this DPS occurred in French Frigate Shoals and half of that nesting occurred on East Island. In 2018, Hurricane Walaka demolished East Island and placed Central North Pacific green turtles in a natural experiment whose outcome remains uncertain, although in 2019, 251 females were identified at Tern Island with a total of 583 green turtles (males and females) tagged throughout French Frigate Shoals (https://www.fisheries.noaa.gov/feature-story/researching-and-rescuing-hawaiian-monk-seals-and-green-sea-turtles). Final results for 2019 indicate that 263 females nested at Tern Island, and 645 females nested there in 2021 (there are no data for 2020 due to COVID 19; NMFS unpublished data).

While the 645 nesting females in 2021 does not exceed the high of 808 in 2011, it does provide evidence that the Central North Pacific green sea turtle is successfully using alternate nesting habitat following the loss of East Islands. It is likely that some females are using additional nesting habitat at French Frigate Shoals that are not currently monitored so the estimated 645 nesting females at Tern may be a minimum estimated for 2021 nesting. Tiwari et al. (2010) suggested that other nesting beaches that could support substantially more nesting activity are available to Central North Pacific green turtles within their current distribution, but no one knows if these green turtles will distribute to those beaches, successfully produce nests on those beaches, and do so in numbers that offset the loss of nesting activity at East Island. More importantly, no one knows how many years this process might require.

As described in the Exposure Analysis, all individuals that were measured would be considered juveniles with one possible subadult. Assuming total abundance for Central North Pacific green sea turtles is 682,296 and assuming a mean of 13 (95th percentile: 39) individuals from this DPS may interact with the DSLL fishery, this would equal 0.0019% (95th percentile: 0.0057%) of the DPS total population. The mortality of 12 (95th percentile: 37) individual Central North Pacific green sea turtles assuming total abundance equals 682,296 represents 0.0018% (95th percentile: 0.0054%) of the total population. For the maximum 5-year running average, 15.4 captures and

14.8 mortalities represents 0.0023% of the DPS' population captured and 0.0022% of the population killed. The maximum 5-year running sum of 15.4 Central North Pacific green sea turtles suggests that over 10 years we would anticipate 154 total captures, of these, 148 would be expected to die. This level of interaction would represent 0.023% of the population being impacted and 0.022% of the population being killed. As this population is increasing in size (Balazs et al. 2015; Seminoff et al. 2015), these values will decrease at 10, 20, 30, and 40 years into the future as we anticipate higher population sizes at those time points. Over a 40 year time horizon, if captures remain consistent with historical numbers we would anticipate 616 interactions and of those 592 would be expected to die.

Our assessment illustrates that the DSLL fishery interacts with (incidentally captures) a very small portion of the Central North Pacific green sea turtle. At projected abundance levels (682,296 individuals), incidentally capturing and killing an average of three Central North Pacific green sea turtle per year constitutes the death of 0.00044% of the population. We consider the implications of this effect on Central North Pacific green sea turtles in the Integration and Synthesis section of this opinion.

Central South Pacific

As noted in the Status of Listed Resources section, there is evidence that this DPS is declining but no data to quantify the rate of decline (Seminoff et al. 2015). We estimate the total population size in 2015 to be 631,745 individuals greater than one year old. In the Environmental Baseline section we present CPUE trend data for green sea turtles captured in the WCPO longline fisheries (Figure 30). Based on this analysis the CPUE of green sea turtles is declining at a rate of 5.8% (95% CI: -7.6% to -3.9%) per year. In the absence of specific trend data for the Central South Pacific green sea turtle, we use this rate of decline as an estimate for the population trend. Therefore we estimate the population size in 2022 to be 420,939 individuals greater than one year old.

As described in the Exposure Analysis, all observed green sea turtles that were measured between 2004 and 2022 are considered juveniles with one possible subadult. Assuming total abundance for the Central South Pacific green sea turtle in 2022 is 420,939 individuals and assuming a mean of 13 (95th percentile: 39) individuals from this DPS may interact with the DSLL fishery, this would equal 0.0031% (95th percentile: 0.0093%) of the DPS' total population. The mortality of 12 (95th percentile: 37) individual Central South Pacific green sea turtles assuming total abundance equals 420,939 represents 0.0029% (95th percentile: 0.0088%) of the total population. For the maximum 5-year running average, 15.4 captures and 14.8 mortalities represents 0.0037% of the DPS' population captured and 0.0035% of the population killed. The maximum 5-year running sum of 15.4 Central South Pacific green sea turtles suggests that over 10 years we would anticipate 154 total captures, of these, 148 would be expected to die. This level of interaction would represent 0.037% of the population being impacted and 0.035% of the population being killed.

As the DPS declines, the population size in 2032 would be 235,682; 131,959 in 2042, 73,884 in 2052, and 41,369 by 2062. When we consider 154 interactions over 10 years (extrapolated from the maximum 5-yr running average of 15.4 per year), the mortality resulting from interactions with the fishery would be 0.063% of the total population size estimated for 2032. Between 2032 and 2042, if the number of expected interactions stay the same, the fishery would interact with 0.11% of the total population size estimated in 2042, 0.20% of the Central South Pacific green sea turtle estimated in 2052, and 0.36% of the population in 2062 based on a 5.8% per year decline.

Our assessment illustrates that the DSLL fishery interacts with (incidentally captures) a very small portion of the Central South Pacific green sea turtle. At the 2022 projected abundance levels (420,939 individuals), incidentally capturing and killing an average of 15.4 Central South Pacific green sea turtles per year constitutes the death of 0.0035% of the population. If captures remain the same as the population declines the fishery would incidentally capture and kill 0.036% of the population in 40 years which we consider to be the most conservative reasonable scenario as capture rates are likely to decline if the population declines by 90% over this time period as predicted from a 5.8% per year decline. Without the mortalities caused by the DSLL, from 2022 to 2023 the population's rate of decline would decrease to 5.796% per year; from 2022 to 2062 the population's rate of decline would decrease to 5.786%. Given the estimated standard deviation for the population growth rate (0.018), and based on a power analysis for detecting differences in means with the same variance (Cohen 1988), more than 200,000 years of data would be required to detect a statistically significant difference in these values (see Appendix C for details on this analysis).We consider the implications of this effect on Central South Pacific green sea turtles in the Integration and Synthesis section of this opinion.

Central West Pacific

As noted in the Status of Listed Resources section, there is evidence that this DPS is declining but no data to quantify the rate of decline (Seminoff et al. 2015). We estimate the total population size in 2015 to be 1,543,625 individuals greater than one year old. In the Environmental Baseline section we present CPUE trend data for green sea turtles captured in the WCPO longline fisheries (Figure 30). Based on this analysis the CPUE of green sea turtles is declining at a rate of 5.8% per year (95% CI: -7.6% to -3.9%). In the absence of specific trend data for the Central West Pacific green sea turtle, we use this rate of decline as an estimate for the population trend. Therefore we estimate the population size in 2022 to be 1,028,533 individuals greater than one year old.

As described in the Exposure Analysis, all observed green sea turtles that were measured between 2004 and 2022 are considered juveniles with one possible subadult based on carapace lengths. Assuming total abundance for Central West Pacific green sea turtles is 1,028,533 individuals and assuming a mean of 13 (95th percentile: 39) individuals from this DPS may interact with the DSLL fishery, this would equal 0.0013% (95th percentile: 0.0038%) of the DPS' total population. The mortality of 12 (95th percentile: 37) individual Central West Pacific green sea turtles assuming total abundance equals 1,028,533 represents 0.0012% (95th percentile: 0.0036%) of the total population. For the maximum 5-year running average, 15.4 captures and 14.8 mortalities represents 0.0015% of the DPS' population captured and 0.0014% of the population killed. The maximum 5-year running sum of 15.4 Central West Pacific green sea turtles suggests that over 10 years we would anticipate 154 total captures, of these, 148 would be expected to die. This level of interaction would represent 0.015% of the population being impacted and 0.014% of the population killed.

As the DPS declines, the population size in 2032 would be 575,874; 322,430 in 2042; 180,529 in 2052, and 101,078 by 2062. When we consider 154 interactions and 148 mortalities over 10 years (extrapolated from the maximum 5-yr running average of 15.4 per year), the mortality resulting from interactions with the fishery would be 0.026% of the total population size estimated for 2032. Between 2032 and 2042 if the number of expected interactions stay the same, the fishery would interact with 0.046% of the total population size estimated in 2042, 0.085% of the Central West Pacific green sea turtle estimated in 2052, and 0.15% of the population in 2062 based on a 5.8% per year decline.

Our assessment illustrates that the DSLL fishery interacts with (incidentally captures) a very small portion of the Central West Pacific green sea turtle. At the 2022 projected abundance levels (1,028,533 individuals), incidentally capturing and killing an average of 15.4 Central West Pacific green sea turtles per year constitutes death of 0.0014% of the population. If captures remain the same as the population declines the fishery would incidentally kill 0.015% of the population in 40 years which we consider to be the most conservative reasonable scenario as capture rates are likely to decline if the population declines by 90% as predicted from a 5.8% per year decline. Without the mortalities caused by the DSLL, from 2022 to 2023 the population's rate of decline would decrease to 5.7985% per year; from 2022 to 2062 the population's rate of decline would decrease to 5.794% per year. Given the estimated standard deviation for the population growth rate (0.018), and based on a power analysis for detecting differences in means with the same variance (Cohen 1988), more than one million years of data would be required to detect a statistically significant difference in these values (see Appendix C for details on this analysis). We consider the implications of this effect on Central West Pacific green sea turtles in the Integration and Synthesis section of this opinion.

East Indian-West Pacific

As noted in the Status of Listed Resources section, we estimate the total population size in 2015 was 18,171,565 individuals greater than one year old and that the population is declining at a rate of 1.11% per year (range -3.6% to +6.7). Therefore the estimated population size in 2022 is 16,813,093. As described in the Exposure Analysis, all observed green sea turtles that were measured between 2004 and 2022 are considered juveniles with one possible subadult. Assuming total abundance for East Indian-West Pacific green sea turtles is 16,813,093 individuals and assuming a mean of 13 (95th percentile: 39) individuals from this DPS may interact with the DSLL fishery, this would equal 0.000077% (95th percentile: 0.00023%) of the DPS' total population. The mortality of 12 (95th percentile: 37) individual East Indian-West Pacific green sea turtles assuming total abundance equals 1,028,533 represents 0.000071% (95th percentile: 0.00022%) of the total population. For the maximum 5-year running average, 15.4 captures and 14.8 mortalities represents 0.000092% of the DPS' population captured and 0.000088% of the population killed. The maximum 5-year running sum of 15.4 East Indian-West Pacific green sea turtles suggests that over 10 years we would anticipate 154 total captures, of which, 148 would be expected to die. This level of interaction would represent 0.00092% of the population being impacted and 0.00088% being killed.

As the DPS declines, the population size in 2032 would be 15,046,688; 13,465,864 in 2042; 12,051,123 in 2052, and 10,785,016 by 2062. When we consider 154 interactions and 148 mortalities over 10 years (extrapolated from the maximum 5-yr running average of 15.4 per year), the rate of mortality with the fishery would be 0.00098% of the total population size estimated for 2032. Between 2032 and 2042 if the number of expected interactions stay the same, the fishery would kill 0.0011% of the total population size estimated in 2042, 0.0012% of the East Indian-West Pacific green sea turtle estimated in 2052, and 0.0014% of the population in 2062 based on a 1.11% per year decline.

Our assessment illustrates that the DSLL fishery interacts with (incidentally captures) a very small portion of the East Indian-West Pacific green sea turtle. At the 2022 projected abundance levels (16,813,093 individuals), incidentally capturing 15.4 and killing an average of 14.8 East Indian-West Pacific green sea turtles per year constitutes death of 0.000088% of the population. If captures remain the same as the population declines the fishery would incidentally capture and kill 0.00014% of the population in 40 years which we consider to be the most conservative reasonable scenario as capture rates are likely to decline if the population declines by 36% as predicted from a 1.11% per year decline. Without the mortalities caused by the DSLL, from 2022 to 2023 the population's rate of decline decreases to 1.10991% per year; from 2022 to 2062 the population's rate of decline decreases to 1.10989% per year decline. Given the estimated standard deviation for the population growth rate (0.028), and based on a power analysis for detecting differences in means with the same variance (Cohen 1988), more than one million years of data would be required to detect a statistically significant difference in these values (see Appendix C for details on this analysis). We consider the implications of this effect on East Indian-West Pacific green sea turtles in the Integration and Synthesis section of this opinion.

East Pacific

As noted in the Status of Listed Resources section, we estimate that there were 3,580,207 East Pacific green sea turtles greater than one year old in 2015 and that the trend for this DPS is likely increasing. As described in the Exposure Analysis, all individuals observed green sea turtles that were measured between 2004 and 2022 would be considered juveniles with one possible subadult. Assuming total abundance for East Pacific green sea turtles is 3,580,207 individuals and assuming a mean of 13 (95th percentile: 39) individuals from this DPS may interact with the DSLL fishery, this would equal 0.00036% (95th percentile: 0.0011%) of the DPS' total population. The mortality of 12 (95th percentile: 37) individual East Pacific green sea turtles assuming total abundance equals 1,028,533 represents 0.00034% (95th percentile: 0.0010%) of the total population. For the maximum 5-year running average, 15.4 captures and 14.8 mortalities represents 0.00043% of the DPS' population captured and 0.00041% of the population killed.

The maximum 5-year running average of 15.4 East Pacific green sea turtles suggests that over 10 years we would anticipate 154 total captures, of these, 148 would be expected to die. This level of interaction would represent 0.00043% of the population being impacted and 0.00042% of the population being killed. As this population is increasing in size (Seminoff et al. 2015), these values will decrease at 10, 20, 30, and 40 years into the future as we anticipate higher population sizes at those time points.

Our assessment illustrates that the DSLL fishery interacts with (incidentally captures) a very small portion of the East Pacific green sea turtle. At projected abundance levels (3,580,207 individuals), incidentally capturing 15.4 and killing an average of 14.8 East Pacific green sea turtles per year constitutes the death of 0.00041% of the population. We consider the implications of this effect on East Pacific green sea turtles in the Integration and Synthesis section of this opinion.

Southwest Pacific

As noted in the Status of Listed Resources section, we estimate that there were 14,762,154 Southwest Pacific green sea turtles greater than one year old in 2015 and that the trend for this DPS is likely increasing. As described in the Exposure Analysis, all individuals that were measured would be considered juveniles with one possible subadult. Assuming total abundance for Southwest Pacific green sea turtles is 14,762,154 individuals and assuming a mean of 13 (95th percentile: 39) individuals from this DPS may interact with the DSLL fishery, this would equal 0.000088% (95th percentile: 0.00026%) of the DPS total population. The mortality of 12 (95th percentile: 37) individual Southwest Pacific green sea turtles assuming total abundance equals 1,028,533 represents 0.000081% (95th percentile: 0.00025%) of the total population. For the maximum 5-year running average, 15.4 captures and 14.8 mortalities represents 0.00010% of the DPS' population captured and 0.00010% of the population killed. The maximum 5-year running sum of 15.4 Southwest Pacific green sea turtles suggests that over 10 years we would anticipate 154 total captures, of these, 148 would be expected to die. This level of interaction would represent 0.0010% of the population being impacted and 0.0010% being killed. As this population is increasing in size (Seminoff et al. 2015), these values will decrease at 10, 20, 30, and 40 years into the future as we anticipate higher population sizes at those time points.

Our assessment illustrates that the DSLL fishery interacts with (incidentally captures) a very small portion of the Southwest Pacific green sea turtle. At projected abundance levels (14,762,154 individuals), incidentally capturing 15.4 and killing an average of 14.8 Southwest Pacific green sea turtle per year constitutes the death of 0.00010% of the population. We consider the implications of this effect on Southwest Pacific green sea turtles in the Integration and Synthesis section of this opinion.

4.3.5 Leatherback Sea Turtle

Exposure Analysis

Leatherback sea turtles have been captured in the DSLL fishery in 18 of the 19 years between 2004 and 2022. During this 19-year time interval fishery observers reported that 46 leatherback sea turtles were captured in the DSLL fishery. When these reports are adjusted to account for the percentage of observer coverage, about 246 leatherback sea turtles are likely to have been captured in the fishery in the 19-year period from 2004 to 2022 (McCracken 2019a; McCracken and Cooper 2020a, 2021a, 2022a). The value for 2022 was estimated based on the expansion factor with 5 observed captures and observer coverage of 20.22% (5/[20.22/100]=24.7 rounded to 25).

Based on mitochondrial and nuclear DNA analyses of tissue samples collected from 16 leatherback turtles captured in the DSLL fishery, 93.75% of the leatherback turtles were from the Western Pacific population that occurs in Papua-Barat (Indonesia), Papua New Guinea, and

Solomon Islands (Dutton pers. comm. 12 April 2018). The remaining 6.25% of the leatherback turtles (or one of the 16) were from the Eastern Pacific population.

Of the 46 observed leatherback sea turtle interactions, 38 had carapace lengths that were either directly measured if the turtle was landed or estimated by the observer if the turtle was not landed. Lengths of leatherback sea turtles larger than 122 cm SCL were estimated by the observer looking over the rail and down at the turtle in the water and therefore have a high degree of uncertainty. These values are generally reported in feet, and the most common length reported was 4 ft. which converts to 122 cm SCL; 40% or 10 of the 25 turtles with estimated lengths of 4 ft. or greater were reported as 4 ft. (Figure 55). The smallest reported nesting adult female in the West Pacific population is 138.5 cm CCL (Lontoh 2014) or approximately 131 cm SCL based on the relationship between SCL and CCL in Tucker and Frazer (1991); 131 cm SCL converts to 4.3 ft. Given the uncertainty associated with the methods used to estimate the size of unlanded leatherback sea turtles, it is possible that a proportion of leatherback sea turtles estimated as 4-ft. are adults. To estimate this proportion, we first assume 4 ft. is the mean of the distribution of the true lengths and therefore 50% of leatherbacks estimated at 4 ft. have their size overestimated (i.e. they are smaller than 4 ft.) and we consider them juveniles. We further assume that the other 50% of 4 ft. captures are between 4 and 4.5 ft, or 122 cm SCL to 137 cm SCL, in other words, actual measurements of leatherbacks estimated as 4 ft. range from 3.5 to 4.5 ft. Siders et al. (2023) used a logistic function to assign probabilities of being mature to smaller turtles (Figure 54).

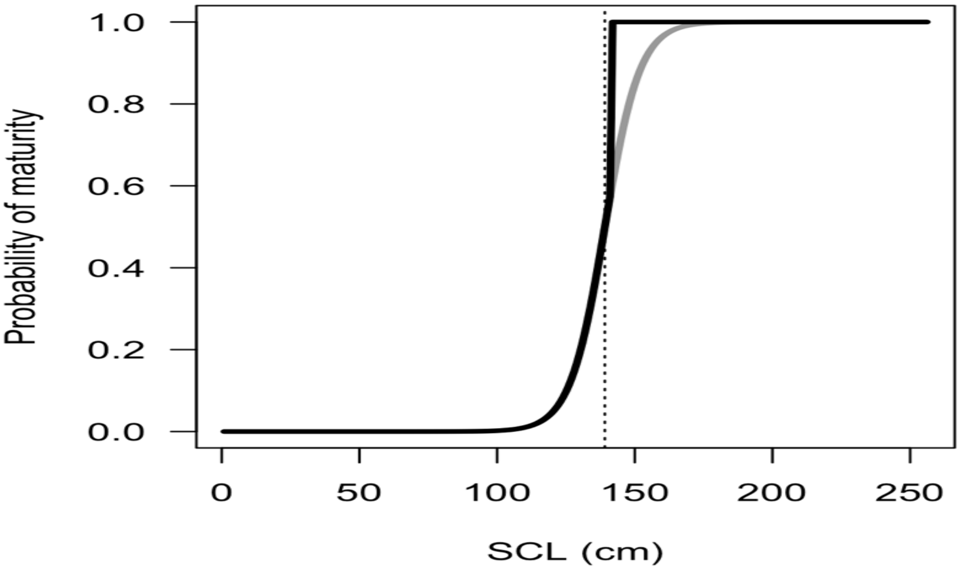

Figure 54. Relationship used by Siders et al. (2023) to assign the probability of being mature to leatherback sea turtles greater than 110 cm SCL.

Based on this relationship, approximately 5% of 122 cm SCL leatherbacks are mature while 35% of 137 cm SCL leatherbacks are mature. Therefore, we estimate that between 2.5% (50%*5% or

50*5/100) and 17.5% (50%*35% or 50*35/100) of leatherbacks estimated as 4 ft. are mature. Given the multiple assumptions necessary for this analysis and the uncertainty they introduce, we rely on the higher value of 17.5% of turtles estimated as 4 ft. being mature. For example, if we had assumed that the range of actual measurements was from 3 ft. to 5 ft. for turtles estimated at 4 ft. (rather than 3.5 ft. to 4.5 ft.), the probability of a 4 ft. turtle being mature would range from 2.5% to 50% with a midpoint of 26.25%. We think the range of 3 ft. to 5 ft. is too large an error for trained observers and that 26.25% would likely overestimate the proportion of 4 ft. turtles that are adult. We therefore consider the upper range of our original estimate (17.5%) to be a reasonable estimate that is unlikely to underestimate the proportion of 4 ft. turtles that are adult but at the same time not inflate the number.

Of the turtles with measurements, 34% ([13/38]*100; 95% CI = 21 to 50%) were juveniles less than 115 cm SCL, 39% ([15/38]*100; 95% CI = 26 to 55%) were adults with estimated carapace length greater than 145 cm SCL, and 26% ([10/38]*100; 95% CI = 15 to 42%) had estimated lengths of 4-ft. Using the estimates above for the proportion of 4 ft. turtles that may be adults, we estimate that 44% (39%+[26%*17.5%]) of leatherbacks captured in the DSLL are adults.

We note that the proportion of adults captured by the fishery is markedly different from our previous biological opinion (NMFS 2014) which stated that the fishery interacted predominantly with juveniles. As noted above, for large leatherback sea turtles that are not landed, lengths are generally estimated by the observer, and these lengths are not recorded in the field typically used for recording lengths on the datasheet, but rather are reported within the comment fields. During this consultation, we closely examined all the available data on the size of leatherback sea turtles captured in this fishery, including all observer notes, to better estimate the effect of the action on leatherback turtles by age classes. Once we examined the length data from these comment fields, we realized that the fishery observed interactions included records for seven adult sized leatherback sea turtles that were captured between 2004 and 2014 with an additional six individuals recorded as 4 ft. in length. From 2015 to 2021, an additional eight adult-sized and four 4 ft. leatherback sea turtles were captured from 2015 to 2022. In total, 15 adult-sized leatherback sea turtles and 10 individuals recorded as 4 ft. have been observed interacting with this fishery since 2004.

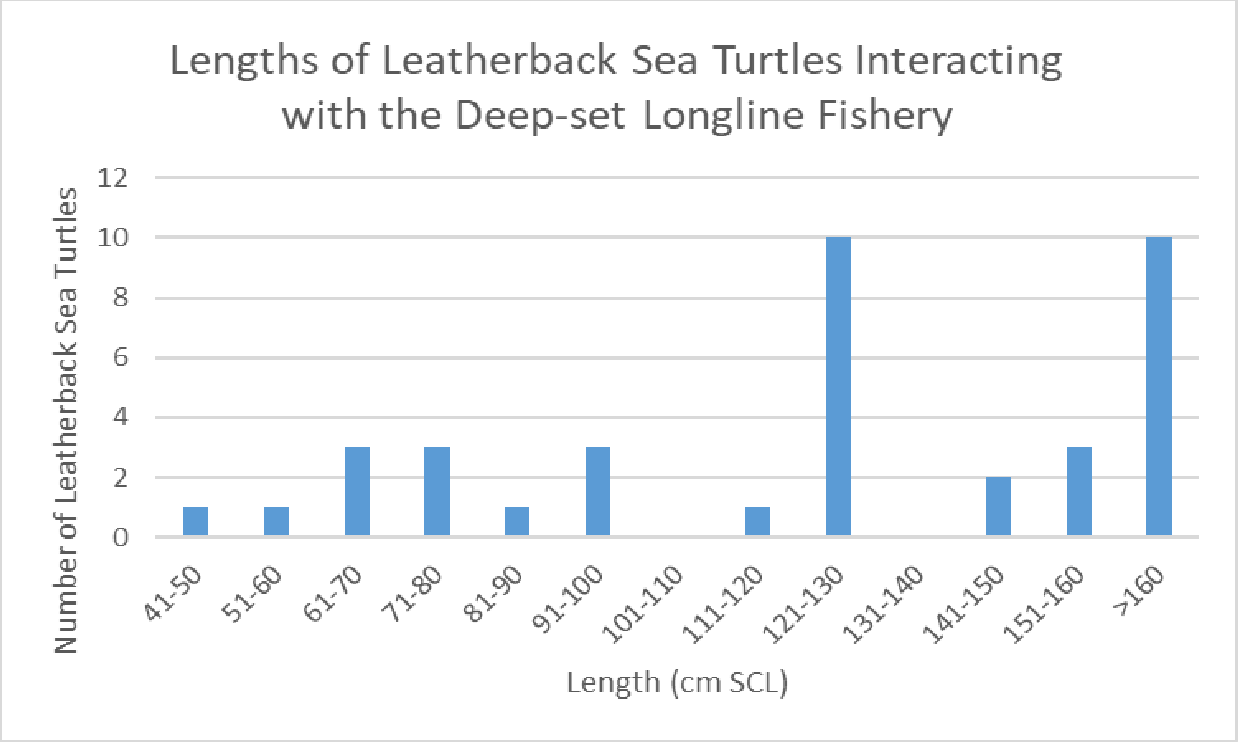

Figure 55. Lengths of observed leatherback sea turtles interacting with the DSLL fishery from 2004 to 2022. The size bin of 121 – 130 cm SCL were all estimated at 4 ft. by observers which converts to approximately 122 cm. Note that all lengths greater than 121 cm were estimated by observers in feet and converted to cm for this figure.

We examined the probability of leatherback sea turtles being captured on shallow hooks, defined as the first three hooks on either side of the catenary curve versus deep hooks (defined as the hooks between the shallow hooks). For observer records that reported both the hook number on which the turtle was caught and the number of hooks per float, we determined whether the hook on which the animal was caught was shallow or deep as defined above. For leatherback sea turtles, 35 captures between 2004 and 2022 contained the hook information required for the analysis, and of those 15 were captured on shallow hooks for a mean of 42.9% (95% CI: 28.0% to 59.2%).

Leatherback turtles were captured throughout the Action Area with about 74% of captures between 160° and 170° West longitude and south of the MHI (Figure 56). Figure 57 compares the heatmaps of observed leatherback sea turtle interactions (left; *n* = 40 versus aggregated fishing effort during this same time frame (right; *n* = 65,245 from 2004 to 2020) to better illustrate the locations where this species is caught in the fishery.

Twenty-three of the 46 (or 50%) of observed leatherback captures occurred south of 15°N while 10.4% of the fishery effort from 2004 to 2022 occurred south of 15°N. Of note, of the five observed leatherback turtles in 2022, two of them were captured south of 15°N while the other three were captured between 15°N and 17°N.

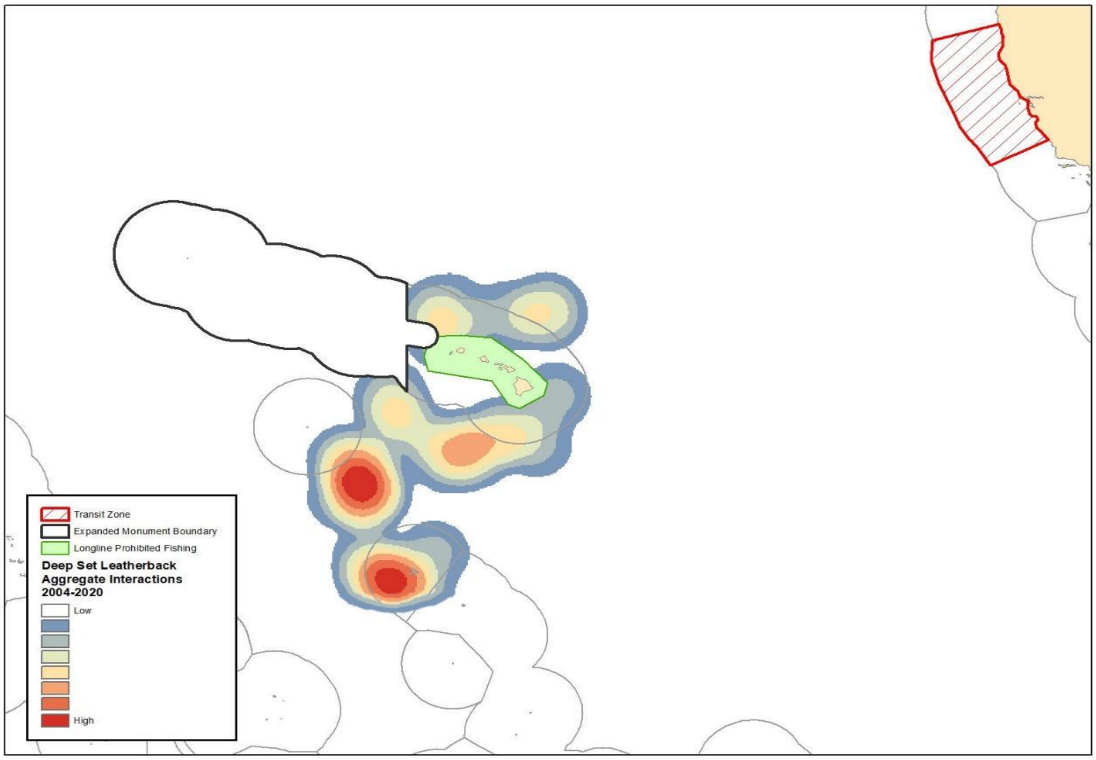

Figure 56. Geographical representation of observed aggregate leatherback sea turtle interactions with the DSLL fishery between 2004 and 2020 (*n* = 40).

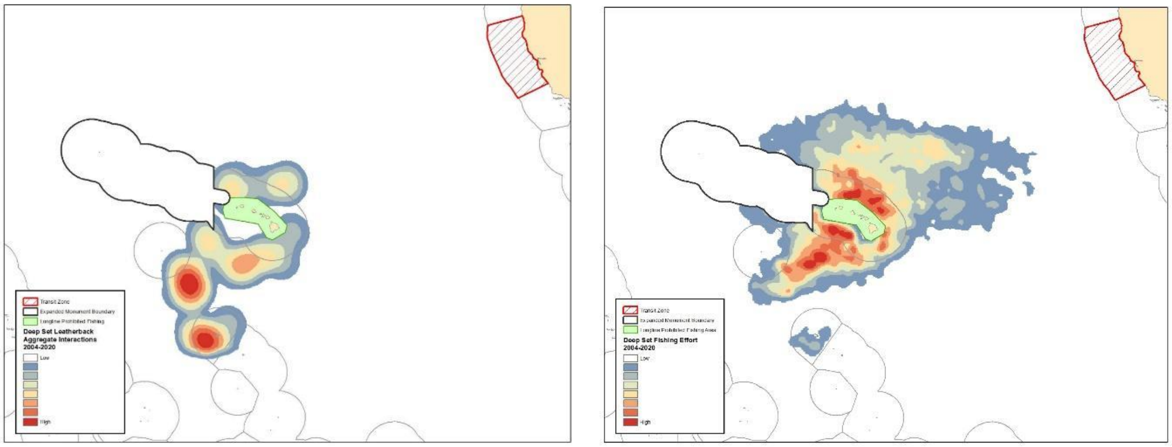

Figure 57. Heatmap comparison of observed leatherback sea turtle interactions (left; *n* = 40 versus aggregated observed fishing effort during this same time frame (right; *n*= 65,245 from 2004 to 2020).

Most leatherback turtles are caught in the first four months of the year. About 25% of all leatherback turtles captured from 2004 to 2020 were captured in April with smaller peaks in

January, July, and November. Combined captures from January through April represented about 63% of leatherback turtle captures (Figure 58). Of note, of all sets that occurred south of 15°N, 20% of them occur in April and 58% of them occur from March to May. Of the 11 observed leatherback interactions that occurred in April, nine of them, or 82% occurred south of 15°N.

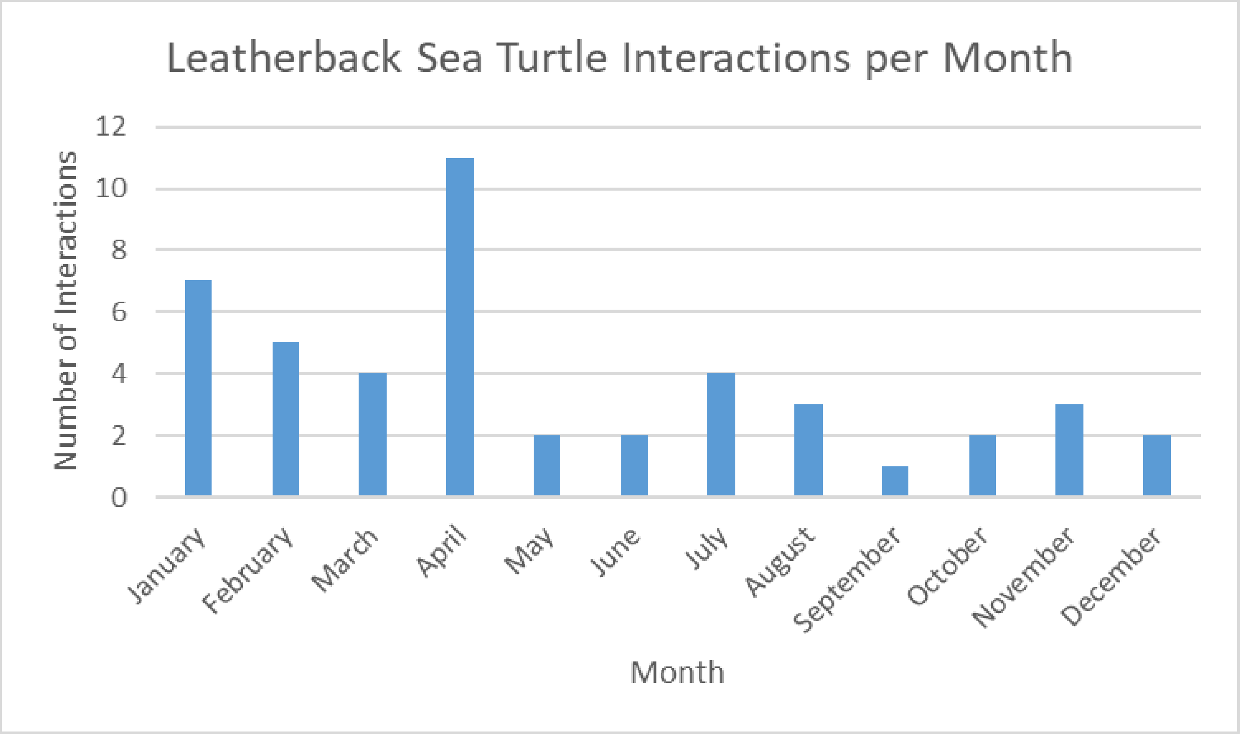

Figure 58. Numbers of observed leatherback sea turtles captured per month in the DSLL fishery from 2004 to 2022.

Annual observed and estimated leatherback sea turtle interactions peaked in 2014 with 7 observed and 38 estimated captures (Figure 59; McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a).

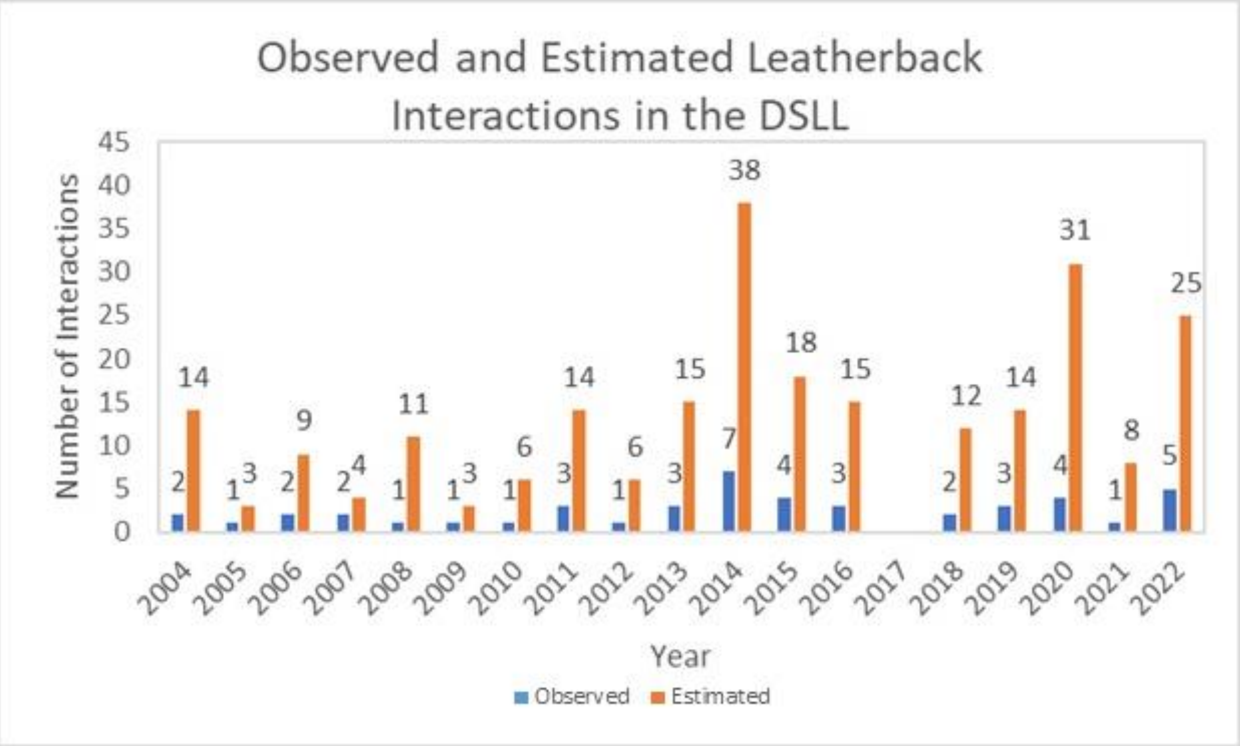

Figure 59. Leatherback sea turtle interactions per year in the DSLL fishery between 2004 and 2022 for observed interactions (blue bars) and estimated interactions (orange bars; McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a). Estimated interactions for 2022 were calculated using the expansion factor based on the observed captures and the observer coverage.

Estimates of the number of leatherback turtles that are likely to be captured in the DSLL fishery are presented in Table 33 (estimates are from McCracken 2019a). On average, we would expect a mean of 17 (95th Percentile: 43) leatherback turtles to be captured each year in the DSLL fishery, although the number of leatherback turtles captured in any given year could be as high as 43. Only a portion of these turtle captures would be reported by observers because the DSLL fishery has only 20% observer coverage. We also report the maximum 5-yr running average and the maximum cumulative number of captures over 5 years (Table 33). There is no 95th percentile metric associated with the running average.

Table 33. The estimated number of leatherback sea turtles that are likely to be captured by the DSLL per year (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021) and the maximum 5-yr running average and maximum running sum calculated using observed and estimated captures from 2004 to 2022.

	DSLL Encounters	
Rate	**Mean**	**95th Percentile**
Annual	17	43
Maximum 5-Yr Running Annual Average	18.4	N/A
Maximum 5-Yr Running Sum	92	N/A

Response Analysis

If the patterns found in the observer data from 2004 to 2022 are representative, the majority of leatherback turtles would be hooked rather than entangled. Over this time period, an average of about 87% (95% CI = 74 to 94%) of captured leatherbacks were hooked, and three of those that were hooked were also entangled. About 20% (95% CI = 11 to 33%) of leatherback turtles captured in the DSLL fishery were dead when they were brought alongside the fishing vessel. Separating juveniles, from adults, 13% (95% CI = 4 to 38%) of adults were dead at-vessel while 54% (95% CI = 29 to 77%) of juveniles were dead at vessel (none of the turtles estimated at 4 ft. were dead at-vessel); this assessment assumes these percentages are applicable to future captures.

All of the observed leatherbacks that were dead at-vessel were captured at latitudes south of 15°N and there have been no observed at-vessel moralities of leatherbacks in the DSLL since 2016. The proportion of sets that occur south of 15°N has generally declined since 2004 and may partially explain the absence of observed leatherback mortalities in the fishery (Figure 60). From 2004 to 2016 the DSLL averaged 12.2% of sets south of 15°N while from 2017 to 2022 the average was 7.0% and this difference is statistically significant (P=0.012 for a two-tailed T-Test). Of particular note, 56% of the observed dead leatherbacks occurred in the southernmost area of captures around Palmyra (Figure 56), and no sets have occurred in this area since 2016. We also note that captures in this area have a higher proportion of juveniles with a mean of 62.5% and juveniles have higher at-vessel mortality rates than subadult/adult turtles.

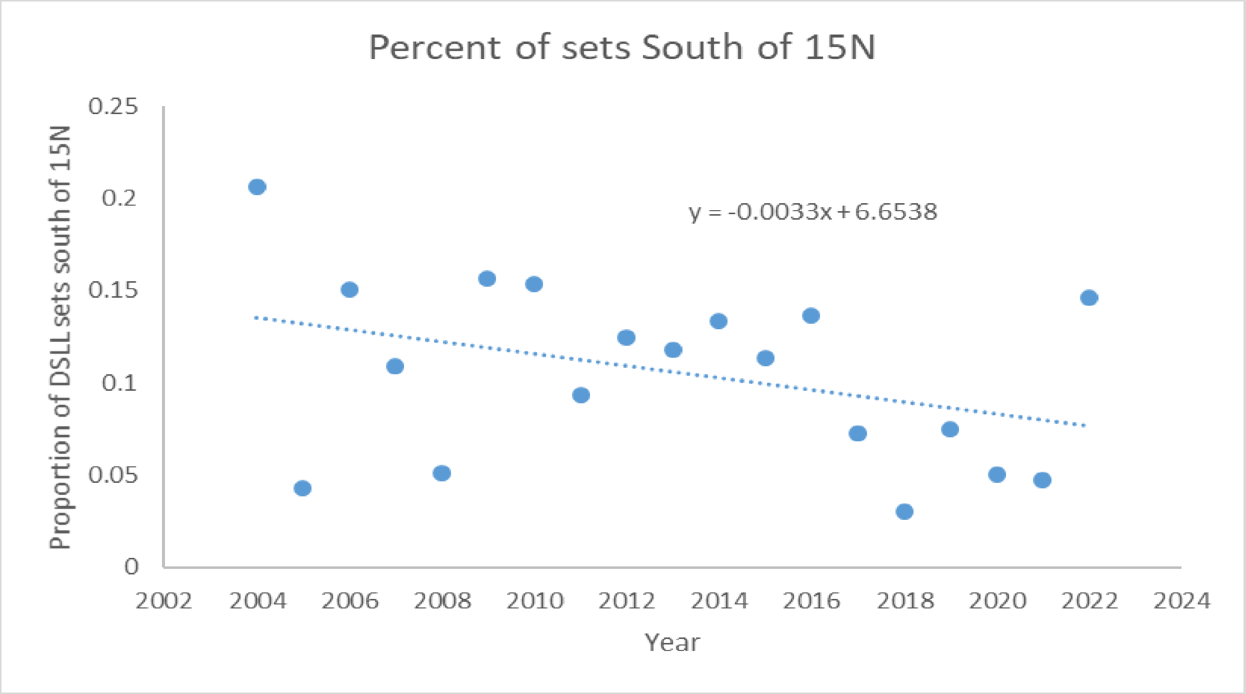

Figure 60. Proportion of DSLL sets occurring south of 15°N from 2004 to 2022.

We used Ryder et al. (2006) to assign post-interaction mortality rates to leatherback sea turtles that were alive when they were brought alongside the fishing vessel (Table 13). In some cases there was not enough information to assign a single mortality rate as it was not clear from the observer information how much gear was remaining on the turtle. In these cases low and high mortality rates were recorded and the midpoint of the two estimates was used as the mortality rate.

Table 34 shows the number of leatherback sea turtles in each injury category and the corresponding post-interaction mortality rate for the 37 observed turtles released alive. Instances where there are 0.5 individuals represent when a midpoint mortality was used and we assigned half of the turtle to each injury category. Table 35 and Table 36 show the same data separated by juveniles, subadults/adults where subadults are those turtles recorded as 4 ft. Because we consider a portion of subadults to be adults, we combine post-interaction mortalities for those categories to estimate adult fishery mortality. When at-vessel mortality and post-release mortality are combined, the effective mortality rate for leatherback turtles captured in this fishery is 35% (95% CI = 23% to 50%). Again, separating juveniles from subadults/adults, 24% (95% CI = 12% to 43%) of adults are expected to die as a result of their interaction with the fishery while 60% (95% CI = 34% to 81%) of juveniles are expected to die. Over our analytical time frame of 40 years[10] for this consultation, an estimated 736 leatherback sea turtles would be captured and 259 would die from their capture.

As noted above we estimate that 44% of interactions are anticipated to be adults. If 17 leatherback sea turtles are captured each year, we estimate that up to 8 of them may be adults, and of those about 2 (upper 95% CI=3) of them are likely to die. If 43 turtles are captured, we estimate that up to 19 of them would be adults and of those about 5 (upper 95% CI= 8) would die (Table 37). For the maximum 5-year running average, we expect a 5-year running average of 18.4 captures per year and we anticipate that 8.1 (18.4*44%) will be adults and of those, 1.9 (8.1*24%; upper 95% CI= 4.0) will die.

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[10] Note that we use an analytical timeframe of 20 years in our risk assessment of West Pacific leatherbacks due to uncertainty in the long term trend which limits our ability to predict the effects on the population; though we have enough data on the effects to maintain the 40-year timeframe for East Pacific leatherbacks to be consistent with the other species analyzed in this biological opinion.

Table 34. Number of leatherback sea turtles released alive in the DSLL fishery between 2004 and 2022 by their injury and release condition. Numbers in parentheses are the corresponding post-interaction mortality rates from Ryder et al. 2006. See Table 13 for definitions.

	Release Condition			
Injury Category	**Line < 1/2 SCL**	**Line ≥ 1/2 SCL**	**No Gear**	**Grand Total**
Entangled	2 (0.6)	-	3 (0.02)	5
Hooked Externally	9.5 (0.15)	6 (0.3)	11 (0.10)	26.5
Hook Insertion Point Not Visible	-	-	-	-
Hook Insertion Point Visible	1.5 (0.45)-	-	-	1.5
Jaw Hooked	2 (0.45)	-	2 (0.15)	4

Table 35. Number of juvenile leatherback sea turtles less than or equal to 113 cm SCL released alive in the DSLL fishery between 2004 and 2022 by their injury and release condition. Numbers in parentheses are the corresponding post-interaction mortality rates from Ryder et al. 2006. See Table 13 for definitions.

	Release Condition			
Injury Category	**Line < 1/2 SCL**	**Line ≥ 1/2 SCL**	**No Gear**	**Grand Total**
Entangled	-	-	1 (0.02)	1
Hooked Externally	1 (0.15)	1 (0.3)	3 (0.10)	5
Hook Insertion Point Not Visible	-	-	-	-
Hook Insertion Point Visible	-	-	-	-
Jaw Hooked	-	-	-	-

Table 36. Number of subadult (4 ft.) and adult leatherback sea turtles released alive in the DSLL fishery between 2004 and 2022 by their injury and release condition. Numbers in parentheses are the corresponding post-interaction mortality rates from Ryder et al. 2006. See Table 13 for definitions.

	Release Condition			
Injury Category	**Line < 1/2 SCL**	**Line ≥ 1/2 SCL**	**No Gear**	**Grand Total**
Entangled	-	-	2 (0.02)	2
Hooked Externally	5.5 (0.15)	3 (0.3)	7 (0.10)	15.5
Hook Insertion Point Not Visible	-	-	-	-
Hook Insertion Point Visible	1.5 (0.45)	-	-	1.5
Jaw Hooked	2 (0.30)	-	2 (0.15)	4

Table 37. Number of leatherback sea turtles that are likely to die as a result of being captured in the DSLL each year based on the mean effective mortality rate (35% for all size classes, 30% for adults).

	Anticipated mortalities based on the mean mortality rates			
Rate	**Mean**	**95th Percentile**	**Maximum 5-yr Running Average**	**Maximum 5-yr Running Sum**
		All Age Classes		
Annual	6	15	7	32
		Adults		
Annual	2	5	2	10

If we allocate these mortalities between the East and West Pacific leatherback turtle populations, we expect 6 of the 16 leatherback turtles captured from the West Pacific population to die as a result of their capture in any given year, and we would not expect any of the leatherback turtles from the East Pacific population to die as a result of their capture. If the number of leatherback turtles captured are as high as 43, we would expect 14 leatherback turtles from the West Pacific population to die as a result of their capture and 1 leatherback turtle from the East Pacific population to die as a result of its capture. Over our analytical time frame of 20 years for West Pacific leatherback, we anticipate that a total 345 leatherback sea turtles from the West Pacific population will be captured and of those 141 would die. Over this same timeframe we would expect 23 East Pacific leatherback captures and of those, 8 would die. Over the 40-year timeframe we anticipate that a total of 46 leatherback sea turtles from the East Pacific population will be captured and of those, 16 would die.

Risk Analyses

East Pacific Ocean Population

The East Pacific Ocean population of leatherback turtles, which once supported about 39% of the global abundance of leatherback turtles, has declined by more than 95% over the past three generations (Eckert 1993; Martinez et al. 2007; Wallace et al. 2013b; NMFS and FWS 2020a). As noted in the Status of Listed Resources section, we estimate that there were a total of 1,274 adults inclusive of both males and females with a total population size of 60,611 (CI: 34,050 to 95,462) individuals as of 2020. Note that we do not have CI for the estimated number of adults as NMFS and FWS (2020a) provided the single mean value, but we construct CI on the total population size estimate based on the mean and CI of the percent of a leatherback population represented by adults from Jones et al. (2012). We also estimate that the population will continue to decline at a rate of 8.1% per year (NMFS and FWS 2020a), giving us an estimated 2022 population abundance of 1,081 adult turtles or 51,412 total population size.

As described previously, NMFS predicts that the Hawaii-based DSLL fishery will interact with a mean of 17 (95th percentile: 43) leatherback sea turtles per year and of those, 1 (95th percentile: 3) will be from the East Pacific population. Of these we anticipate that a mean of 0.4 (95th percentile: 1) will die each year from their interaction, in other words, based on the mean, we do not expect a mortality each year but over 5 years we anticipate 2 total mortalities. Based on the maximum 5-year running average of 18.4, we expect one of them to be from the East Pacific population and we again do not expect a mortality to occur every year. Based on the maximum 5-year running sum of 92, we expect 184 total leatherback captures over 10 years of those 11 would be from the East Pacific population and we expect four of those to die over the 10 year period. Over a 40 year time horizon, if captures remain consistent with historical numbers we would anticipate a cumulative total of 736 captures, of those 46 would be from the East Pacific population and we would anticipate that 16 would die as a result of their interactions.

We looked at the percent of the population that would be captured or killed by the fishery over the next 40 years assuming interaction rates in the DSLL remain stable, acknowledging that this assumption is not realistic if the population continues to decline as predicted (Table 38). We looked at these proportions for the estimated mean number of adults and the mean of the estimated total population size. For adults, the percent killed by the fishery in any one year was less than 0.287% and this is assuming fishery captures remain constant while the population declines by 97%, which is not likely to occur. Therefore, we expect the actual impact of the DSLL on the adult portion of the East Pacific leatherback to be less than 0.287% of the population.

The impact of these losses on the rate of decline in this population are much smaller than our confidence in the underlying data. As noted in the Status of Listed Resources section, trends on individual nesting beaches range from -15.5% to +9.5% per year, readily encompassing the predicted impact of the DSLL in terms of shifts in the average trend (Table 5). We estimate the standard deviation for the East Pacific population trend to be 0.0979 and used this value in a power analysis for detecting differences in means with the same variance (Cohen 1988). The results indicate that almost 150,000 years of data would be required to detect a statistically significant difference in these values (see Appendix C for details on this analysis). A change of 0.287% cannot be statistically detected when the 95% CI around the estimate of a value ranges 25%. We consider our analysis to represent the most conservative reasonable scenario because we assume fishery captures will remain the same as the population declines. However, even if it did occur, it would not be statistically detectable. Therefore, we are reasonably certain the consequences of the proposed action will not be appreciable biologically to the numbers, reproduction, or distribution of the West Pacific leatherback sea turtle population. We consider the implications of actions' effect on leatherback sea turtles, as a species, in the Integration and Synthesis section of this opinion.

Table 38. Impact of DSLL incidental captures and mortalities on East Pacific leatherback sea turtles assuming captures remain constant over time. Capture numbers are based on the 5-year running average and sum (Table 8). Population growth rates without DSLL mortalities calculated as baseline population growth rate (-8.11% per year) plus the percent of abundance killed by the fishery each year.

Population Segment and Statistic	**Abundance**	**Captures**	**Mortalities**	**% of Abundance Captured**	**% of Abundance Killed**	**Population Growth Rate with DSLL Mortalities(% per year)**	**Population Growth Rate without DSLL Mortalities (% per year)**
2022							
Mean Adults	1,083	0.5	0.1	0.047	0.011	-8.11	-8.10
Mean Total Population	51,536	1.2	0.4	0.002	0.001	-8.11	-8.11
2032							
Mean Adults	481	0.5	0.1	0.105	0.025	-8.11	-8.09
Mean Total Population	22,903	1.2	0.4	0.005	0.002	-8.11	-8.11
2042							
Mean Adults	214	0.5	0.1	0.237	0.057	-8.11	-8.08
Mean Total Population	10,178	1.2	0.4	0.011	0.004	-8.11	-8.11
2052							
Mean Adults	95	0.5	0.1	0.532	0.128	-8.11	-8.06
Mean Total Population	4,523	1.2	0.4	0.025	0.009	-8.11	-8.11

Population Segment and Statistic	**Abundance**	**Captures**	**Mortalities**	**% of Abundance Captured**	**% of Abundance Killed**	**Population Growth Rate with DSLL Mortalities(% per year)**	**Population Growth Rate without DSLL Mortalities (% per year)**
2062							
Mean Adults	42	0.5	0.1	1.197	0.287	-8.11	-8.02
Mean Total Population	2,010	1.2	0.4	0.057	0.020	-8.11	-8.10

West Pacific Ocean Population

As discussed in Section 1.3.1, the data do not allow us to reliably predict the proposed action's effects on West Pacific leatherbacks beyond 20 years. Therefore, we use an analytical timeframe of 20 years here. Leatherback sea turtles in the West Pacific Ocean population have declined 83% during the past three generations, and in 2017 the population was composed of an estimated 1,443 mature adults. Based on NMFS' PVA model that used nesting beach data from 2001 to 2017 (Martin et al. 2020a and updated by Siders et al. 2023), leatherback sea turtles in the West Pacific Ocean population are declining at about 6% per year (95% CI: -23.8% to 12.2%; standard deviation: 0.894), and the population as indicated by the index beaches is at risk of falling to less than half of its 2017 abundance by 2029 (95% CI, 2022 to 2042) if the 6% per year decline continued from 2017. PVA modeled estimates suggest the index beaches of Jamursba-Medi and Wermon consisted of about 790 adult female leatherback sea turtles as of 2017 (median sum of total reproductive females, LCI = 666). As noted in the Status of Listed Resources section of this biological opinion, these two nesting beaches represent approximately 50-75% of the total West Pacific population nesting. Assuming a 73% female sex ratio and based on NMFS' results for median nest counts (Martin et al. 2020a), the total number of adult female leatherback sea turtles in 2017 in the West Pacific Ocean population would be 1,053 (95% CI: 888-1,256) if the index beaches represent 75% of the population.

Preliminary data from the Jamursba Medi and Wermon index beaches indicate that nest numbers were relatively stable from 2017 to 2021 (Lontoh et al. In Prep, see Figure 24) but, as discussed in Section 1.2.2, the data are not yet available in sufficient detail to update the model of Martin et al. (2020a). Hence we acknowledge that there is a great deal of uncertainty associated with the current status of West Pacific leatherback sea turtles, as represented by the two index beaches.

Martin et al. (2020b) and Siders et al. (2023) present essentially the same PVA as described in Martin et al. (2020a), however they consider the impact of the DSLL fishery on the West Pacific leatherback sea turtle and Siders et al. (2023) updated the model as described in Section 1.4.2 by incorporating DSLL observed leatherback capture data up to 2021. For both modeling efforts, they 'removed' the historic impact of the fishery by adding back to the population observed and estimated leatherback sea turtle mortalities caused by the fishery. An ANE was assigned to each juvenile-sized sea turtles (see Section 1.4.2), and the lost adult nester equivalent was added back into the year when it would have first nested and all subsequent years when it would have nested based on remigration intervals and adult survival rates (for this historic impact of the fishery only).

Siders et al. (2023) then conducted the updated PVA with and without the fishery by projecting numbers of nesting females into the future, removing adult nester equivalents each year based on probabilistic draws for total annual captures and size and fishery mortality for each modeled capture. The results of this analysis found that the probability of the population reaching 50% of 2017 abundances within the next 5 years, or by 2026, increased by 2.4% with the impact of the DSLL and using the updated method for determining mortality (Table 39). The probability of the population reaching 25% of 2017 abundances within the next 10 years (or by 2031) increased by 3.9% with the fishery compared to without. These probabilities amount to a mean of about one year sooner to reach the 50% threshold with the fishery and about 2 years sooner to reach the 25% threshold with the fishery, if the 6% per year population declines continued (Table 40).

Overall the population is expected to reach 50% of current abundance in a mean of 7.85 years (95% CI: 1 to 21 years) without the fishery and 7.33 years (95% CI: 1 to 19 years) with the fishery (Table 40).

Note that these probabilities are higher than those presented in Table 10 of Martin et al. (2020b). For example, the median probability of reaching 50% of 2017 population sizes in 10 years (or by 2027) with and without the fishery is 0.39 in Martin et al. (2020b). In Siders et al. (2023) the probability of reaching 50% of 2017 population sizes in 5 years starting in 2021 (or by 2026) is 0.34 without the fishery and 0.40 with the fishery. The difference in the apparent impact of the fishery is because the population was projected to decline at a mean rate of 6% per year but fishery captures remained essentially the same, making the captures represent a higher proportion of the population. The difference in the time to reach thresholds is because the probabilities in Siders et al. (2023) are based on time from 2021 while the probabilities in Martin et al. (2020b) are based on time from 2017, a difference of four years.

The probabilities of reaching thresholds and the time to reach thresholds in Siders et al. (2023) include the conservative assumptions that all DSLL fishery mortalities impact only the Jamursba-Medi and Wermon index beaches, and that fishery captures will remain the same as the population declines. The second assumption implies that the proportion of the population impacted by the fishery increases over time as the population declines. Therefore, we can consider these results to be the most conservative reasonable scenarios as: 1) leatherbacks interacting with the fishery are not likely to all be from the Jamursba-Medi and Wermon index beaches, 2) between 2017 and 2021, the population did not decline at the rate of 6% per year (Lontoh et al. In Prep; Figure 24) and 3) interactions are likely to decline as the population declines. This means that the proportion of the total population impacted by the fishery would be less and we can expect the magnitude of the impact to be less.

Table 39. Probability of reaching population thresholds of 50% and 25% of 2017 nester abundances (Siders et al. 2023). Note that Siders et al. (2023) include values for a 12.5% threshold but those results are beyond our analytical timeframe of 20 years.

			Maturity Determined as in Martin et al. (2020b)			**New Method for Maturity**		
Threshold	**Metric**	**Scenario**	**5 yr**	**10 yr**	**25 yr**	**5 yr**	**10 yr**	**25 yr**
50%	**Median**	**No Take**	0.341	0.725	0.989	0.388	0.758	0.995
		Take	0.403	0.807	0.995	0.412	0.787	0.997
		Δ(NT-T)	-0.062	-0.082	-0.006	-0.024	-0.029	-0.002
	Lower 95% CI	**No Take**	0.331	0.716	0.988	0.378	0.75	0.994
		Take	0.393	0.799	0.994	0.401	0.779	0.997

			Maturity Determined as in Martin et al. (2020b)			**New Method for Maturity**		
		Δ(NT-T)	-0.062	-0.083	-0.007	-0.023	-0.029	-0.002
	Upper 95% CI	**No Take**	0.35	0.734	0.992	0.398	0.767	0.997
		Take	0.413	0.816	0.997	0.422	0.796	0.999
		Δ(NT-T)	-0.063	-0.081	-0.004	-0.024	-0.029	-0.002
25%	**Median**	**No Take**	0.001	0.068	0.799	0	0.081	0.815
		Take	0.003	0.13	0.909	0	0.12	0.9
		Δ(NT-T)	-0.002	-0.062	-0.11	0	-0.039	-0.085
	Lower 95% CI	**No Take**	0	0.062	0.791	0	0.076	0.808
		Take	0.001	0.122	0.904	0	0.113	0.894
		Δ(NT-T)	-0.001	-0.06	-0.112	0	-0.038	-0.082
	Upper 95% CI	**No Take**	0.001	0.072	0.808	0	0.087	0.824
		Take	0.003	0.137	0.916	0	0.127	0.907
		Δ(NT-T)	-0.002	-0.064	-0.108	0	-0.04	-0.082

Table 40. Time to reach population thresholds with and without the DSLL fishery (Siders et al. 2023). Note that Siders et al. (2023) include values for a 12.5% threshold but those results are beyond our analytical timeframe of 20 years.

Threshold	**Scenario**	**Mean**	**Median**	**Lower 95% CI**	**Upper 95% CI**
50%	No Take	7.85	7	1	21
	Take	7.33	6	1	19
	Δ(NT-T)	0.53	1	0	2
25%	No Take	19.24	18	9	36
	Take	17.14	16	8	32

Threshold	**Scenario**	**Mean**	**Median**	**Lower 95% CI**	**Upper 95% CI**
	Δ(NT-T)	2.11	2	1	4

As described previously, NMFS predicts that the DSLL will interact with a mean of 17 (95th percentile: 43) leatherback sea turtles per year and of those, 16 (95th percentile: 40) will be from the West Pacific population. Of these we anticipate that a mean of 6 (95th percentile: 14) will die each year from their interaction based on the mean interaction mortality rate of 35%. The maximum 5-year running average is 18.4. We estimate that an average of 1.8 adults may be killed by the fishery in a year (note: these are adults, not juveniles extrapolated to adult equivalents as in Martin et al. [2020b] and Siders et al. [2023]). Applying the sex ratio of 0.73, these adults equate to 1.2 adult females (not ANE).

We looked at the percent of the population that would be captured or killed by the fishery over the next 20 years assuming average annual numbers of captures in the DSLL remain stable (Table 41). Given the preliminary data of Lontoh et al. (In Prep), the adult population size appeared stable from 2017 to 2021 and we assume that the population size in 2022 was the same as 2017. Starting in 2022, we consider the possibility of the population declining by 6% per year over the next 20 years and the impact of the DSLL is captures remain constant by considering the proportion of the population captured and killed by the fishery. We looked at these proportions for the estimated mean number of adults and the mean of the estimated total population size.

For adults, the percent of the 2017 and 2022 population abundances killed by the fishery in any one year represents 0.13% of West Pacific adults and 0.01% of the total population size (Table 41). By 2042, if the population declines by 6% per year after 2022 while fishery captures remain the same, the fishery would kill 0.42% of the adult population abundance and 0.02% of total population abundance per year. We consider the 2023 and 2042 values the most conservative reasonable scenarios as 1) fishery interactions are not likely to stay the same if the population declines by 70% from 2022 to 2042 abundances as predicted by a mean decline of 6% per year, and 2) based on preliminary nesting beach data from 2018 to 2021, the population does not appear to be declining and the population sizes at 2032 and 2042 may be pessimistic. We also considered the scenario were the DSLL mortalities were added back to the population each year and estimated the resulting population growth rates without the impact of the DSLL for the time periods 2022-2023, 2022 to 2032 and 2022-2042 (Table 41).

We explore the impact of the fishery mortalities on time to reach population thresholds using the difference in population growth rates presented in Table 41 and the exponential population growth equation:

$$N_t = N_0 * \exp(rt)$$

Where N_0 is the starting population size, N_t is the population size after time duration t, and r is the population growth rate. This equation can be rewritten as:

$$N_t/N_0 = \exp(rt) \text{ or } \ln(N_t/N_0) = rt\text{, which can be solved for time as:}$$

$$t = \ln(N_t/N_0)/r.$$

Hence, the time for the West Pacific leatherback adult population to reach 50% of 2022 abundances with the fishery is ln(0.50)/-0.06 or 11.55 years. Removing the impact of the fishery this value becomes ln(0.50)/-0.0587 or 11.81 years for a difference of 0.26 years (Table 42). These values would be reached by 2033 to 2034. This is very comparable to the results of Siders et al. (2023) who predict median times of 7.85 without the fishery and 7.33 years with the fishery from 2021. If you add the 4 yr difference from 2017 to 2021, the time would be 11.85 without the fishery and 11.33 yrs with the fishery with a similar difference of 0.53 years. We conducted these calculations for the mean impact of the fishery from 2022 to 2042 (Table 42). Assuming current (2022) population sizes are similar to 2017 then we expect a mean difference of times for the adult population to reach 50%, 25% and 12.5% of 2022 abundances of less than 0.77 yr with and without the fishery. Under the most conservative reasonable scenario of the population continuing to decline while fishery captures remain the same, the difference with and without the fishery is a maximum of 0.76 years for the population to reach 12.5% of 2022 abundances (Table 42).

The impact of these losses on the rate of decline in this population are much smaller than our confidence in the underlying data. As noted in the Status of Listed Resources section, the 95% credible interval for the population growth rate estimated by Martin et al. (2020a) is -23.8% to 12.2% and the standard deviation of 0.0896. The greatest impact we computed is the mortality of 0.42% of adults in year 20, under the conservative assumption that interactions stay the same despite the population declining by 70%. If the growth rate has improved from a mean of -6% as recent evidence suggests, population numbers will be higher and the proposed action's effect on the growth rate will be lower than we calculated.

Even at the mean rate of decline of 6% per year and the most conservative reasonable scenario of fishery captures remaining the same as the population declines, the difference between the scenarios with and without the fishery is an increase in the population trend to -5.75%. Based on a power analysis and assuming the same standard deviation of 0.0896, over 15,000 years of data would be required to detect a statistically significant difference in these values (see Appendix C for details on this analysis).This difference cannot be statistically detected when the 95% CI around the estimate of a value ranges 36%. Even if the most conservative reasonable scenario happens, it would not be statistically detectable. Therefore, we are reasonably certain the consequences of the proposed action will not be appreciable biologically to the numbers, reproduction, or distribution of the West Pacific leatherback sea turtle population.

We acknowledge the uncertainty in the status of this population and the reliance of the status on continued nesting beach monitoring and protection programs. If necessary, we will reassess our analysis of the impacts as more updated and detailed nesting beach data become available. We consider the implications of the effect of the fishery on leatherback sea turtles, as a species, in the Integration and Synthesis section of this opinion.

Table 41. Impact of DSLL incidental captures and mortalities on West Pacific leatherback sea turtles assuming captures remain constant over time. Capture numbers are based on the 5-year running average and sum (Table 8). Population growth rates with DSLL mortalities calculated as baseline population growth rate (-6% per year) minus the percent of abundance killed by the fishery each year.

Population Segment and Statistic	Abundance	Captures	Mortalities	% of Abundance Captured	% of Abundance Killed	Population Growth Rate With DSLL Mortalities (% per year; Martin et al. 2020b)	Population Growth Rate DSLL Mortalities Removed (% per year)
2017							
Mean Adults	1,442	7.6	1.8	0.53	0.13	-6.0	-5.87
Mean Total Population	99,921	17.3	6.0	0.02	0.01	-6.0	-5.99
2022 (assuming stable population since 2017)							
Mean Adults	1,442	7.6	1.8	0.53	0.13	-6.0	-5.87

Population Segment and Statistic	Abundance	Captures	Mortalities	% of Abundance Captured	% of Abundance Killed	Population Growth Rate With DSLL Mortalities (% per year; Martin et al. 2020b)	Population Growth Rate DSLL Mortalities Removed (% per year)
Mean Total Population	99,921	17.3	6.0	0.02	0.01	-6.0	-5.99
2032 (assuming 6% per year decline)							
Mean Adults	792	7.6	1.8	0.96	0.23	-6.0	-5.82
Mean Total Population	54,838	17.3	6.0	0.03	0.01	-6.0	-5.99
2042 (assuming 6% per year decline)							

Population Segment and Statistic	Abundance	Captures	Mortalities	% of Abundance Captured	% of Abundance Killed	Population Growth Rate With DSLL Mortalities (% per year; Martin et al. 2020b)	Population Growth Rate DSLL Mortalities Removed (% per year)
Mean Adults	434	7.6	1.8	1.75	0.42	-6.0	-5.75
Mean Total Population	30,096	17.3	6.0	0.06	0.02	-6.0	-5.99

Table 42. Mean times for the adult portion of the West Pacific leatherback to reach 2017 population thresholds with and without the DSLL fishery based on population growth rates in Table. Zeros indicate that the threshold will have been reached by that year, based on mean time to threshold.

Threshold	50% of 2017 Population Size = 721	25% of 2017 Population Size = 361	12.5% of 2017 Population Size = 180
Time from 2017 to threshold with DSLL (yr.)	11.55	23.11	34.66
2017 Time to threshold without DSLL (yr.)	11.81	23.62	35.43
2017 Difference (yr.)	0.26	0.51	0.77
Time from 2022 to threshold with DSLL (yr.)	11.55	23.11	34.66
Time from 2022 to threshold without DSLL (yr.)	11.81	23.62	35.43
2022 Difference (yr.)	0.26	0.51	0.77
Time from 2032 to threshold with DSLL (yr.)	1.56	13.11	24.66
Time from 2032 to threshold without DSLL (yr.)	1.61	13.52	25.43

Threshold	50% of 2017 Population Size = 721	25% of 2017 Population Size = 361	12.5% of 2017 Population Size = 180
2032 Difference (yr.)	0.05	0.41	0.76
Time from 2042 to threshold with DSLL (yr.)	0	3.11	14.66
Time from 2042 to threshold without DSLL (yr.)	0	3.25	15.30
2042 Difference (yr.)	0	0.14	0.64

4.3.6 North Pacific Loggerhead Sea Turtle

Exposure Analysis

Loggerhead sea turtles have been captured in the DSLL fishery in 10 of the 19 years between 2004 and 2022. During this 19-year time interval fishery observers reported that 19 loggerhead sea turtles were captured in the DSLL fishery. When these reports are adjusted to account for the percentage of observer coverage, about 94 loggerhead sea turtles are likely to have been captured in the fishery in the 18-year period from 2004 to 2022 (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a). The value for 2022 was estimated based on the expansion factor with 3 observed captures and observer coverage of 20.22% (5/[20.22/100]=14.8 rounded to 15).

All of the loggerhead turtles captured in this fishery would be North Pacific loggerhead sea turtles, which nest at one or more of the nesting beaches in Japan. Seventeen of the 19 loggerhead sea turtles observed captured in the DSLL had carapace length recorded. Based on these 17 turtles, sizes of loggerhead sea turtles captured by the DSLL fishery ranged from 57.5 to 76 cm SCL (Figure 61). The minimum size of nesting North Pacific loggerhead sea turtles is 75 cm SCL (Hatase et al. 2004). Therefore loggerhead sea turtles interacting with the fishery are primarily juveniles, although the six largest, ranging from 73.5 to 76 cm SCL may be subadults or adults.

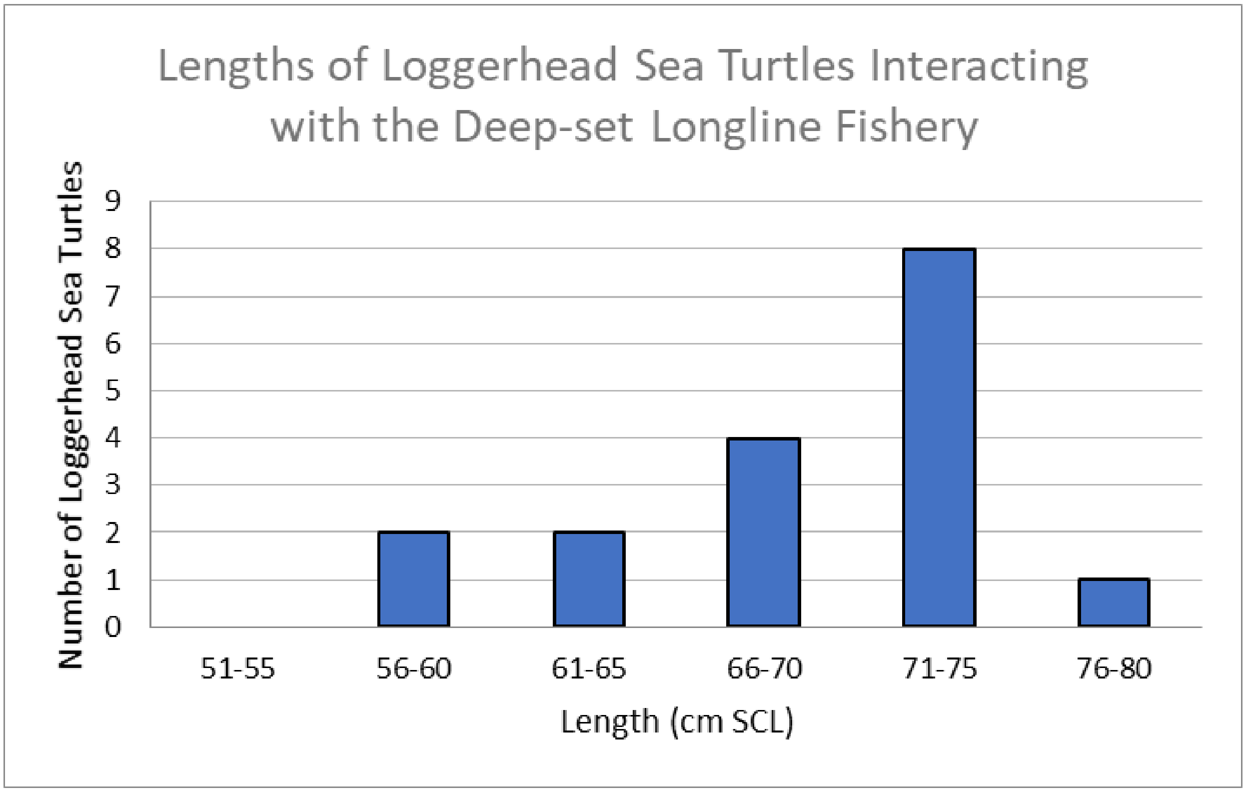

Figure 61. Lengths of observed North Pacific loggerhead sea turtles interacting with the DSLL fishery from 2004 to 2022 (*n*= 17).

Spatially, North Pacific loggerhead turtles were captured across the longitudinal range of the highest DSLL fishery effort (Figure 63), but all captures occurred north of 14° North latitude. As with all species considered in this biological opinion, a hotspot analysis was conducted using ArcGIS to map the observed interactions with loggerhead sea turtles (Figure 62). We highlight, blue areas have lower density of interactions than those areas with orange or reds. The key was adjusted to illustrate low versus high-density values and is consistent with all other spatial maps presented. Therefore, the data show most interactions with the DPS occur to the northeast of the MHI just outside of the EEZ. Additionally, while the density was strong enough to create a heatmap, we caution that a small number of data were used to generate this map and some locations did not have sufficient density to project. However, this DPS does not follow the same spatial trends as identified in other species caught in this fishery. Although 2021 and 2022 data are not shown in the heatmap analysis, the incidental interactions with this DPS (*n* =4) occurred within the same geographic area as depicted in the figure. Therefore, we would not expect any significant changes in the data, but only strengthen the density values depicted within this geographical area.

About 56% of loggerhead turtles captured between 2004 and 2021 were captured in February and March (Figure 64). Smaller numbers of loggerheads would be expected to be captured in January, May, August, September, October, and December and to-date, no observed captures have occurred in April, June, or July.

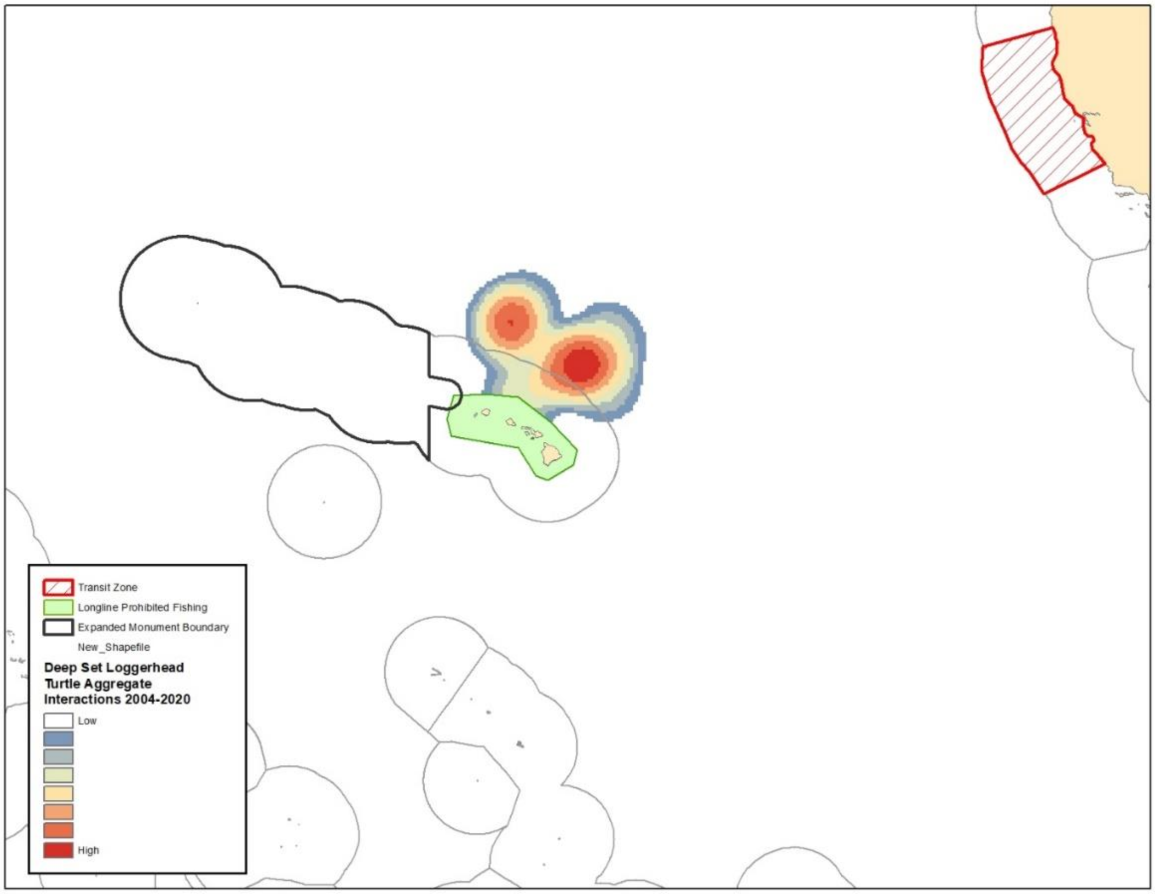

Figure 62. Geographical representation of observed aggregate North Pacific loggerhead sea turtle interactions with the DSLL fishery between 2004 and 2020 (*n* = 15).

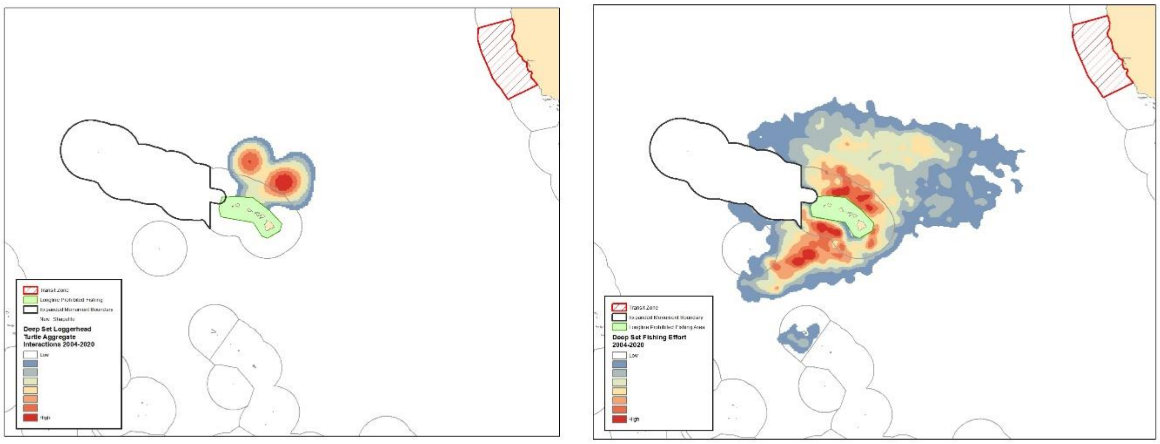

Figure 63. Heatmap comparison of observed North Pacific loggerhead interactions (left; *n* = 15) versus aggregated observed fishing effort during this same time frame (right; *n*= 65,245 from 2004 to 2020).

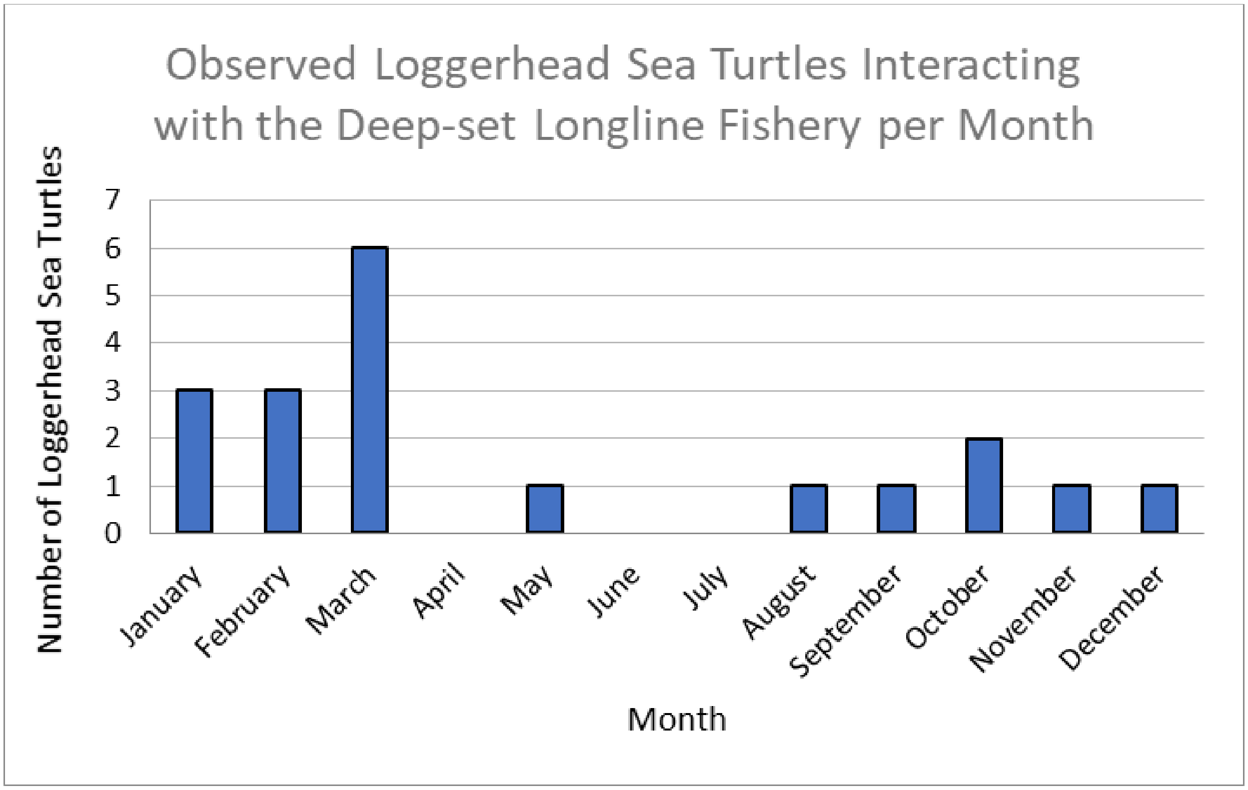

Figure 64. Numbers of observed North Pacific loggerhead sea turtles captured per month in the DSLL fishery from 2004 to 2022 (*n*= 19).

Annual observed and estimated North Pacific loggerhead sea turtle interactions peaked in 2020 with 3 observed and 19 estimated (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a) captures (Figure 65).

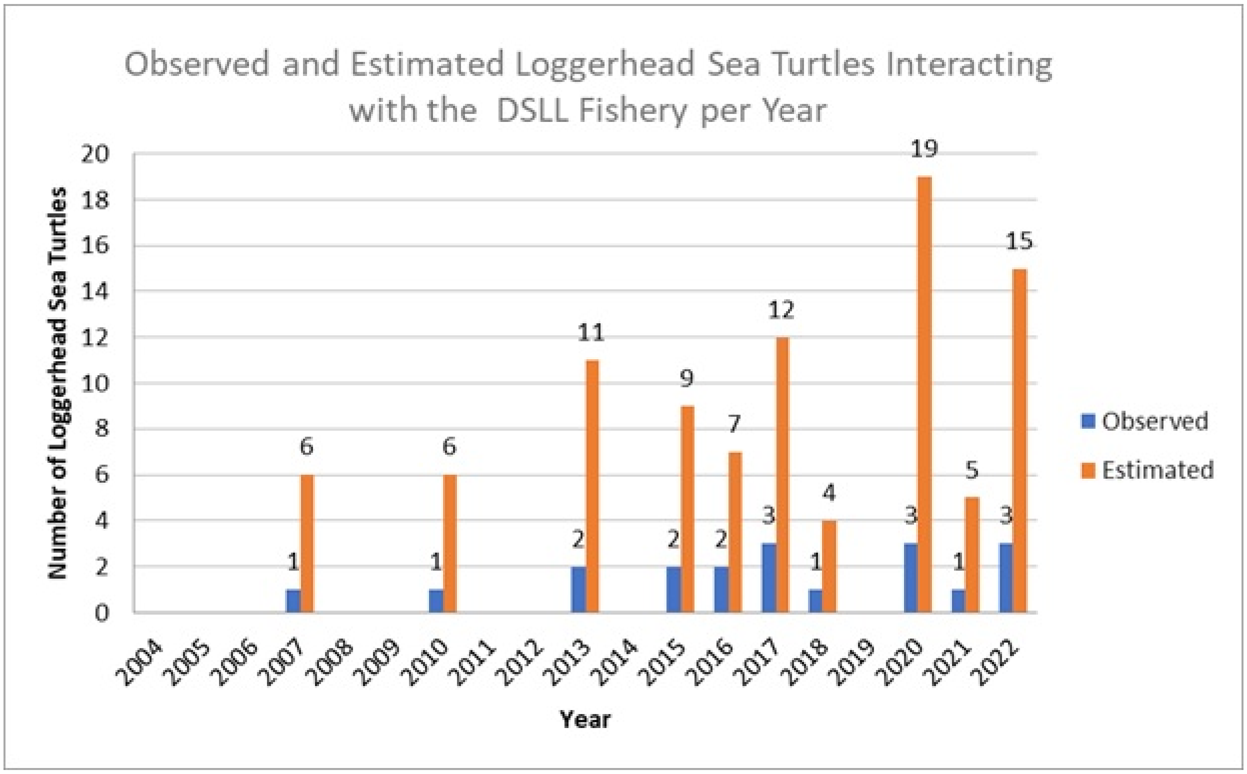

Figure 65. North Pacific loggerhead sea turtle interactions per year in the DSLL fishery between 2004 and 2022 for observed (blue bars) and estimated interactions (orange bars; McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a). Estimated interactions for 2022 were calculated using the expansion factor based on the observed captures and the observer coverage.

Estimates of the number of North Pacific loggerhead turtles that are likely to be captured in the DSLL fishery each year are presented in Table 43 (estimates are from McCracken 2019a, McCracken and Cooper 2020a, 2020b, 2021a, 2022a). On average, we would expect 9 loggerhead turtles to be captured each year in the DSLL fishery, although the number captured in any given year could be as high as 27. As with green and leatherback turtles, only a portion of these turtle captures would be reported by observers because the DSLL fishery has only 20% observer coverage. We also report the maximum 5-yr running average and the cumulative number of captures over 5 years (Table 43). There is no 95th percentile metric associated with the running average.

Table 43. Number of North Pacific loggerhead sea turtles expected to be encountered (exposed) in the DSLL per year (McCracken 2019a) ; and the maximum 5-yr running average and running sum from 2004 to 2022 based on estimates of McCracken (2019a) and McCracken and Cooper (2020a, 2020b, 2021a, 2022a).

	DSLL Encounters	
Rate	**Mean**	**95th Percentile**
Annual	9.1	27.3
Maximum 5-Yr Running Average	8.6	N/A
Maximum 5-Yr Running Sum	43	N/A

Response Analysis

If the patterns found in the observer data from 2004 to 2022 are representative, all the loggerhead turtles captured in the fishery would be hooked rather than entangled. About 47% (95% CI = 27% to 68%) of loggerhead turtles captured in the DSLL fishery have been dead when they were hauled back to the fishing vessel; this assessment assumes these percentages are applicable to future captures.

We used the post-interaction mortality rates of Ryder et al. (2006) for hard-shell sea turtles to estimated post-interaction mortality of loggerhead sea turtles that were alive when they were captured. Table 44 shows the number of loggerhead sea turtles in each injury category and the corresponding post-interaction mortality rate for the seven observed loggerhead sea turtles released alive. When at-vessel mortality and post-release mortality are combined, the effective mortality rate for loggerhead sea turtles captured in this fishery is 56% (95% CI = 35% to 75%).

At this mortality rate, if 9 loggerhead turtles are captured in the fishery in any given year, we would expect an average of 5 (95% CI = 3 to 7) of them to die as a result of being captured; if 27 of these turtles are captured, we would expect an average of 15 (95% CI = 9 to 21) of them to die as a result of their capture (Table 45). Based on the maximum 5-year running average of 8.4, over a 5-year period we would expect 4.8 (95% CI = 3.0 to 6.5) loggerhead turtles to die each year on average. Over 5 years, we would expect 26 (95% CI = 16 to 34) loggerhead sea turtles to die.

Table 44. Number of North Pacific loggerhead sea turtles released alive in the DSLL fishery between 2004 and 2022 by their injury and release condition. Numbers in parentheses are the corresponding post-interaction mortality rates from Ryder et al. 2006.

	Release Condition			
Injury Category	**Line < 1/2 SCL**	**Line ≥ 1/2 SCL**	**No Gear**	**Grand Total**
Entangled	-	-	-	-
External	-	-	2 (0.05)	2
Insertion Not Visible	-	-	-	-
Insertion Visible	1 (0.45)	-	2 (0.25)	3
Jaw	1 (0.20)	-	4 (0.10)	5

Table 45. Number of North Pacific loggerhead sea turtles that are likely to die as a result of being captured in the DSLL each year based on mean estimated mortality rate (56%).

	Anticipated mortalities based on the mean (56%) mortality rates	
Rate	**Mean**	**95th Percentile**
Annual	5	15
5-year Running Average	4.8	N/A
5-year Running Sum	24	N/A

Risk Analysis

The total abundance (all life stages) of North Pacific loggerhead sea turtles has been estimated at about 340,800 sea turtles (T. Jones pers. comm. 2019). The current median abundance of adult females has been estimated at about 4,541 (95% CI = 4,074 to 5,063) from Martin et al.'s (2020a) PVA model results. This model used nesting data from three index beaches in Yakushima, Japan-Inakahama, Maehama, and Yotsusehama; which represent approximately 52% of loggerhead nesting in Japan. Martin et al. (2020a) estimate an increasing trend of 2.3% per year for these index beaches. We do not have data to ascertain trends in the remaining portion of the DPS and in our risk analysis we assume the population is stable.

Martin et al. (2020b) present the same PVA as described in Martin et al. (2020a), considering the effect of the DSLL fishery on the North Pacific loggerhead. They 'removed' the historic impact of the fishery by adding back to the population observed and estimated loggerhead sea turtle mortalities caused by the fishery. For juvenile-sized sea turtles, their ANE was estimated (see Section 2.3.2 Simulations using Bayesian State-Space Population Viability Analyses for North Pacific loggerhead Sea Turtles and West Pacific Population of Leatherback Sea Turtles), and the lost adult nester equivalent was added back into the year when it would have first nested and all subsequent years when it would have nested based on remigration intervals. The PVA was then conducted with and without the fishery by projecting numbers of nesting females into the future, removing adult nester equivalents in the same way they were added back in to remove fishery effects. The results of this analysis suggest that without the fishery, the North Pacific loggerhead sea turtle is increasing at an average of 2.4% per year, compared to 2.3% per year with the fishery. They found essentially no difference in the probabilities of reaching 50%, 25%, and 12.5% of current abundance in the population projections with and without the fishery (Table 46).

Table 46. Median probabilities of North Pacific loggerhead sea turtles reaching abundance thresholds equal to 50%, 25%, and 12.5% of current abundance at 5, 10, 25, 50, and 100 years from the end of the nesting data time series or 2015 (Martin et al. 2020b). "No Fishery" indicates the probabilities without the DSLL fishery and "Fishery" is with the fishery at historic interaction rates. In every scenario at all timeframes, the upper and lower values of the 95% CI were equal to the median value except for the interaction scenario at 100 yrs. where the 95% CI was 0.31 to 0.32.

Threshold	**Scenario**	**5 yr**	**10 yr**	**25 yr**	**50 yr**	**100 yr**
50%	No Fishery	0.01	0.09	0.21	0.28	0.32
	Fishery	0.01	0.09	0.21	0.28	0.32
25%	No Fishery	0	0.01	0.12	0.21	0.28
	Fishery	0	0.01	0.12	0.21	0.28
12.5%	No Fishery	0	0	0.06	0.16	0.25
	Fishery	0	0	0.06	0.16	0.25

As described in the Exposure Analysis, all individuals that were measured would be considered juveniles with three possible subadults. As noted above, the abundance of the North Pacific loggerhead sea turtle is estimated at 342,179 (T. Jones pers. comm. 2019; Martin et al. 2020a). If we assume that 9 (95th percentile: 27) individuals from this DPS may interact with the DSLL fishery, this would equal 0.0026% (95th percentile: 0.0079%) of the DPS total population. The mortality of 5 (95th percentile: 157) individual loggerhead sea turtles assuming total abundance equals 342,179 represents 0.0015% (95th percentile: 0.0044%) of the total population. The maximum 5-year running average of 8.6 turtles per year over five years, of which 4.8 would die, would represent 0.0025% of the population for interactions and 0.0014% of the population for mortalities. Based on the maximum 5-year running sum of 43 turtles, we anticipate 86 North Pacific loggerhead turtles to be captured by the DSLL over 10 years, and of those 48 are expected to die.

When we consider 86 interactions over the next 10 years, the rate of interaction with the fishery would be 0.025% of the total population. The mortality of 48 North Pacific loggerhead sea turtles over 10 years would represent 0.014% of the population. As noted above, while Martin et al. (2020a) estimates the Yakushima subpopulation is increasing at approximately 2.3%, given the lack of trend information for the entire DPS we assume a stable trend for the DPS (Martin et al. 2020a). Therefore, these percentages represent the proportion of the population captured and killed at 40 years into the future. Over a 40 year time horizon, if captures remain consistent with historical numbers we would anticipate a cumulative total of 344 captures and would anticipate that 193 would die as a result of their interactions. Our assessment illustrates that the DSLL fishery interacts with (incidentally captures) a very small portion of the North Pacific loggerhead sea turtle. At projected abundance levels (342,179 individuals), incidentally capturing and killing an average of five North Pacific loggerhead sea turtles per year constitutes the death of 0.0015% of the population. We consider the implications of this effect on North Pacific loggerhead sea turtles in the Integration and Synthesis section of this opinion.

4.3.7 Olive Ridley Sea Turtle

Exposure Analysis

Olive ridley sea turtles have been captured in the DSLL fishery in each of the 19 years between 2004 and 2022. During this 19-year time interval fishery observers reported that 222 olive ridleys were captured in the DSLL fishery. When these reports are adjusted to account for the percentage of observer coverage, about 1,104 olive ridley turtles are likely to have been captured in the fishery in the 19-year period from 2004 to 2022(McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a). The value for 2022 was estimated based on the expansion factor with 10 observed captures and observer coverage of 20.22% (10/[20.22/100]=49.46 rounded to 49).

Based on mitochondrial and nuclear DNA analyses of tissue samples collected from 135 olive ridley turtles captured in the DSLL fishery, 73.3% of these turtles appear to be from the both the endangered population and the threatened East Pacific population that occurs in Mexico, Costa Rica, and other areas of Central America (Dutton pers. comm. June 29, 2018). The remaining 26.7% are part of the West Pacific (Dutton pers. comm. June 29, 2018). If we apply the results of genetic analyses conducted by NMFS, about 163 of the observed captures would have represented olive ridley sea turtles from the eastern Pacific nesting aggregations (or 809 when adjusted for observer coverage) and 59 would have been from Western Indo-Pacific nesting aggregations (or 295 when adjusted for observer coverage).

Sizes of olive ridley sea turtles captured by the DSLL fishery ranged from 25 to 86 cm SCL (Figure 66). The minimum size of nesting olive ridley sea turtles is 53 cm SCL (Zug et al. 2006), therefore approximately 74% of olive ridley sea turtles interacting with the fishery may be adults.

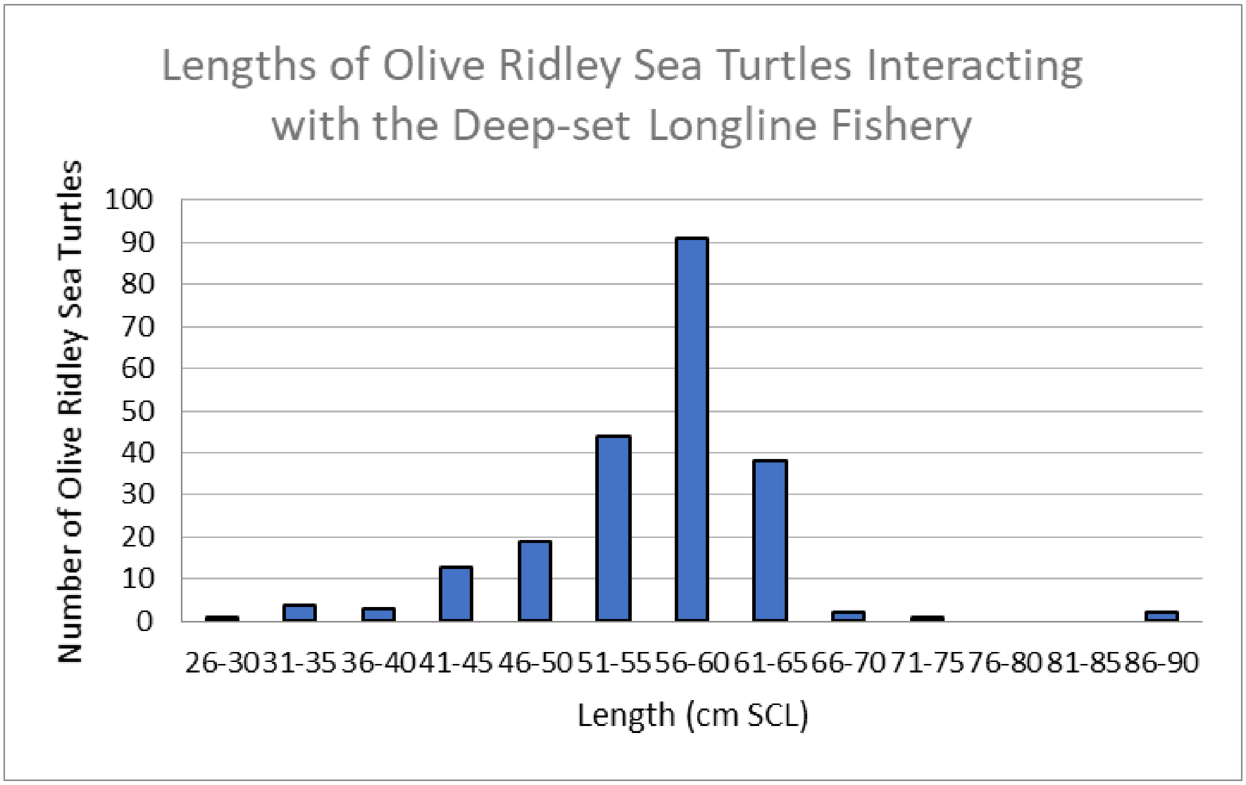

Figure 66. Lengths of observed olive ridley sea turtles interacting with the DSLL fishery from 2004 to 2022 (*n* = 218).

Spatially, the majority (82.9%) of olive ridley turtles were captured between 10° and 25° North latitude and 88.5% were captured between 150 and 170 West longitude as depicted by Figure 67. Olive ridley turtle captures are likely to occur throughout the year, with minor peaks in February and October (about 23.0% of olive ridley turtles captured between 2004 and 2022 were captured in these months; Figure 69).

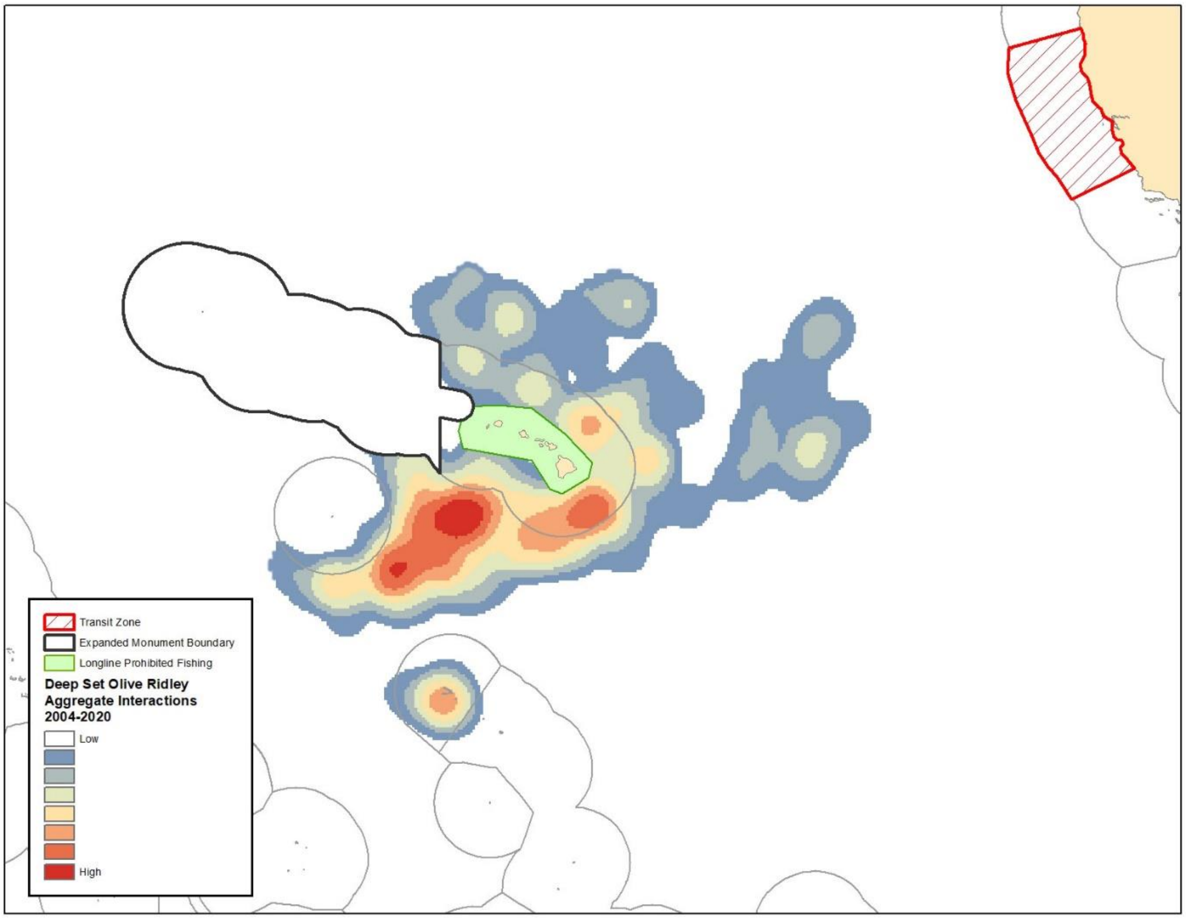

Figure 67. Geographical representation of observed aggregate olive ridley sea turtle interactions with the DSLL fishery between 2004 and 2020 (*n*= 201).

When comparing aggregate interactions throughout the time series (2004 to 2020), it is apparent that higher numbers of interactions are occurring in the same areas where there are higher numbers of interactions with oceanic whitetip sharks and giant manta rays. That is, to the south and southwest of the MHI and with a spatially distinct hotspot in Palmyra. These trends do not appear to be consistent with fishing effort (Figure 68). Fishing effort is much more diffuse and expansive primarily to the north of the MHI. Higher numbers of olive ridleys are caught south of the MHI. Although 2021 and preliminary 2022 data is not shown. All incidental interactions occurred within the area depicted in the heatmap. Almost all interactions are in the area southwest of the MHI. Therefore, we would not expect any significant changes in the data, but only strengthen the density values shown within this geographical area.

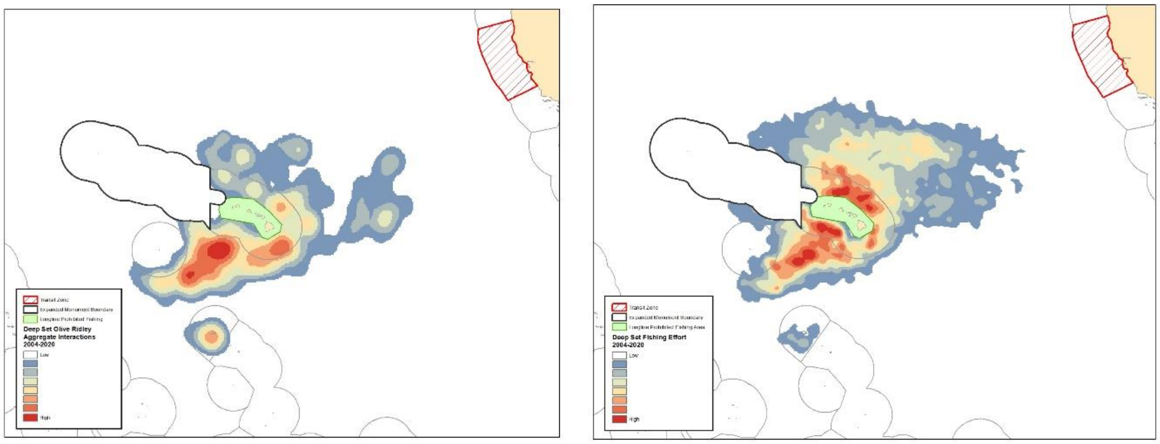

Figure 68. Heatmap comparison of observed olive ridley sea turtle interactions (left; *n* = 201) versus aggregated observed fishing effort during this same time frame (right; *n*= 65,245 from 2004 to 2020).

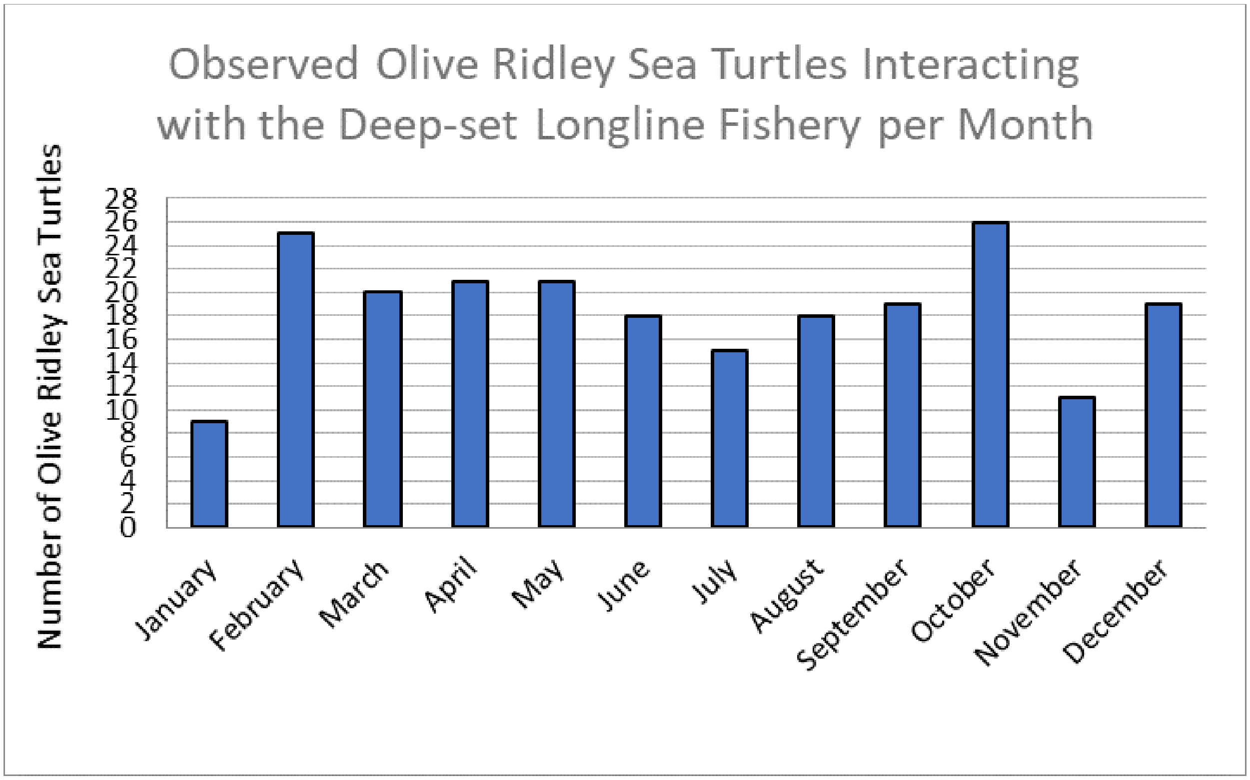

Figure 69. Numbers of observed olive ridley sea turtles captured per month in the DSLL fishery from 2004 to 2022 (*n* = 222).

Annual observed and estimated olive ridley sea turtle interactions peaked in 2016 with 30 observed and 160 estimated (McCracken 2019a) captures (Figure 70).

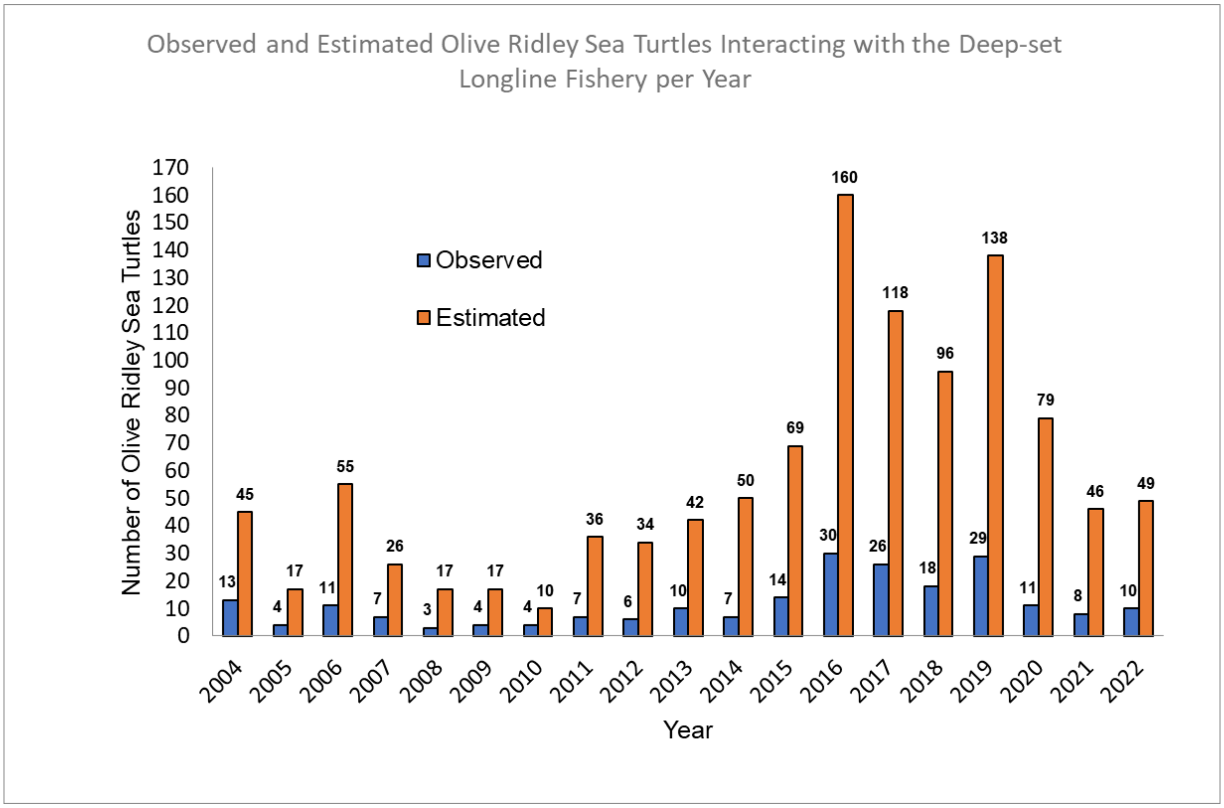

Figure 70. Olive ridley sea turtle interactions per year in the DSLL fishery between 2004 and 2022 for observed (blue bars) and estimated interactions (orange bars; McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a). Estimated interactions for 2022 were calculated using the expansion factor based on the observed captures and the observer coverage.

Estimates of the number of olive ridley turtles that are likely to be captured in the DSLL fishery each year are presented in Table 47 (estimates are from McCracken 2019a). On average, we would expect 71 olive ridley turtles to be captured each year in the DSLL fishery, although this number could be as high as 171 in any given year (McCracken 2019a). As with the turtles discussed previously, only a portion of these turtle captures would be reported by observers because the DSLL fishery has only 20% observer coverage (though this number dipped to 15.2% in 2020 due to restrictions associated with the COVID 19 pandemic). We also report the maximum 5-yr running average and the cumulative number of captures over 5 years (Table 47).

Based on the genetic data, we would expect 52 of the 71 olive ridley turtles that might be captured in the DSLL fishery in an average year to represent eastern Pacific nesting aggregations, both the endangered and threatened populations. The remaining 19 would be from the Western Indo-Pacific nesting aggregations. In a year in which the number of olive ridley turtles captured are as high as 171, we would expect 125 of those turtles to be from the eastern Pacific aggregations and 46 from the Western Indo-Pacific aggregations.

As noted in the introduction to Section 4.3, hooks per year in the DSLL have been increasing, specifically at a rate of ~1,802,000 hooks per year (95% CI: 1,557,699 to 2,046,303; Figure 35) and this rate of increase is expected to continue into the future. We found a statistically significant relationship between observed hooks per year and observed olive ridley sea turtles captures per year with $R^2 = 0.20$ and $P = 0.03$ (Figure 71a). As noted in Section 2 - Status of Listed Species, both the threatened East and West Pacific olive ridley sea turtles populations as well as the endangered Mexico population are at least stable with some nesting aggregations showing increasing trends. Hence the increasing trend in captures may be related to increasing densities of olive ridley sea turtles in the Action Area and not just increasing numbers of hooks set by the fishery. We examine this by looking at the trend in CPUE for observed olive ridley sea turtles where we find a similar increasing trend to our analysis of observed captures and observed hooks although this relationship is not statistically significant (Figure 71b; $R^2 = 0.05$; $P = 0.18$). This relationship is a per-1,000 hook number of captures and is therefore not influenced by the increasing numbers of hooks in the fishery.

This suggests the trend is due to either increasing abundances of olive ridley sea turtles or other environmental factors that are increasing the probability of capture. Our CPUE regression indicates that we anticipate an additional 4.5×10^{-5} (or 0.000045) observed olive ridley sea turtle captures per year per 1,000 hooks (CI: 0 to 1.1×10^{-4}). Given the anticipated increase in hooks per year for the DSLL, this equates to an additional 0.08 (CI: 0.07 to 0.09) observed olive ridley sea turtles per year. In 10 years this would equate to an additional 0.8 (CI: 0.7 to 0.9) observed olive ridley sea turtle captures attributable to the increasing hooks if the rate of increase continues. Given all of the uncertainties associated with factors that may be driving the trend in CPUE for olive ridley sea turtles, and the low number of additional observed olive ridley sea turtles expected to be captured as a result (approximately one over 10 years), at this time we are not adjusting our anticipated future captures for the fishery based on the expected increase in hooks per year but acknowledge that captures will need to be closely monitored to assess if the fishery's impact to the population is greater than we have analyzed here.

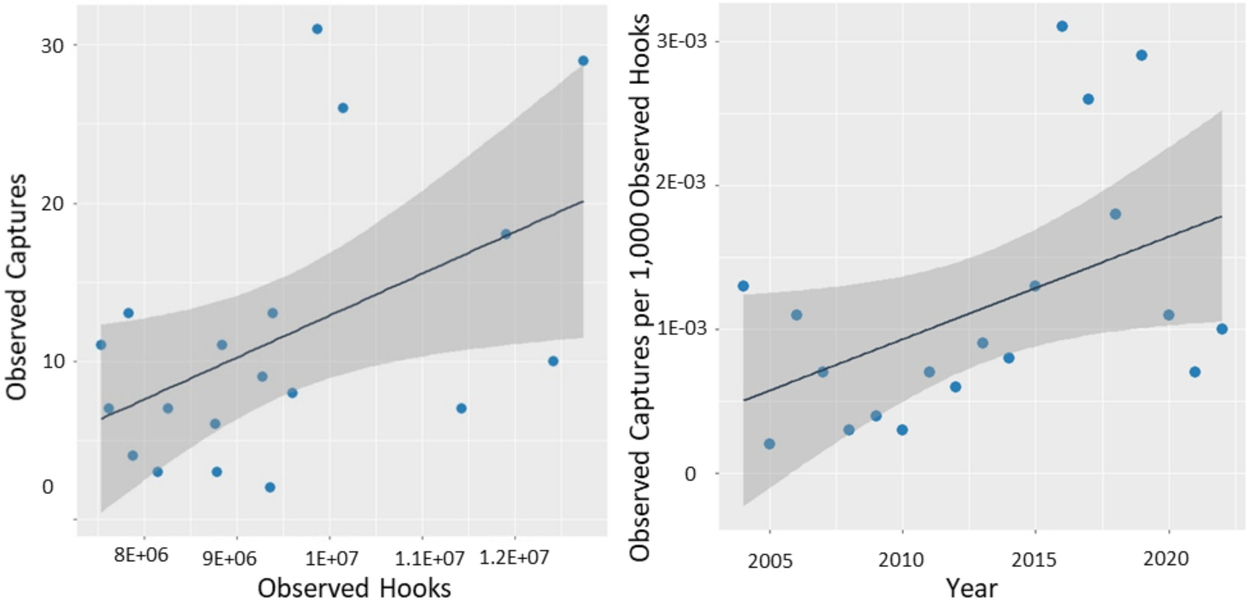

Figure 71. Relationships between observed hooks and observed olive ridley sea turtle captures in the DSLL. Left is the relationship between the number of observed olive ridley sea turtles captured per year and the number of observed hooks per year (blue dots). Right is the relationship between the annual number of olive ridley sea turtles captured per 1,000 hooks (CPUE) and year (blue dots). In both plots, the solid black line is a linear regression and the shaded area represents the 95% CI. See the text for the regression statistics.

Table 47. Number of olive ridley sea turtles expected to be encountered (exposed) in the DSLL per year (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a) and the maximum 5-yr running average and running sum based on observed and estimated captures from 2004 to 2022.

	DSLL Encounters	
Rate	**Mean**	**95th Percentile**
Annual	71	171
Max 5-Yr Running Average	118.4	N/A
5-Yr Running Sum	592	N/A

Response Analysis

If the patterns found in the observer data from 2004 to 2021 are representative, 99.1% of the olive ridley turtles captured in the fishery would be hooked rather than entangled (95% CI = 96.8% to 99.8%). About 91.0% (95% CI = 86.5% to 94.1%) of olive ridley turtles captured in the

DSLL fishery have been dead when they were hauled back to the fishing vessel; this assessment assumes these percentages are applicable to future captures.

We used the post-interaction mortality rates of Ryder et al. (2006) for hard-shell sea turtles to estimated post-interaction mortality of olive ridley sea turtles that were alive when they were captured. Table 48 shows the number of olive ridley sea turtles in each injury category and the corresponding post-interaction mortality rate for the 20 observed olive ridley sea turtles released alive. When at-vessel mortality and post-release mortality are combined, the effective mortality rate for olive ridley turtles captured in the DSLL fishery would be 92.1% (95% CI = 87.7 to 95.0%).

At this mortality rate, if 71 olive ridley turtles are captured in the fishery in an average year, about 65 of them are likely to die as a result of being captured (95% CI = 62 to 67; Table 49); if 171 of these turtles are captured, we would expect about 157 of them to die as a result of their capture (95% CI = 150 to 162). About 48 (95% CI = 46 to 49) of the olive ridley turtles that are likely to die in an average year would originate from eastern Pacific nesting aggregations with 17 (95% CI = 17 to 18) originating from Western Indo-Pacific aggregations. In a year that approximates the high estimate of 171 captures, about 115 (95% CI = 110 to 119) of the olive ridley turtles that are likely to die would originate from eastern Pacific nesting aggregations with another 42 (95% CI = 40 to 43) originating from the Western Indo-Pacific. For the 5-year running average, we would expect that 80 (95% CI = 76 to 82) of the olive ridley sea turtles that are expected to die would originate from the eastern Pacific nesting aggregations with another 29 (95% CI = 28 to 30) originating from the western Indo-Pacific. Those likely to die from the eastern Pacific nesting aggregations would include both the endangered and threatened populations.

Table 48. Number of olive ridley sea turtles released alive in the DSLL fishery between 2004 and 2022 by their injury and release condition. Numbers in parentheses are the corresponding post-interaction mortality rates from Ryder et al. (2006).

	Release Condition			
Injury Category	**Line < 1/2 SCL**	**Line ≥ 1/2 SCL**	**No Gear**	**Grand Total**
Entangled; Comatose	-	-	2 (0.60)	2
External	1 (0.10)	-	13 (0.05)	14
Insertion Not Visible	-	-	-	-
Insertion Visible	-	-	4 (0.10)	4
Jaw	-	-	-	-

Table 49. Number of olive ridley sea turtles that are likely to die as a result of being captured in the DSLL each year based on the mean of the effective mortality rate (93%). Note that there is no 95th Percentile associated with the 5-year running average and sum.

	Anticipated mortalities based on the mean (93%) mortality rates	
Rate	**Mean**	**95th Percentile**
Annual	65	157
5-Year Running Sum	109	N/A
5-Year Running Average	545	N/A

Risk Analysis

As discussed above, there were 222 observed interactions with olive ridley sea turtles in the DSLL from 2004-2022, with an estimated total of 1,104 interactions. NMFS predicts that the DSLL fishery will interact with a mean of 71 (95th percentile: 171) olive ridley sea turtles per year with a maximum 5-year running average of 118.4 and up to 1,184 turtles over 10 years. We estimate a mean mortality rate of 92.1% (95% CI: 87.7% to 95.0%), resulting in a mean of 65 (95th percentile: 157) mortalities. The maximum 5-year running average indicates that up to 109 turtles will die annually on average with up to 1,090 over 10 years. Over a 40 year time horizon, if captures remain consistent with historical numbers we would anticipate a cumulative total of 4,736 captures with 4,361 mortalities.

About 48 of the 65 olive ridley turtles that are likely to die in an average year would originate from either the endangered or the threatened East Pacific population with 17 originating from West Pacific population. In a year that approximates the high estimate of 171 captures, about 115 of the 157 olive ridley turtles that are likely to die would originate from either the endangered or the threatened East Pacific population Pacific nesting populations with another 42 originating from the West Pacific population. For the maximum 5-year running average, out of the 109 olive ridley sea turtles expected to die, about 80 would originate from either the endangered or the threatened East Pacific population, , with 29 originating from West Pacific population. Over a 40 year time horizon, if captures remain consistent with historical numbers we would anticipate a cumulative total of 3,471 captures with 3,197 mortalities for olive ridley sea turtles originating from East Pacific populations (both endangered and threatened) and 1,165 captures with 1,179 mortalities for the West Pacific population.

East Pacific Populations (Threatened and Endangered)

As noted in the Status of Listed Resources section, the endangered Mexico population of olive ridley sea turtles is comprised of more than 1,000,000 nesting females (Abreu-Grobois and Plotkin 2008; Valverde et al. 2012; NMFS and FWS 2014). The portion of the threatened population that nests in the East Pacific counts of adult olive ridley sea turtles on eastern Pacific nesting beaches similarly number at more than 1,000,000 nesting females on beaches from Costa Rica and Nicaragua (Chaves et al. 2005 as cited in NMFS and FWS 2014; Valverde et al. 2012; NMFS and FWS 2014).

Assuming total abundance of olive ridley sea turtles for the endangered population is a minimum of one million individuals, and we apply all captures from the East Pacific to the endangered population, annual mortalities of 48 (95th percentile: 79) olive ridleys would correspond to 0.0048% to 0.0079% of the total population. When we consider 868 interactions and 799 mortalities over the next 10 years, the rate of interaction with the fishery would be 0.087% and 0.080% of the total population for the endangered Mexico populations. As this population is at least stable (NMFS and FWS 2014), these percentages of the population being impacted will not increase over time. Therefore, captures between 30 and 40 years into the future will impact no more than 0.087% (captures) and 0.080% (mortalities) of the total population.

Similarly, the threatened portion of the East Pacific populations is considered to number more than 1 million individuals with a population trend that is at least stable. Therefore, the numbers we provided above for the endangered portion of the population apply to the threatened portion of the East Pacific population. We anticipate that captures between 30 and 40 years into the future will impact no more than 0.087% (captures) and 0.080% (mortalities) of the total population. We consider the implications of this effect on both the threatened and endangered populations of olive ridley sea turtle that nest in the eastern Pacific in the Integration and Synthesis section of this opinion.

West Pacific Populations

As noted in the Status of Listed Resources section, in the Western Pacific, the largest remaining breeding aggregations are in Northern Australia where it’s estimated that at least 5,000 females nest annually (Commonwealth of Australia 2017). Considering remigration intervals, we estimate this population at 10,000 adult females, assuming a 50% sex ratio, this would equate to 20,000 adults. When added to a well-studied nesting aggregation in Alas Purwo National Park, East Java, Indonesia, which is currently increasing; suggests a minimum adult abundance of 24,400 for West Pacific olive ridley sea turtle populations. We estimate the West Pacific population to have a minimum of 316,883 juveniles and adults based on an adult population size of 24,400 with 7.7% of the population occurring in that stage and excluding hatchlings (Maulany et al. 2017).

Based on the maximum 5-yr running average, we anticipate an average of 31.6 West Pacific olive ridley sea turtles to be captured in a year with 29.1 mortalities. These numbers would represent 0.010% of the abundance captured and 0.009% killed. Based on the maximum 5-yr running sum, we anticipate 316 captures over 10 years, with 291 mortalities. These numbers would represent 0.10% of the abundance captured and 0.092% killed over 10 years. As West Pacific abundance is at least stable, we expect these percentages to not increase over our 40 year analytical timeframe. We consider the implications of this effect on the West Pacific olive ridley sea turtle in the Integration and Synthesis section of this opinion.

4.3.8 Sperm Whale

Exposure Analysis

Evidence from other fisheries demonstrates that sperm whales are attracted to longline fishing gear and remove bait and catch (Ashford et al. 1996; Hill et al. 1999; Purves et al. 2004; Kock et al. 2005). Ashford et al. (1996) reported sperm whales that trailed fishing vessels for days while Crespo et al. (1997) documented sperm whales depredating catch in the Southern Ocean. In Subarea 48.3 of the Convention for the Conservation of Antarctic Marine Living Resources region, sperm whales were recorded during 24% of hauling operations (WG-FSA 2003 Report). After examining the heads, lips, and jaws of sperm whales, Kock et al. (2005) concluded that sperm whales strip toothfish off lines during soaks. Observer data from Chilean waters adjacent to the Convention Area suggests that about 3% of toothfish are taken from the line by sperm and killer whales (WG-FSA-03/95). As we discussed in the Status of the Listed Resources section of this opinion, observers aboard longline vessels fishing for sablefish and halibut in the Gulf of Alaska have also reported sperm whales feeding on catch (Hill and Demaster 1998).

Between 2004 and 2022, only one sperm whale has been observed interacting with the Hawaii DSLL fishery and this occurred in 2011. Adjusting for the 20% observer coverage in the fishery, the actual number of sperm whales captured in the fishery in 2011 may have been as high as six (McCracken 2019a). Estimates of the number of sperm whales that are likely to be captured in the DSLL fishery each year are presented in Table 50 (estimates are from McCracken 2019a). Although these estimates are based on a single observed interaction and may initially appear questionable, it is important to note that another sperm whale was entangled in an experimental longline cruise testing gear configurations for this fishery in 2002 (NMFS unpublished data). Hence the possibility exists of sperm whale captures in any given year, particularly in the unobserved portion of the fishery.

Based on the models of McCracken (2019a), on average, sperm whales have slightly more than a 50% probability of being captured each year in the DSLL fishery, although the number of sperm whales captured in any given year could be as high as 3. Based on the single year estimate of six sperm whale, we estimate a maximum 5-year running average of 1.2 sperm whales per year with a maximum of 6 sperm whale captures over 5 years. As with all of the species discussed thus far, only a portion of these whale captures would be reported by observers because the DSLL fishery only has approximately 20% observer coverage.

Table 50. Number of sperm whales expected to be encountered (exposed) in the DSLL fishery per year (McCracken 2019a; NMFS 2018b) and the maximum 5-yr running average and running sum from 2004 to 2022.

Rate	**Mean**	**95th Percentile**
Annual	0.6	3
5-Yr Running Average	1.2	N/A
5-Yr Running Sum	6	N/A

Response Analysis

The sperm whale captured in the 2002 experimental longline cruise was entangled rather than hooked, while the capture condition of the Hawaii DSLL-caught sperm whale in 2011 is unclear. The sperm whale captured in the experimental longline cruise was released with no line attached. It was not clear whether the sperm whale captured in 2011 had any remaining gear attached after our review of the raw observer data. For large whales where the severity of an injury cannot be determined, NMFS (2023b) guidance applicable to an interaction with a fishery is to assume an entanglement and use the proportion of 0.75 to assign an injury to the M/SI category (i.e. 75% of injuries are M/SI). McCracken (2019a) applied this ratio to the estimated annual and 95th percentile captures from Table 50, suggesting that a mean of 0.45 or up to 2 mortalities or serious injuries may occur to sperm whales annually. We note that the proportion of 0.75 used by NMFS (2023b) is based on severe entanglement events that caused health derogation or death of the individual. At this time, PRD is not aware of any photographic evidence to suggest DSLL gear remains on sperm whales that may interact with this fishery. We therefore do not think it is likely that 75% of sperm whales interacting with the fishery are likely to sustain M/SI.

While sperm whales heavily depredate other NMFS fisheries and present global longline fishery concerns (e.g., see Hanselman et al. 2018) we do not have enough data to determine mortality for the species in this fishery. Muto et al. (2021) report an annual M/SI of 3.3 animals for NMFS authorized fisheries in the North Pacific (various Alaska fisheries), although they indicate that no data exists for the DSLL. Muto et al.'s (2021) results were based on reported incidents from unobserved sets between 2014 and 2018 in the Alaskan fisheries. Donoghue et al. (2002) reported a sperm whale that died while entangled in longline gear in an artisanal fishery off Chile.

Although most whales that have become entangled in longline gear have either disentangled themselves or were disentangled with help, some whales have died as result of entanglement. The probable consequences of capturing an average of 1 whale every two years or up to 3 sperm whales in a year, will likely result in those whales being entangled with smaller risks of them dying as a result of their interaction. Currently, we have no observer data to suggest that any sperm whales have died in this fishery and given their size and the rare occasions they are observed interacting with the fishery we do not anticipate any mortalities. Sperm whales are large enough to part the mainline and depredate catch with efficiency. Often, when mainlines are broken, the crew is unaware of what actually caused the line to break and aren't always due to whales. Mainlines can part for various reasons in an open ocean environment. However, if whales are observed diving on the line, inferences can be made that interactions have occurred, especially when depredated bait begins to appear on the vessel. Often, depredated bait can be attributed to specific predatory species based on the bite marks and tooth indentations on the carcass(es). Whales will remove multiple fish in series as they let the line flow through the corner of their mouths.

Behavioral modifications may occur once an animal interacts with the fishery and could result in an animal either avoiding the gear, or being attracted to it. Once sperm whales learn how to effectively depredate bait and catch from lines, they may in fact increase their potential exposure to the fishery by following vessels during retrieval. Which is the case in the NMFS authorized Alaskan longline fisheries (particularly the sablefish fishery). However, there are operational differences between this fishery and the sablefish fishery, target catch and depths, and the species' abundance varies greatly between these fisheries and geographical locations. Given the small number of expected interactions, the species and fisheries distributions, target catch, and greater spatial flexibility of the DSLL fleet, it is highly unlikely sperm whales will associate vessels as a food source. We are not aware of any records from the fleet, or from observer records, that sperm whales are following DSLL vessels and diving on the line.

Risk Analysis

Sperm whale abundance in the North Pacific Ocean has been estimated at between 26,300 and 32,100 whales (Barlow and Taylor 2005). We estimate an average interaction rate of 1 whale every two years (up to 3 per year). Over a 40 year time horizon we anticipate a cumulative total of 48 interactions, but anticipate that none will die as a result. Our assessment illustrates that the DSLL fishery interacts with a very small percentage of sperm whales in the North Pacific Ocean. We anticipate an annual average of up to 0.002% will be incidentally captured and none will be killed. This level of effect will not change the status or trend of sperm whales in the North Pacific Ocean. We consider this level of effect on sperm whales as a species in the Integration and Synthesis section below.

4.3.9 Main Hawaiian Island Insular False Killer Whale

Exposure Analysis

No MHI IFKWs have been observed caught in the DSLL fishery between 2004 and 2022 (NMFS Observer data). Therefore, we use interaction data for all false killer whales around Hawaii to estimate MHI IFKW interactions, as the best available information. This is consistent with how we estimate M/SI of MHI IFKW under the MMPA (McCracken 2019a, McCracken and Cooper 2021, 2022). False killer whales around Hawaii are comprised of three MMPA recognized stocks, the NWHI and pelagic stocks, which are not listed under the ESA, and the endangered MHI IFKW. Total captures and captures that result in M/SI of false killer whales by the DSLL inside and outside of Hawaii EEZ are estimated annually based on observed interactions (McCracken 2019a; McCracken and Cooper 2021, 2022). These estimates are used to assign total captures and captures that result in M/SI, the latter of which are reported in annually-updated MMPA Stock Assessment Reports (SARs; Caretta et al. 2020, 2021, 2022). In these estimates, observed captures reported as 'blackfish' within the Hawaii EEZ are attributed to false killer whales. These estimated interactions are proportioned among the three false killer whale stocks based on the number of fishery sets made within the range of each stock, the estimated density of each stock and the relative density of each stock in areas where they overlap. We applied the same methods used to proportion total M/SI captures to the three stocks (Caretta et al. 2020, 2021, 2022) to proportion total captures and estimate the potential annual captures of MHI IFKW (Table 51) and to calculate the maximum 5-yr running average and sum. The data for 2020 and 2021 are presented in the 2022 SAR which is not yet publicly available (NMFS In Review; Table 51).

A Take Reduction Team was established under the MMPA in January 2010 (75 FR 2853) for false killer whale interactions with the Hawaii longline fisheries. The resulting Take Reduction Plan, which became effective on December 31, 2012 (77 FR 71260); included the elimination of the seasonal contraction of the Longline Exclusion Zone (LLEZ) around the MHI which reduced the overlap between the fishery and the MHI IFKW to that shown in Figure 72 which has been consistent since 2013. Adjustments to bycatch estimation methods were implemented for 2013 to account for changes implemented by the TRP to reduce the false killer whale serious injury rate (McCracken 2015). We therefore focus our analysis on estimated captures from 2013 to 2021. While we know that no false killer whales were observed captured in the overlap area in 2022, estimates for the number of MHI IFKW potential captures for 2022 are not yet available (see below for a discussion on the calculation of captures).

Currently, approximately 5% of the DPS' range overlaps with the Hawaii DSLL fishery (see dashed area in Figure 72), accounting for approximately 4,223 square miles where interactions with the DPS are still possible, though their probability is low. From 2013 to 2021, a total of 343 sets have been made by the DSLL in this area and 22 of them have been observed (Table 51) for an average of 8.2% observer coverage (range 0% to 23%).

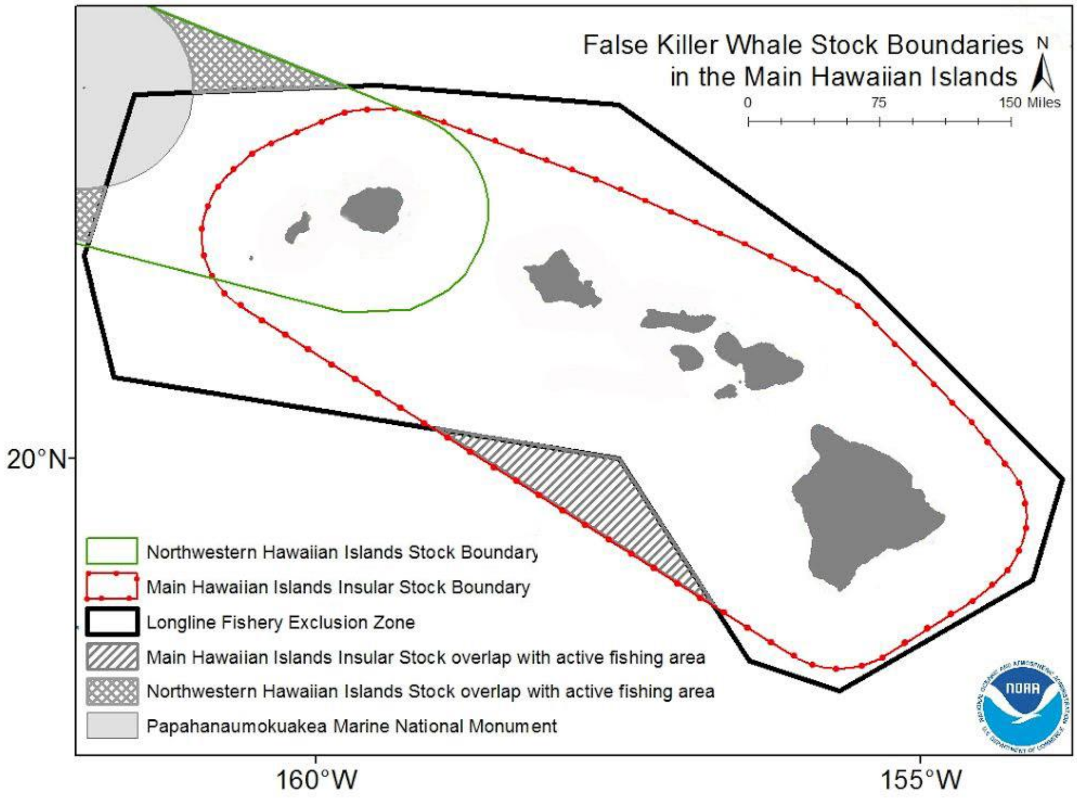

Figure 72. Area of overlap (i.e., the triangle area south of the MHI) between the MHI IFKW and the DSLL fishery. The red line is the MHI IFKW's range around the MHI, the SEZ is to the south and not projected, however this overlap zone occurs within the SEZ boundaries (NMFS 2021c).

Table 51. Summary of DSLL effort, observer coverage and estimated captures for MHI IFKW. Captures are from the MMPA SARs (Caretta et al. 2020, 2021, 2022; NMFS unpublished data).

Year	**Total Sets in Overlap**	**Observed Sets in Overlap**	**% Observer Coverage**	**Estimated MHI IFKW Captures (using MMPA SARs methods*)**	**Estimated MHI IFKW Captures with M/SI (from MMPA SARs*)**
2013	8	0	0.00%	0.003	0.002
2014	13	3	23.08%	0.013	0.012
2015	22	2	9.09%	0.000	0.000
2016	20	0	0.00%	0.010	0.008

Year	Total Sets in Overlap	Observed Sets in Overlap	% Observer Coverage	Estimated MHI IFKW Captures (using MMPA SARs methods*)	Estimated MHI IFKW Captures with M/SI (from MMPA SARs*)
2017	63	9	14.29%	0.070	0.054
2018	94	7	7.45%	0.106	0.094
2019	5	1	20.00%	0.013	0.011
2020	14	0	0.00%	0.008	0.007
2021	104	0	0.00%	0.230	0.198

*Data for 2020 and 2021 are not yet published in a SAR.

Table 52 contains NMFS' best estimates of interactions with MHI IFKW in this fishery (McCracken 2019a). These estimates suggest that the annual probability of capturing a MHI IFKW is small, and these small probabilities accumulate enough to suggest that on average, 1 capture in 23 years is expected. But capture may be more frequent as indicated by the 95th percentile. Based on the proration attributing estimated incidental captures of false killer whales in the DSLL to MHI IFKW (Caretta et al. 2020, 2021, 2022) from 2013 to 2021 (Table 51), the maximum 5-yr running average is 0.085 and the maximum 5-yr running sum is 0.427. These numbers imply that about one MHI IFKW capture in 12 years is expected. This capture level is higher than the estimate of McCracken (2019a) because of the higher estimated captures in 2021 of 0.23, an order of magnitude higher than previous years (Table 51).

Table 52. Mean and 95th percentile of MHI IFKW posterior anticipated captures of MHI IFKW in the Hawaii DSLL fishery annually and over 5 years (McCracken 2019a). The maximum 5-yr running average and sum are based on the capture estimates from 2013 to 2021 in Table 51.

Period	Mean	95th Percentile
Annual	0.043	0.130
5-year sum	0.214	0.414
Maximum 5-yr Running Average	0.085	N/A
Maximum 5-yr Running Sum	0.427	N/A

Response Analysis

There is evidence, based on scarring and mouth and dorsal fin disfigurements, that there are interactions between MHI IFKW and unidentified hook-and-line fisheries (Baird and Gorgone 2005; Baird et al. 2014; Beach 2015; Baird et al. 2017a; Baird et al. 2023). There are also differences in these injury rates between the MHI IFKW social clusters, suggesting that different social clusters may have different habits that lead to different rates of exposure to fisheries. Certain behaviors, like depredation and following fishing vessels, may be learned behaviors that are passed down to individuals within a social cluster. These types of behaviors may affect the growth rate of the social clusters differently (Sargeant and Mann 2009; Beach 2015). It is not clear what fisheries are responsible for these injuries and whether or how much these injuries are affecting the fitness of MHI IFKW individuals. Interactions between this fishery and false killer whales are well documented for the pelagic stock of false killer whales, and in most cases interactions between pelagic false killer whales in this fishery are considered a serious injury.

The prevalence of physical evidence of hooking or entanglement with MHI IFKWs along with the bias of injuries toward females suggests that fishery injuries may be particularly problematic for this DPS. Nearshore commercial and recreational fisheries are believed to be the primary source of these injuries. Interactions with the Hawaii DSLL fishery, while likely rare, cannot be ruled out. Baird et al. (2017a) found that 23.3% (17 out of 73 individuals with >50% of mouthline visible in photos of acceptable quality) of adults have mouthline injuries consistent with fisheries interactions (10 females, 4 males, 3 unknown), and 9.1% (15 out of 165 distinctive individuals with good quality photos) of MHI IFKWs have dorsal fin injuries. These results show an increase from earlier studies in which Beach (2015) reported 22% (16 out of 72 individuals) with mouthline injuries consistent with fisheries interactions, and Baird et al. (2014) reported 7.5% of MHI IFKWs (12 out of 160 individuals) with dorsal fin injuries.

Interaction rates based on mouthline injuries are likely negatively biased because only 58% of individuals are able to be observed with ≥50% of their mouthline visible (Beach 2015), so some mouth-line injuries are likely concealed below the water during observation and data collection. However, this rate of scarring combined with the location density map (Figure 28) and other evidence, like hook types that have been recovered in the stomach of a small number of whales, suggest that Hawaii State managed fisheries are the predominant cause of these injuries.

Even so, we are reasonably certain that the Hawaii DSLL fishery will interact with MHI IFKW, albeit infrequently (see Table 53), and those interactions are expected to result in mortality or serious injury. Based on the estimates in Table 51 and Table 53, we would expect that the Hawaii DSLL fishery would kill or seriously injure one MHI IFKW approximately every 30 (mean) years (Table 53). Based on the maximum 5-yr running average, we would expect that the Hawaii DSLL fishery would kill or seriously injure one MHI IFKW approximately every 14 years (Table 53). Again the discrepancy between the mean value of McCracken (2019a) and the maximum 5-yr running average is due to the high estimated captures from 2021.

Table 53. Mean and 95th percentile of MHI IFKW posterior anticipated mortality and serious injury levels in the Hawaii DSLL fishery annually and over 5 years (McCracken 2019a). The maximum 5-yr running average and sum are based on the M/SI values from 2013 in Table 51.

Period of ATL	Mean	95th Percentile
Annual	0.033	0.102
5-year	0.166	0.329
Maximum 5-yr Running Average	0.073	N/A
Maximum 5-yr Running Sum	0.364	N/A

Risk Analysis

The MHI IFKW is comprised of a single small population, which places it at greater risk than a species that is comprised of multiple dispersed populations. As noted in the Status of the Listed Resources, MHI IFKW abundance is estimated to be 138 (95% credible interval [CRI]=120–160) animals in 2021 with an estimated annual trend of -3.51 (95% CRI: -8.40 to 2.04) over the entire time series considered (1999 to 2021) and -5.53 (95% CRI: -9.91 to -1.61) over the past 10 years.

As discussed above, the MHI insular population has a high rate of dorsal fin disfigurements and mouthline injuries consistent with injuries from fishing lines (Baird and Gorgone 2005; Baird et al. 2014, 2017; Beach et al. 2015; Baird 2019). We note, the fisheries which are responsible for these injuries (whether commercial or recreational) are unknown. While the DSLL fishery is the only federally managed commercial fishery which overlaps with the DPS' range, the overlap area is small. Whereas the DPS' range is coextensive with Hawaii state fisheries including State commercial fisheries that use hook types similar to those used in the DSLL fishery. Even so, we know that false killer whales are one of the most frequently hooked or entangled cetaceans in tuna-targeting longline sets (Forney and Kobayashi 2007; McCracken and Forney 2010). When longline fisheries within the MHI IFKW/pelagic overlap zone (see Figure 72) hook or entangle a false killer whale in this area, genetic samples or photographs are needed to determine what DPS or stock was impacted and NMFS prorates the interaction to either the pelagic or MHI insular population based on population densities.

McCracken (2019a) estimates a mean of 0.033 (95th percentile: 0.102) whales will sustain serious injuries or be killed per year; over 5 years, 0.214 whales are expected to sustain serious injuries or be killed. Based on the methods used in the MMPA SARs (Caretta et al. 2020, 2021, 2022), we estimate a maximum 5 yr running average of 0.073 M/SI per year. However, we acknowledge that these estimates are based on inference as there have been no observed interactions between the DPS and the DSLL fishery.

Assuming total abundance for the MHI IFKW in 2021 was 138, a 5.53% per year decline indicates that the population abundance in 2022 is 131. Assuming a mean of 0.043 (95th percentile: 0.13) individuals from this DPS may interact with the DSLL fishery, this would equal

0.033% (95th percentile: 0.10%) of the DPS' total population. The mortality (assuming all M/SI result in mortality) of 0.033 (95th percentile: 0.10) individual MHI IFKWs assuming total abundance equals 131 represents 0.025% (95th percentile: 0.078%) of the total population. For the maximum 5-year running average, 0.085 captures and 0.073 mortalities represents 0.065% of the DPS' population captured and 0.056% of the population killed. The maximum 5-year running sum of 0.43 MHI IFKWs suggests that over 10 years we would anticipate 0.85 total captures, of these, 0.73 would be expected to die. This level of interaction would represent 0.65% of the population being impacted and 0.56% of the population being killed.

If we assume 0.073 mortalities per year, and we remove the mortalities from the fishery, from 2022 to 2023, the rate of population decline would be reduced from 5.53% per year to 5.47% per year. From 2022 to 2062 the rate of population decline would be reduced to 5.33% per year. Given the estimated standard deviation for the population growth rate (0.018), and based on a power analysis for detecting differences in means with the same variance (Cohen 1988), more than 1,400 years of data would be required to detect a statistically significant difference in these values (see Appendix C for details on this analysis). While these potential reductions in population growth rates are small, they may have severe consequences for a very small population with a decreasing trend. The mortality of one individual from a population size of 138 represents 0.72% of the population. We consider the implications of this effect on MHI IFKW in the Integration and Synthesis section of this opinion.

Because the observer coverage of the overlap area between the MHI IFKW and the DSLL is lower than the fishery in general (8% compared to 20%) we believe it critically important that NMFS take action to confirm the data it relies upon to make decisions for this DPS are reliable and increase observer coverage if the data are not reliable. The MHI IFKW are comprised of a single small population with a declining population growth rate, which means it has an inherently higher risk to threats that can appreciably affect the DPS' patterns of population growth; therefore a reasonable action to improve the agency's understanding of the effect of this fishery in the overlap zone is to increase monitoring in this area.

4.4 Cumulative Effects

"Cumulative effects", as defined in the ESA implementing regulations, are limited to the effects of future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area considered in this opinion (see 50 CFR 402.02). For an action to be considered reasonably certain to occur, it must be based on clear and substantial information, or otherwise have a firm basis to support a conclusion that a consequence of an action is likely. Some factors we consider when evaluating an action for potential cumulative effects and whether those effects are reasonably certain to occur include our past experiences from similar actions, existing plans for the activity or action, and hurdles, like economic and legal requirements, that must be met before the action can go forward (see 50 CFR 402.17). Future federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

NMFS searched for information on future State, tribal, local, or private actions that were reasonably certain to occur in the Action Area. Most of the Action Area is outside of territorial waters of the United States of America, which would preclude the possibility of future state, tribal, or local action that would not require some form of federal funding or authorization. NMFS conducted electronic searches of business journals, trade journals, and newspapers using Google scholar, WorldCat, and other electronic search engines. Those searches produced no evidence of future private action and their effects in the Action Area that would not require federal authorization or funding and is reasonably certain to occur.

While we considered various state managed vessel-based fisheries which exist in Hawaiian waters, we do not believe they will overlap in geographical space for fishing activities and would only overlap the vessel paths from this fishery when they transit to Hawaiian ports, and we consider the probability of exposure to impacts from transiting vessels by the ESA-listed resources considered in this biological opinion to be discountable. The same could be said for recreational boating around the MHI as well. The primary effects we would expect from State fisheries and recreational boating, would include injury and mortality from ship strikes and fishing, as well possibly changes in local prey numbers and distribution. NMFS is not aware of any other actions that are likely to occur in the Action Area during the foreseeable future.

5 INTEGRATION AND SYNTHESIS

The Status of the Listed Resources, Environmental Baseline, and Cumulative Effects described the pre-existing condition of the listed species globally and within the Action Area given the effects of activities such as commercial fisheries, direct harvests and modification or degradation of habitat caused by marine debris and climate change. The pre-existing condition of these species in the Pacific serves as the point of reference for our conclusions. The Effects of the Action section of this biological opinion describes the direct and indirect effects of the continued authorization of the DSLL fishery, as currently managed. NMFS approved the Pelagic FMP in 1987 and established the Federal longline permit and logbook reporting requirements in 1991.

This section of this biological opinion recapitulates, integrates, and synthesizes the information that has been presented thus far to evaluate the risks that continuing the DSLL fishery poses to Central North Pacific green sea turtles, Central South Pacific green sea turtles, Central West Pacific green sea turtles, East Indian – West Pacific green sea turtles, East Pacific green sea turtles, Southwest Pacific green sea turtles, leatherback sea turtles, North Pacific loggerhead sea turtles, the Mexico breeding population of olive ridley sea turtles and threatened populations of olive ridley sea turtles, Indo-West Pacific scalloped hammerhead sharks, oceanic whitetip sharks, giant manta rays, sperm whales, and MHI IFKWs in the Pacific Ocean.

The "risks" this section of the opinion considers are (1) increases in the extirpation/extinction probability of particular populations and of the species as they have been listed; and (2) reductions in their probability of being conserved (that is, of reaching the point where they no longer warrant the protections of the ESA). These two probabilities correspond to the species' likelihood of surviving in the wild (that is, avoiding extinction) and their likelihood of recovering in the wild (that is, being conserved). Our analyses give equal consideration to both probabilities; however, to satisfy the explicit purposes of the ESA and NMFS' obligation to use its programs to further those purposes (16 US.C. 1536(a)(1)), a species' probability of being conserved has greater influence on our conclusions and jeopardy determinations. As part of these analyses, we consider the action's effects on the reproduction, numbers, and distribution of each species.

Our analyses find that the proposed action, while it results in the death of individual endangered North Pacific loggerhead sea turtles, leatherback sea turtles; East Pacific green sea turtles, Central North Pacific green sea turtles, East Indian-West Pacific green sea turtles, Central West Pacific green sea turtles, Southwest Pacific green sea turtles, Central South Pacific green sea turtles, endangered Mexico breeding population of olive ridley sea turtles, and threatened (other) populations of olive ridley sea turtles, threatened Indo-West Pacific scalloped hammerhead shark, oceanic whitetip shark, and threatened giant manta, it has very small effects on the dynamics of the populations those individuals represent or the species those populations comprise. We acknowledge that the action has the potential to cause mortality or serious injury to individual MHI IFKW but again, with the rarity of these events, this would have a very small effect on the population dynamics of this DPS. We do not expect mortalities for sperm whales.

We explain the basis for this conclusion for each species in the following sections. These summaries integrate the results of the exposure, response, and risk analyses we presented earlier in this biological opinion with background information from the Status of the Species and Environmental Baseline sections of this biological opinion to assess the effect that the Hawaii-based DSLL fishery is likely to pose to endangered and threatened individuals, the population or populations those individuals represent, and the "species" as it was listed pursuant to the ESA of 1973, as amended.

5.1 Giant Manta Ray

The Hawaii-based DSLL fishery is expected to capture an average of 23 (95th percentile= 68) threatened giant manta ray each year. However, because this is an average estimate in some years, higher or lower numbers of giant manta ray may be incidentally captured. On average, 10 of the manta rays captured are likely to die as a result of their capture. Over a 40 year analytical time frame we would anticipate a cumulative total of 1,588 captures and 678 mortalities.

As described in the Environmental Baseline, giant manta ray are currently effected by several stressors within the Action Area, including climate change, fisheries, vessel strikes, and marine debris; however both direct harvest and bycatch in fisheries is the dominant stressor in the baseline that affects the species. The Hawaii SSLL fishery is estimated to capture up to 13 giant manta rays annually with up to four mortalities (NMFS 2019a). United States fisheries that operate out of the West Coast regions are not known to capture giant manta ray. International fisheries also capture and kill giant manta ray within the Action Area.

The number of manta populations exposed to domestic longline fisheries and the size of those populations is unknown; however, both determine the effect of fishery-related incidental captures and mortalities on giant manta ray populations. The species appears to have a population substructure that is composed of independent demographic units or subpopulations with small distinct home ranges. Nevertheless, giant manta ray populations appear to be regionally distinct (Beale et al. 2019, Lewis et al. 2015; Moazzam 2018, Stewart et al. 2016) and may have distinct home ranges (Stewart et al. 2016). The degree to which subpopulations are connected by migration is unclear but is assumed to be low so the decline of the small subpopulations may result in regional depletions or extinctions with the reduced possibility of successful recolonization (Marshall et al. 2018). For example, in Indonesia, gill net fisheries in the Lembeh Strait captured 1,424 manta rays (*Mobula* spp.) in a 10-month period (Cochrane 1997), resulting in an apparent local extirpation (D. Djalal and A. Doali, pers. comm. cited in Beale et al. 2019). Overall, in many regions, there is a declining trend for giant manta rays, with up to as much as 80% over the last 75 years, and >30% globally (Marshall et al. 2011). Additionally, declines have been noted to be up to 95% or even extirpation in some locations (Lewis et al. 2015; Miller and Klimovich 2017; 83 FR 2916), although we have no information on trends for giant manta ray in the Action Area.

To lay the foundation for our effects analysis, and because the preponderance of evidence suggests that giant manta rays occur in regionally distinct subpopulations with abundances of at least 1,000 individuals to be genetically viable (Frankham et al. 2014), we used the distribution of observed incidental captures combined with information from published literature to estimate the central locations of potential giant manta ray subpopulations that occur within the Action Area and are effected by the continuing operation of the DSLL fishery. This allowed us to identify three potential subpopulations that may be impacted by this fishery, Hawaii, Johnston Atoll, and Palmyra (Figure 42). To assess the potential effect of the incidental captures and mortalities on these giant manta ray subpopulations, we developed scenarios that projected the annual incidental captures over 10, 20, and 40 years.

Our analysis included estimates of the effects of the action considering two different intrinsic rates of population growth from published literature on giant manta rays, 0.019 (Rambahiniarison et al. 2018) and 0.042 (Ward-Paige et al. 2013). We consider these under two different scenarios. Similar to the analyses for every other species, Scenario 1 used the maximum 5-year running average of captures. We did consider a second scenario (Scenario 2) in which we focused on the 95th percentile value as this is the maximum number of incidental captures we anticipate in any one year. However, as we do not anticipate this level of capture every year, this scenario likely overestimates the impacts of the fishery on the species. Therefore, we consider Scenario 1 as our best estimate for the impacts to the species.

Because our subpopulation structure is estimated, and we do not know the size of these subpopulations, we examined the effect of the action on the viability of each subpopulation. To do this assessment we examined a range of initial subpopulation abundances based on values gathered from the literature (abundances were 60, 100, 200, 400, 600, 800, 1,000, 1,500, 1,875 or 2,464 individuals). As described above and in the Exposure Analysis section, we focused most of our attention on population sizes larger than 1,000 individuals as this is likely the minimum size of viable populations. The smaller population sizes, while in the literature, are likely a result of incomplete sampling of individuals. However, since they may also represent exploited populations that may be at higher risk of extirpation, each scenario includes subpopulation sizes of less than 1,000 individuals.

Results of Scenario 1 indicates that if initial abundance is greater than 600 individuals, under the proposed action all three subpopulations experience positive population growth out to the 40 year projection for $r_{max} = 0.019$. Our analysis (see Appendix B) shows that some putative subpopulations may start to experience declines or extirpation based on the initial size of the population under 400 individuals given the anticipated take levels (see numbers in red in Appendix B). Specifically, if initial population sizes are 100 or fewer individuals, all three putative subpopulations experience substantial declines (defined as greater than 5%) or extirpation within 10 years and by 40 years, all three would be extirpated. At an initial abundance of 200 individuals, one of the three subpopulations (Palmyra) would experience a serious decline over 10 years, and by 40 years, all three would experience substantial declines.

In total, we looked at 360 combinations of fishery interaction rates, initial population abundance, and r_{max} values to assess the potential impact of the fishery on giant manta ray subpopulations in the Action Area in the absence of specific information on population abundances or trends. As noted earlier, we consider those population trajectories using $r_{max} = 0.019$ to be the most likely, however we acknowledge that $r_{max} = 0.042$ cannot be discounted. We also consider the results of Scenario 1 to be the most representative of the impacts of the fishery in giant manta ray subpopulations in the Action Area. Finally, NMFS does not expect that the proposed action would extirpate any of the putative subpopulations of giant manta ray in the Action Area under the assumption that population sizes are at least 1,000 individuals. Moreover, because the most likely scenario – Scenario 1 – allows population increases over the next 40 years for all three subpopulations assuming a minimum initial size of 600 for Palmyra and 400 for Hawaii and Johnston Atoll, we do not find that the proposed action causes material changes having biological consequences to the species' numbers, reproduction, or distribution. In accordance with Section 1.3.1 (Jeopardy Analyses) above, NMFS does not anticipate the DSLL fishery will reduce appreciably the likelihood of the survival or recovery of the giant manta ray in the wild by reducing its reproduction, numbers, or distribution.

5.2 Indo-West Pacific Scalloped Hammerhead Shark

Three identified scalloped hammerhead DPSs occur in the Pacific, two of which are listed under the ESA that are potentially exposed to fishing activities in the Action Area. However, only the Indo-West Pacific scalloped hammerhead shark is expected to be adversely affected by the fishery (see Appendix A for the discussion on the eastern Pacific scalloped hammerhead shark) and is listed as threatened. As discussed in the Status of Listed Resources, the global population is estimated to be 280,000 adults based on an analysis of effective population size but there are no estimates for the Indo-West Pacific DPS. Overutilization by industrial/commercial fisheries, artisanal fisheries, and illegal fishing are the most serious threats to the persistence of the Indo-West Pacific scalloped hammerhead shark. Indo-West Pacific scalloped hammerhead sharks are both targeted and incidentally captured as bycatch in many fisheries. Contributing to the Indo-West Pacific scalloped hammerhead shark's biological vulnerability is the fact that these sharks are obligate ram ventilators and suffer very high at-vessel fishing mortality in fisheries (Morgan and Burgess 2007; Macbeth et al. 2009; Miller et al. 2014; Dapp et al. 2016).

As discussed in the Environmental Baseline, because the Indo-West Pacific scalloped hammerhead shark range is comprised of open ocean environments occurring over broad geographic ranges, large-scale impacts such as global climate change that affect ocean temperatures, currents, and potentially food chain (trophic) dynamics, climate change may pose a risk to this DPS. Chin et al. (2010) conducted an integrated risk assessment which determined scalloped hammerhead sharks were ranked as having a low overall vulnerability to climate change, with low vulnerability to each of the assessed climate change factors. However, much of the DSLL fishery does not overlap with the geographical range of the DPS and many effects from the stress regimes created by the Environmental Baseline are unknown for this particular DPS. No other fisheries which overlap the DSLL's Action Area are known to capture this DPS.

Historically, there have been 3 documented Indo-west scalloped hammerhead sharks observed captured with an estimated total of 13 interactions from the DSLL fishery between 2004 and 2021 (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021b, 2022b; NMFS 2018b).

As discussed in the Exposure Analysis section, estimates of the number of Indo-West Pacific scalloped hammerhead sharks that are likely to be captured in the DSLL fishery each year are presented in Table 22 (estimates are from McCracken 2019a). On average, 1 (95th Percentile: 4) of these sharks is likely to be captured each year in the DSLL fishery. We expect over 5 years, a maximum cumulative sum of 14 Indo West scalloped hammerhead sharks will be caught in the fishery with a 5-year maximum running average of 2.8 sharks. These levels of capture would only be expected if the fishery operates within the range of the Indo-West Pacific hammerhead, which is essential south of 10° N and where the fishery hasn't operated since 2016, although there are no restrictions on the fishery operating in that area.

As discussed in the Exposure and Response Analysis sections, we estimate an at-vessel mortality rate of 28.13% and a post-interaction mortality rate of 19.5% for a total mortality rate of 42.14%. Based on the maximum 5-yr running average we would expect an average of 1.2 mortalities each year or 6 mortalities over 5 years.

As discussed previously in the Status of Listed Resources, Pacoureau et al. (2020) indicates a 67% decline from 1970 to 2018 equating to a 2.31% decline per year for the global population and we estimated a 2.28% decline for the Indo-West Pacific scalloped hammerhead shark based on ocean basin trends reported in Pacoureau et al. (2020).

As discussed in the Risk section, as we do not have abundance estimates for this DPS, we assessed the impact of removing one mortality per year to assess if the DSLL is having a measurable impact on the DPS' survival and recovery. We found that the rate of decline (2.28% per year) did not change by removing the potential impacts of the DSLL. Therefore, we are reasonably certain the action will not change the number of populations that comprise the DPS or the spatial distribution of those populations. While the proposed action will remove animals from one of the populations, it will not appreciably reduce its expected patterns of growth and decline over time. Based on the available evidence, we do not expect the loss of 1.2 individuals per year to affect the viability of the population that these individuals represent considering the number of interactions expected to occur in this fishery, the corresponding mortality, the DPS' abundance, and the DPS' ability to persist. We do not expect the action to change the number of populations that comprise the DPS, the spatial distribution of those populations, or their expected patterns of growth and decline over time. As a result, NMFS does not anticipate the DSLL fishery will reduce appreciably the likelihood of the survival or recovery of Indo-West Pacific scalloped hammerhead sharks in the wild by reducing their reproduction, numbers, or distribution.

5.3 Oceanic Whitetip Shark

On average the Hawaii DSLL fishery is expected to capture a mean of 1,708 (95th percentile: 3,185) threatened oceanic whitetip sharks each year, in some years, numbers higher or lower than the mean are likely to be incidentally captured. Based on a total mortality rate of 0.326, we expect an average of 557 of the oceanic whitetip sharks incidentally captured are likely to die as a result of their capture each year. Over a 40-year analytical period we would anticipate a cumulative total of 84,712 incidental captures and 27,616 mortalities. This differs slightly from the recently published supplement (NMFS 2022b) because we incorporated additional data through 2022.

As discussed in the Status of Listed Resources section of this biological opinion, the best available information suggests that oceanic whitetip sharks in the Pacific Ocean are likely a single population, which is distinct from oceanic whitetip sharks in other parts of the species global range. NMFS estimated, based on the work of Tremblay-Boyer et al. (2019) that the portion of the population represented by the West Pacific stock is composed of about 775,000 oceanic whitetip sharks. Given that this estimate represents only part of the Pacific population, we analyzed the species under two scenarios: the West Pacific stock estimate is a reasonable *minimum* population size for the species in the Pacific Ocean (N= ~775,000); and the West Pacific stock estimate represents about 60% of the total number of oceanic whitetip sharks that comprise the total Pacific Ocean population (N= ~1.2M). Oceanic whitetip sharks have low fecundities for sharks (between 0 and 15 pups) and a biennial reproductive cycle.

Oceanic whitetip sharks are listed as threatened throughout their range and are classified as overfished and have experienced substantial declines in abundance, total biomass, spawning biomass, and recruitment levels (Rice and Harley 2012; Futerman 2018). The potential impacts from climate change on oceanic whitetip shark habitat are highly uncertain, but given their broad distribution in various habitat types, these species may be able to move to areas that suit their biological and ecological needs. Therefore, while effects from climate change have the potential to pose a threat to sharks in general, including habitat changes such as changes in currents and ocean circulation and potential impacts to prey species, species-specific impacts to oceanic whitetip sharks and their habitat are currently unknown, but Young et al. (2017) believe they are likely to be minimal.

While the primary threat to the oceanic whitetip shark's survival and recovery is fishing, particularly their capture and mortality occurring in longline and purse seine fisheries, we recognize that the Hawaii DSLL fishery and other WCPO longline and purse seine fisheries in the Action Area and throughout the species range have been undertaking a number of measures to reduce capture and mortality from capture in fisheries. During the course of this consultation, the Hawaii DSLL fishery converted to 100% monofilament leaders, and also placed greater emphasis on removing trailing gear to improve capture outcomes. Bigelow et al. (2022) provide evidence that WCPO oceanic whitetip shark population is now increasing due, in part, to these measures. More broadly, WCPFC's CMM 2022-04 prohibits the retention of oceanic whitetip sharks, and includes additional measures on minimizing bycatch (including some gear restrictions), implementing safe release practices and prohibiting wire leaders and shark line for longline fishing.

Due to the uncertainties regarding the current population trend, we conducted our risk assessment based on the assumption of the population experiencing a trend that ranges from slightly negative (-0.13% per year) to increasing at a rate of 0.36% per year which is inclusive of historic mortalities from the DSLL under its current operation (Rice et al. 2021). Although the recent regulations require the removal of trailing gear in the Hawaii DSLL fishery, to the greatest extent practicable (see 50 CFR 665.811), we assume the higher of two post-interaction mortality rates is a more reasonable estimate. This is because we have no data to estimate what proportion will be actually released with or without trailing gear in this fishery.

The action results in the capture of 1,708 (95th percentile: 3,185) oceanic whitetip sharks and the resulting death of 557 (95th percentile: 1,038) sharks per year. The maximum 5-year running average is 2,117.8 sharks with 690.4 mortalities. At projected abundance levels (1,292,023 individuals), this constitutes the death of 0.05% of the population per year. Even when we treat the WCPO stock estimate (775,000 individuals) as if it was a reasonable minimum estimate for the Pacific population, the Hawaii DSLL fishery only removes 0.09% of the current population annually. Because we do not have sufficient data to accurately estimate the current population trend, we conducted our assessment based on a range from -0.13% to +0.36%, which is inclusive of historic mortalities from the DSLL as currently managed. Without the mortalities from the DSLL, the population trends would range from -0.08% to 0.45% per year for 2022 to 2023 and from -0.075% to 0.439% per year from 2022 to 2062.

While we analyzed that range of population trend estimates, Bigelow et al. (2022) indicate the current rate is positive (inclusive of historic mortalities from the DSLL). Therefore, we are reasonably certain the population trend will continue to be positive with the proposed action. Thus, we are reasonably certain the proposed action will not cause material changes having biological consequences to the species' numbers, reproduction, or distribution. In accordance with Section 1.3.1 (Jeopardy Analyses) above, NMFS does not anticipate the DSLL fishery will reduce appreciably the likelihood of the survival or recovery of oceanic whitetip sharks in the wild by reducing their reproduction, numbers, or distribution.

5.4 Central North Pacific Green Sea Turtle

As described in the Status of Listed Species, the green sea turtle Central North Pacific is listed as threatened and since initial nesting surveys at the French Frigate Shoals index beach in 1973, there has been a marked increase of 5.4% in annual green turtle nesting (Balazs and Chaloupka 2004b; Balazs et al. 2015). The current population estimate is approximately 682,296 turtles, and an adult nester population of 3,846 (Balazs et al. 2015; Seminoff et al. 2015; Piacenza et al. 2016).

While the nesting population trajectory is positive and encouraging, more than 96% of nesting occurs at one site in the NWHI and it is highly vulnerable to threats including loss of nesting habitat. While the loss of the main nesting beach on East Island in French Frigate Shoals from Hurricane Walaka is still a concern, recent monitoring suggests that this DPS is successfully using alternate nesting habitat on nearby Tern Island.

Central North Pacific green turtles are exposed to a variety of natural threats both at their nesting beaches and in the open ocean. The beaches where turtles nest and the nest themselves are threatened by hurricanes, tropical storms, storm surges, sand accretion, and rainfall associated with storms.

The concentrated nature and relatively small size of the population make it vulnerable to random variation and stochasticities in the biological and physical environment, including natural catastrophes, as well as changes in climate and resulting effects such as sea level rise, and other anthropogenic threats. This increases its risk of extinction, even though it may have positive population growth (Seminoff et al. 2015). Both non-stochastic as well as stochastic events are significant current and future threats to this small, isolated, concentrated population.

As described in the Environmental Baseline, past and present fisheries interactions have been, and continue to be, a threat to Central North Pacific green turtles within the Action Area. Bycatch of green sea turtles occurs in many fisheries throughout the geographic oceanic range of this DPS. Historically, the DSLL fishery has observed 25 green sea turtles and interacted with an estimated total of 128 individuals between 2004 and 2022 (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a). Of those, approximately 20% were estimated to belong to the Central North Pacific (mean=20%; 95% CI=4-43%); NMFS unpublished data). While mitigation and minimization measures have reduced fisheries sea turtle bycatch in the United States in recent years, large numbers of Central North Pacific green sea turtles are still routinely captured in international, federal, and state commercial fisheries that target other DPSs (Balazs et al. 1987; NMFS 2012b). Some Central North Pacific green turtles also interact with recreational hook-and-line fisheries (Francke et al. 2013). These activities are reasonably likely to continue and may increase over time due to the effects of increased human population and increased human consumption of fish products.

In addition, warming sea surface temperatures may lead to potential fitness consequences in sea turtles resulting from altered seasonality and duration of nesting (Pike et al. 2006). Sea turtles may also expand their range as temperature-dependent distribution limits change (McMahon and Hays 2006; Poloczanska et al. 2009). Further, sea turtles have temperature-dependent sex determination, and many populations produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Newson et al. 2009; Patricio et al. 2017; Jensen et al. 2018). Increases in both the frequency and severity of storms as a result of climate change may lead to further loss of nesting habitat.

Environmental changes associated with climate change are occurring within the Action Area and are expected to continue into the future. Marine populations that are already at risk due to other threats are particularly vulnerable to the direct and indirect effects of climate change.

As described in the Exposure Analysis, hooking and entanglement are the most significant stressors that green sea turtles are exposed to directly from the action. Since 2004, 25 green sea turtles have been observed caught in the DSLL fishery. When these reports are adjusted to account for the percentage of observer coverage, about 128 green sea turtles are likely to have been captured in the fishery in the 19-year period from 2004 to 2022 (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a). Over 10 years, we anticipate that a cumulative mean of 154 Central North Pacific green sea turtles will be captured by the DSLL fishery and of those, 148 are expected to die as a result of their interaction. Assuming total abundance for Central North Pacific green sea turtles is 682,296 individuals suggests that over 10 years this level of interaction would represent 0.023% of the population being impacted and 0.022% of the population being killed.

As this population is increasing in size (Balazs et al. 2015; Seminoff et al. 2015), these values will decrease at 10, 20, 30, and 40 years into the future as we anticipate higher population sizes at those time points. Over a 40 year time horizon, if captures remain consistent with historical numbers we would anticipate a cumulative total of 616 captures and of those, 592 would be expected to die. Although it remains to be seen how this DPS responds to the loss of nesting habitat at French Frigate Shoals, we are reasonably certain the population trend will continue to be positive with the continued authorization of the DSLL fishery as currently managed. Because Central North Pacific green sea turtle abundance will continue to grow with the proposed action, we are reasonably certain it will not cause material changes having biological consequences to the DPS' numbers, reproduction, or distribution. In accordance with Section 1.2.2 (Jeopardy Analyses) above, NMFS does not anticipate the DSLL fishery will reduce appreciably the likelihood of the survival or recovery of Central North Pacific green sea turtles in the wild by reducing their reproduction, numbers, or distribution.

5.5 Central South Pacific Green Sea Turtle

As described in the Status of Listed Resources, the Central South Pacific green sea turtle is listed as endangered and population trends are poorly understood. Based on available data, we estimate the DPS has 2,677 adult females (Seminoff et al. 2015) and based on this we estimate a total population size of 631,745 (range 323,514 to 1,669,441) individuals greater than one year old as of 2013. No sites have long-term monitoring programs, and no single site has had standardized surveys for even five continuous years to allow for trend estimation. Based on historic declines and ongoing threats, this DPS is anticipated to be declining (Seminoff et al. 2015). As a proxy, we use the declining trend in CPUE for green sea turtles captured in the WCPO longline fisheries (-5.8% per year; 95% CI -7.6% to -3.9%) as an estimate of the DPS' trend (Figure 30). At the mean rate of decline we estimate the population size in 2022 to be 420,939.

Central South Pacific green sea turtles are exposed to a variety of natural threats both at their nesting beaches and in the open ocean. Depredation may have been a factor that contributed to the historical decline of this population. The best available data suggest that current nest and hatchling predation on several Central South Pacific green sea turtle nesting beaches and in water habitats is a potential threat to this population. Directed harvest of turtles and their eggs is an ongoing problem in the Central South Pacific in American Samoa, Fiji, Kiribati (e.g., Phoenix Islands), Tuvalu, Tokelau, and the Cook Islands (Balazs 1983; Tuato'o-Bartley et al. 1993; Weaver 1996a, 1996b; NMFS and FWS 1998a; Obura and Stone 2002; Alefaio et al. 2006). Commercial harvest (a major threat), as well as subsistence and ceremonial harvest, are all contributing factors. Other anthropogenic threats include incidental capture in artisanal and commercial fisheries.

As described in the Environmental Baseline, past and present fisheries interactions may have been, and may continue to be, a threat to Central South Pacific green sea turtles within the Action Area. Bycatch of green sea turtles occurs in many fisheries throughout the geographic oceanic range of this DPS. While mitigation and minimization measures have reduced fisheries sea turtle bycatch in the United States in recent years, large numbers of Central South Pacific green sea turtles may still be captured in international and United States commercial fisheries that target other species. These activities are reasonably likely to continue and may increase over time because of increased human population and increased human consumption of fish products.

In addition, warming sea surface temperatures may lead to potential fitness consequences in sea turtles resulting from altered seasonality and duration of nesting (Pike et al. 2006). Sea turtles may also expand their range as temperature-dependent distribution limits change (McMahon and

Hays 2006; Poloczanska et al. 2009). Further, sea turtles have temperature-dependent sex determination, and many populations produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Newson et al. 2009; Patricio et al. 2017; Jensen et al. 2018).

Environmental changes associated with climate change are occurring within the Action Area and are expected to continue into the future. Marine populations that are already at risk due to other threats are particularly vulnerable to the direct and indirect effects of climate change.

As described in the Exposure Analysis, all individuals that were measured would be considered juveniles with one possible subadult. Since 2004, 25 green sea turtles have been observed caught in the DSLL fishery. When these reports are adjusted to account for the percentage of observer coverage, about 128 green sea turtles are likely to have been captured in the fishery in the 19-year period from 2004 to 2022 (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a). Over 10 years, we anticipate that a cumulative mean of 154 Central South Pacific green sea turtles will be captured by the DSLL fishery and of those, 148 are expected to die as a result of their interaction. Over a 40 year time horizon, if captures remain consistent with historical numbers we would anticipate a cumulative total of 616 captures and of those 592 would be expected to die.

As discussed in the Risk Analysis section of this biological opinion, the action results in the capture of 15.4 and the mortality of an average of 14.8 Central South Pacific green sea turtles per year. At the 2022 projected abundance levels (420,939 individuals), incidentally capturing and killing an average of 14.8 Central South Pacific green sea turtle per year constitutes the death of 0.0035% of the population. If captures remain the same as the population declines, the fishery would incidentally kill 0.036% of the population in 40 years which we consider to be the most conservative reasonable scenario, as capture rates are likely to decline if the population declines by 90% over this time period as predicted from a 5.8% per year decline. Assuming the population declines include the historic impact of the DSLL, removing the fishery would result in population declines of between 5.799 to 5.8% with the lower rate of decline representative of the population growth rate over 40 years. These values are well within the 95% CI for our estimate of the rate of the population decline (-7.6% to -3.9%).

A power analysis indicates more than 200,000 years of data would be required to detect this difference and therefore the difference is not statistically detectable given our predictive capabilities. Thus, the trajectory of the population is not substantially changed by the removal of mortalities from the DSLL fishery under a scenario with the highest potential impact to the DPS. Because Central South Pacific green sea turtle abundance will not be substantially impacted by the proposed action, we are reasonably certain it will not cause material changes having biological consequences to the DPS' numbers, reproduction, or distribution. In accordance with Section 1.2.2 (Jeopardy Analyses) above, NMFS does not anticipate the DSLL fishery will reduce appreciably the likelihood of the survival or recovery of Central South Pacific green sea turtles in the wild by reducing their reproduction, numbers, or distribution.

5.6 Central West Pacific Green Sea Turtle

As described in the Status of Listed Resources, the Central West Pacific green sea turtle is listed as endangered and there is insufficient long-term and standardized monitoring information to adequately describe abundance and the population trend. The limited available information suggests a nesting population decrease in some areas like the Marshall Islands, or unknown trends in other areas such as Palau, Papua New Guinea, the Marianas, Solomon Islands, or the FSM (Maison et al. 2010). Currently, NMFS estimates the Central West Pacific has approximately 51 nesting sites with a total of 6,518 nesting females (Seminoff et al. 2015) and based on this we estimate a total population size of 1,543,625 (range 821,772 to 3,592,842) individuals greater than one year old. Based on historic declines and ongoing threats, this DPS is anticipated to be declining (Seminoff et al. 2015). As a proxy, we use the declining trend in CPUE for green sea turtles captured in the WCPO longline fisheries (-5.8% per year) as an estimate of the DPS' trend (Figure 30). At this rate of decline we estimate the population size in 2022 to be 1,028,533.

Central West Pacific green sea turtles are exposed to a variety of natural threats both at their nesting beaches and in the open ocean. Directed harvest of turtles and their eggs is an ongoing problem in the Central West Pacific in the CNMI, FSM, Guam, Kiribati (Gilbert Islands chain), Papua, Papua New Guinea, Republic of the Marshall Islands, and Palau (Eckert 1993; Hitipeuw and Maturbongs 2002; Philip 2002; Humber et al. 2014). In addition to the collection of eggs from nesting beaches, the killing of nesting females continues to threaten the stability of green turtle populations. Other anthropogenic threats include incidental capture in artisanal and commercial fisheries.

As described in the Environmental Baseline, past and present fisheries interactions may have been, and may continue to be, a threat to Central West Pacific green sea turtles within the Action Area. Bycatch of green sea turtles from this population occurs in many fisheries throughout the geographic oceanic range of this DPS. While mitigation and minimization measures have reduced fisheries sea turtle bycatch in the United States in recent years, large numbers of Central West Pacific green sea turtles may still be captured in international and United States commercial fisheries that target other species. These activities are reasonably likely to continue and may increase over time due to the effects of increased human population and increased human consumption of fish products.

In addition, warming sea surface temperatures may lead to potential fitness consequences in sea turtles resulting from altered seasonality and duration of nesting (Pike et al. 2006). Sea turtles may also expand their range as temperature-dependent distribution limits change (McMahon and Hays 2006; Poloczanska et al. 2009). Further, sea turtles have temperature-dependent sex determination, and many populations produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Newson et al. 2009; Patrício et al. 2017; Jensen et al. 2018).

Environmental changes associated with climate change are occurring within the Action Area and are expected to continue into the future. Marine populations that are already at risk due to other threats are particularly vulnerable to the direct and indirect effects of climate change.

As described in the Exposure Analysis, hooking and entanglement are the most significant stressors that green sea turtles are exposed to directly from the action. Between 2004 and 2022, 25 green sea turtles have been observed caught in the DSLL fishery. When these reports are adjusted to account for the percentage of observer coverage, about 128 green sea turtles are likely to have been captured in the fishery in the 18-year period from 2004 to 2022 (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a). Over 10 years, we anticipate that a cumulative mean of 154 Central West Pacific green sea turtles will be captured by the DSLL fishery and of those, 148 are expected to die as a result of their interaction. Over a 40 year time horizon, if captures remain consistent with historical numbers we would anticipate a cumulative total of 616 captures and of those, 592 would be expected to die.

As discussed in the Risk Analysis section of this biological opinion, the action results in the capture of 15.4 and the mortality of an average of 14.8 Central West Pacific green sea turtles per year. At the 2022 projected abundance levels (1,028,533 individuals), incidentally capturing and killing an average of 14.8 Central West Pacific green sea turtle per year constitutes the death of 0.0014% of the population. If captures remain the same as the population declines, the fishery would incidentally kill 0.015% of the population in 40 years which we consider to be the most conservative reasonable scenario, as capture rates are likely to decline if the population declines by 90% over this time period as predicted from a 5.8% per year decline. Assuming the population declines include the historic impact of the DSLL, removing the fishery mortalities would result in population declines of between 5.7942 to 5.7985% with the lower rate of decline representative of the population growth rate over 40 years. These values are well within the 95% CI for our estimate of the rate of the population decline (-7.6% to -3.9%).

A power analysis indicates more than 1 million years of data would be required to detect this difference and therefore the difference is not statistically detectable given our predictive capabilities. Thus, the trajectory of the population is not substantially changed by the removal of mortalities from the DSLL fishery under a scenario with the highest potential impact to the DPS. Because Central West Pacific green sea turtle abundance will not be substantially impacted by the proposed action, we are reasonably certain it will not cause material changes having biological consequences to the DPS' numbers, reproduction, or distribution. In accordance with Section 1.2.2 (Jeopardy Analyses) above, NMFS does not anticipate the DSLL fishery will reduce appreciably the likelihood of the survival or recovery of Central West Pacific green sea turtles in the wild by reducing their reproduction, numbers, or distribution.

5.7 East Pacific Green Sea Turtle

As described in the Status of Listed Resources, the East Pacific green sea turtle is listed as threatened and nesting has been steadily increasing at the primary nesting sites in Michoacán, Mexico, and in the Galapagos Islands since the 1990s (Delgado and Nichols 2005; Senko et al. 2011). Nesting trends at Colola have continued to increase since 2000 with the overall East Pacific green turtle population also increasing at other nesting beaches in the Galapagos and Costa Rica (NMFS and FWS 2007a; Wallace et al. 2010a). Based on nesting beach data, the current adult female nester population for Colola, Michoacan is 11,588 females, which makes this the largest nesting aggregation in the threatened East Pacific green sea turtle, comprising nearly 58% of the total adult female population. The total for the entire East Pacific green sea turtle is estimated at 20,062 nesting females (Seminoff et al. 2015). There are currently no total abundance estimates for this DPS; however, we estimate that the total population of the East Pacific green sea turtle is 3,580,207.

East Pacific green sea turtles are exposed to a variety of natural threats both at their nesting beaches and in the open ocean. The beaches where turtles nest and the nest themselves are threatened by hurricanes, tropical storms, storm surges, sand accretion, and rainfall associated with storms. The largest threat on nesting beaches to the East Pacific green sea turtle is reduced availability of habitat due to heavy armament, subsequent erosion, and coastal development. Incidental capture in artisanal and commercial fisheries is a significant threat to the survival of green turtles throughout the eastern Pacific Ocean. Significant bycatch has been reported in artisanal gill net and longline shark and mahi mahi fisheries operating out of Peru (Kelez et al. 2003; Alfaro-Shigueto et al. 2006) and, to a lesser extent, Chile (Donoso and Dutton 2010).

As described in the Environmental Baseline, past and present fisheries interactions have been, and continue to be, a threat to East Pacific green turtles within the Action Area. Bycatch of green sea turtles in this DPS occurs in many fisheries throughout the geographic oceanic range of this DPS. While mitigation and minimization measures have reduced fisheries sea turtle bycatch in the United States in recent years, large numbers of East Pacific green sea turtles are still routinely captured in international, federal, and state commercial fisheries that target other species.

In addition, warming sea surface temperatures may lead to potential fitness consequences in sea turtles resulting from altered seasonality and duration of nesting (Pike et al. 2006). Sea turtles may also expand their range as temperature-dependent distribution limits change (McMahon and Hays 2006; Poloczanska et al. 2009). Further, sea turtles have temperature-dependent sex determination, and many populations produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Newson et al. 2009; Patricio et al. 2017; Jensen et al. 2018). Increases in both the frequency and severity of storms as a result of climate change may lead to further loss of nesting habitat.

Environmental changes associated with climate change are occurring within the Action Area and are expected to continue into the future. Marine populations that are already at risk due to other threats are particularly vulnerable to the direct and indirect effects of climate change.

As described in the Exposure Analysis, hooking and entanglement are the most significant stressors that green sea turtles are exposed to directly from the action. Since 2004, 25 green sea turtles have been observed caught in the DSLL fishery. When these reports are adjusted to account for the percentage of observer coverage, about 128 green sea turtles are likely to have been captured in the fishery in the 19-year period from 2004 to 2022 (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a).

Over 10 years we anticipate a cumulative mean of 154 total captures of East Pacific green sea turtles, of which, 148 would be expected to die as a result of their interaction. Assuming total abundance for East Pacific green sea turtles is 3,580,207 individuals, suggests that over 10 years this level of interaction would represent 0.0043% of the population being captured and 0.0041% of the population being killed. As this population is increasing in size (Seminoff et al. 2015), these values will decrease at 10, 20, 30, and 40 years into the future as we anticipate higher population sizes at those time points. Over a 40 year time horizon, if captures remain consistent with historical numbers we would anticipate a cumulative total of 616 captures. Of these, on average, 592 would be expected to die.

The action results in the capture and mortality of an average of 13 East Pacific green sea turtles per year. The maximum 5-year running average is 15.4 turtles with 14.8 mortalities. At projected abundance levels (3,580,207 individuals), this constitutes the death of 0.0022% of the population per year. As this DPS has been experiencing increasing trends, this level of impact will not change the DPS' overall trajectory.

Therefore, we are reasonably certain the population trend will continue to be positive with the continued authorization of the DSLL fishery as currently managed. Because East Pacific green sea turtle abundance will continue to grow with the proposed action, we are reasonably certain it will not cause material changes having biological consequences to the DPS' numbers, reproduction, or distribution. In accordance with Section 1.2.2 (Jeopardy Analyses) above, NMFS does not anticipate the DSLL fishery will reduce appreciably the likelihood of the survival or recovery of East Pacific green sea turtles in the wild by reducing their reproduction, numbers, or distribution.

5.8 East Indian – West Pacific Green Sea Turtle

As described in the Status of Listed Species, the East Indian-West Pacific green sea turtle is listed as threatened and there are four sites for which 15 or more years of recent data are available for annual nester abundance: Sabah Turtle Islands in Malaysia; Royal Navy Center in Khram Island, Thailand; Redang in Terrengganu, Myanmar; and Thameela Islands, Myanmar. Only Sabah Turtle Islands represent a sizable nesting population, estimated at 7,011 in 2011 with a sex ratio of 1M: 4F (Pilcher 2010). The PVA indicates that the nesters from Sabah Turtle Islands in Malaysia, with an estimated 7,000 nesters, will likely continue to increase, while the nesters from the Royal Navy Center in Khram Island, Thailand (estimated 297 nesters), Redang in Terrengganu, Myanmar (estimated 278 nesters), and Thameela Islands, Myanmar (estimated 109 nesters) will likely continue to decline (Seminoff et al. 2015). The total abundance for this population is estimated at 77,009 nesters (Seminoff et al. 2015) and based on this we estimate total 2015 population size of 18,171,565 (range 9,227,181 to 43,725,766) individuals greater than one year old. Based on the analysis of Mazaris et al. (2017) we estimate this DPS is declining at a rate of 1.11% per year (range -3.6% to +6.7). At the mean rate of decline we estimate the population size in 2022 to be 16,813,093.

East Indian-West Pacific green sea turtles are exposed to a variety of natural threats both at their nesting beaches and in the open ocean. The best available data suggest that current nest and hatchling predation on the East Indian-West Pacific green sea turtle is prevalent. Depredation of nests by feral animals is also widespread throughout the range of the DPS.

As described in the Environmental Baseline, past and present fisheries interactions may have been, and may continue to be, a threat to East Indian-West Pacific green sea turtles within the Action Area. Bycatch of green sea turtles occurs in many fisheries throughout the geographic oceanic range of this DPS. While mitigation and minimization measures have reduced fisheries sea turtle bycatch in the United States in recent years, large numbers of East Indian-West Pacific green sea turtles may still be captured in international and United States commercial fisheries that target other species. These activities are reasonably likely to continue and may increase over time due to the effects of increased human population and increased human consumption of fish products.

In addition, warming sea surface temperatures may lead to potential fitness consequences in sea turtles resulting from altered seasonality and duration of nesting (Pike et al. 2006). Sea turtles may also expand their range as temperature-dependent distribution limits change (McMahon and Hays 2006; Poloczanska et al. 2009). Further, sea turtles have temperature-dependent sex determination, and many populations produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Newson et al. 2009; Patrício et al. 2017; Jensen et al. 2018). Increases in both the frequency and severity of storms as a result of climate change may lead to further loss of nesting habitat.

Environmental changes associated with climate change are occurring within the Action Area and are expected to continue into the future. Marine populations that are already at risk due to other threats are particularly vulnerable to the direct and indirect effects of climate change.

As described in the Exposure Analysis, hooking and entanglement are the most significant stressors that green sea turtles are exposed to directly from the action. Since 2002, 25 green sea turtles have been observed caught in the DSLL fishery. When these reports are adjusted to account for the percentage of observer coverage, about 128 green sea turtles are likely to have been captured in the fishery in the 19-year period from 2004 to 2022 (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a).

Over 10 years, we anticipate that a cumulative mean of 154 East Indian-West Pacific green sea turtles will be captured by the DSLL fishery and of those, 148 are expected to die as a result of their interaction. Over a 40 year time horizon, if captures remain consistent with historical numbers we would anticipate a cumulative total of 616 captures. Of these, on average, we expect 592 to die.

As discussed in the Risk Analysis section, the action results in the capture and mortality of an average of 13 East Indian - West Pacific green sea turtle per year. The maximum 5-year running average is 15.4 turtled with 14.8 mortalities. At the 2022 projected abundance levels (16,813,093 individuals), incidentally capturing and killing an average of 14.8 East Indian - West Pacific green sea turtle per year constitutes the death of 0.000088% of the population. In 40 years we anticipate the population declining by 36% and, if captures remain the same, the fishery would incidentally capture and kill 0.00014% of the population. Assuming the population declines include the historic impact of the DSLL, removing the fishery mortalities would result in population declines of between 1.10989% to 1.10991% with the lower rate of decline representative of the population growth rate over 40 years. Given the estimated standard deviation for the population growth rate (0.028), and based on a power analysis for detecting differences in means with the same variance (Cohen 1988), more than one million years of data would be required to detect a statistically significant difference in these values. Therefore, the difference is not statistically detectable given our predictive capabilities. Because East Indian-West Pacific green sea turtle abundance will not be substantially impacted by the proposed action, we are reasonably certain it will not cause material changes having biological consequences to the DPS' numbers, reproduction, or distribution. In accordance with Section 1.2.2 (Jeopardy Analyses) above, NMFS does not anticipate the DSLL fishery will reduce appreciably the likelihood of the survival or recovery of East Indian-West Pacific green sea turtles in the wild by reducing their reproduction, numbers, or distribution.

5.9 Southwest Pacific Green Sea Turtle

As described in the Status of Listed Resources, the Southwest Pacific green sea turtle is listed as threatened and nesting occurs in many islands throughout their range, but there are only two nesting areas (Raine Island and Heron Island) with long-term (>15 years) annual indices of nesting abundance.

The number of turtles nesting in the Great Barrier Reef area of Australia differs widely from year to year and is well correlated with an index of the Southern Oscillation (Limpus and Nicholls 2000 as cited in Seminoff et al. 2015). There are currently no total population estimates for this DPS, however, we estimate that the total population of the Southwest Pacific green sea turtle to be 14,762,154).

Southwest Pacific green turtles are exposed to a variety of natural threats both at their nesting beaches and in the open ocean. This DPS of green turtles are vulnerable to harvest throughout Australia and neighboring countries such as New Caledonia, Fiji, Vanuatu, Papua New Guinea, and Indonesia (Limpus 2009). Nesting turtles are also vulnerable to the Queensland East Coast Trawl Fisheries and the Torres Strait Prawn Fishery, and to the extent they forage west of Torres Strait, they are also vulnerable to the northern prawn fishery. Total mortality of green turtles in fisheries bycatch is not known because there is not reliable reporting of threatened DPS bycatch in Australian commercial fisheries (Limpus 2009).

As described in the Environmental Baseline, past and present fisheries interactions may have been, and may continue to be, a threat to Southwest Pacific green sea turtles within the Action Area. Bycatch of green sea turtles occurs in many fisheries throughout the geographic oceanic range of this DPS. While mitigation and minimization measures have reduced fisheries sea turtle bycatch in the United States in recent years, large numbers of Southwest Pacific green sea turtles may still be captured in international and United States commercial fisheries that target other species. These activities are reasonably likely to continue and may increase over time due to the effects of increased human population and increased human consumption of fish products.

In addition, warming sea surface temperatures may lead to potential fitness consequences in sea turtles resulting from altered seasonality and duration of nesting (Pike et al. 2006). Sea turtles may also expand their range as temperature-dependent distribution limits change (McMahon and Hays 2006; Poloczanska et al. 2009). Further, sea turtles have temperature-dependent sex determination, and many populations produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Newson et al. 2009; Patricio et al. 2017; Jensen et al. 2018). Increases in both the frequency and severity of storms as a result of climate change may lead to further loss of nesting habitat.

Environmental changes associated with climate change are occurring within the Action Area and are expected to continue into the future. Marine populations that are already at risk due to other threats are particularly vulnerable to the direct and indirect effects of climate change.

As described in the Exposure Analysis, hooking and entanglement are the most significant stressors that green sea turtles are exposed to directly from the action. Between 2004 and 2022, 25 green sea turtles have been observed caught in the DSLL fishery. When these reports are adjusted to account for the percentage of observer coverage, about 128 green sea turtles are likely to have been captured in the fishery in the 18-year period from 2004 to 2022 (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a). Over 10 years, we anticipate that a cumulative mean of 154 Southwest Pacific green sea turtles will be captured by the DSLL fishery and of those, 148 are expected to die as a result of their interaction. Assuming total abundance for Southwest Pacific green sea turtles is 14,762,154 individuals suggests that over 10 years this level of interaction would represent 0.0010% of the population being captured and killed.

The action results in the capture of a mean of 13 (95th percentile: 39) and mortality of a mean of 12 (95th percentile: 37) Southwest Pacific green sea turtle per year. The maximum 5-year running average is 15.4 turtle with 14.8 mortalities. At projected abundance levels (14,762,154 individuals), this constitutes the death of 0.00010% of the population per year. As this DPS has been experiencing increasing trends, this level of impact will not change the DPS' overall trajectory.

Therefore, we are reasonably certain the population trend will continue to be positive with the continued authorization of the DSLL fishery as currently managed. Because Southwest Pacific green sea turtle abundance will continue to grow with the proposed action, we are reasonably certain it will not cause material changes having biological consequences to the DPS' numbers, reproduction, or distribution. In accordance with Section 1.2.2 (Jeopardy Analyses) above, NMFS does not anticipate the DSLL fishery will reduce appreciably the likelihood of the survival or recovery of Southwest Pacific green sea turtles in the wild by reducing their reproduction, numbers, or distribution.

5.10 Leatherback Sea Turtle

As discussed in the Status of Listed Species section of this biological opinion, leatherback sea turtles are globally listed as endangered. The species is composed of seven populations, and the proposed action adversely affects the West Pacific Ocean and East Pacific Ocean populations. Because these two populations are demographically independent, we considered the effect of the action on these populations separately.

As described in the Exposure Analysis, hooking and entanglement are the most significant stressors that leatherback sea turtles are exposed to directly from the action. On average, we expect a mean of 17 (95th percentile: 43) leatherback sea turtles to be captured each year in the DSLL fishery, although the number captured in any given year may rarely exceed 43 (Table 8). Assuming the patterns found in the observer data from 2004 to 2021 are representative, the majority of leatherback turtles would be hooked rather than entangled (an average of about 87% with a 95% CI = 74% to 94%).

When at-vessel mortality and post-release mortality are combined, the effective mortality rate for leatherback turtles captured in this fishery is 35% (95% CI = 23 to 50%). At this mortality rate, if 17 leatherbacks are caught in any given year, we would expect about 6 of them to die as a result of being captured. If 43 leatherback turtles are captured, we would expect about 15 leatherback sea turtles to die as a result of being captured. Separating juveniles from adults, 13% (95% CI = 4 to 38%) of adults were dead at-vessel while 54% (95% CI = 29 to 77%) of juveniles were dead at vessel; this assessment assumes these percentages are applicable to future captures.

Based on genetic analyses of tissue samples collected from 16 leatherback turtles captured in the DSLL fishery, 93.75% of the leatherback turtles captured in this fishery are from the West Pacific population that occurs in Papua-Barat (Indonesia), Papua New Guinea, and Solomon Islands (Dutton pers. comm. 12 April 2018). The remaining 6.25% of the leatherback turtles are from the East Pacific population. When these percentages are used to assign leatherback mortalities to the two affected populations, of the 6 mortalities anticipated for the mean of 17 annual captures, we would expect a mean of 5 of the 16 leatherback turtles captured from the West Pacific population would die as a result of their capture and we would not expect any of the leatherback sea turtles from the East Pacific population to die in an average year as a result of being captured, though we would expect an average of two mortalities over five years. If 43 leatherback turtles are captured, we would expect 14 leatherback turtles from the West Pacific population would die as a result of their capture and 1 leatherback sea turtle from the East Pacific population would die as a result of being captured.

We add the DSLL effects to an Environmental Baseline of interactions and mortalities from other fisheries, including international and United States fisheries in the Action Area. The fisheries that occur in the Action Area and their effects are reasonably likely to continue and may increase over time due to the effects of increased human population, and increased human consumption of fish products. Between 2013 and 2020, an estimated 40 to 244 leatherbacks were killed annually (606 total estimated mortalities from 2013 to 2022) in WCPO longline fisheries, which is inclusive of the U.S. fisheries. This data includes mortalities outside of the action area, but those mortalities impact the condition of the species within the Action Area. For U.S. fisheries in the Action Area, in addition to the DSLL, the SSLL is anticipated to kill up to three leatherback sea turtles each year (NMFS 2019a).

Climate change may be affecting the species already but will likely increase in the future. Global average sea levels are expected to continue to rise by at least several inches in the next 15 years, and by one to four feet by 2100 (Wuebbles et al. 2017). This could affect migrations and feeding patterns by changing ocean circulation. Leatherback sea turtles were predicted to gain core habitat area by Hazen et al. (2012). Such range shifts could affect foraging success and sea turtle reproductive periodicity (Kashner et al. 2011; Pike 2014). Increased sand temperatures can also cause decreased egg survival and increase the proportion of female hatchlings, skewing sex ratios and affecting the reproductive capacity of the populations.

As described in the Status of Listed Species, two of the seven populations that comprise the leatherback sea turtle species, as it was listed, inhabit the Pacific Ocean and Action Area. Both populations have experienced precipitous declines. Presently the East Pacific leatherback nesting is likely comprised of approximately 1,007 adult females in 2020, with only 106 to 503 females nesting annually at primary and secondary nesting beaches (NMFS and FWS 2020a). Populations have declined by 97.4% during the past three generations (Wallace et al. 2013b). Based on current trends, we estimate the population will continue to decline at 8.1% per year and that in 2022 there are 924 adult females in the population. The West Pacific leatherback nesting was likely composed of 1,053 adult females in 2017 and given the relatively stable nesting numbers from 2018 to 2021 we assume this value represents current population size. Populations declined by 83% from the 1980’s to 2011 (Tiwari et al. 2013). Based on data up to 2017, Martin et al. (2020a) estimated that the adult female portion of the population would continue to decline at a mean of 6% per year (95% CI: -23.8% to 12.2%). However, recent data indicate that the mean rate of decline did not occur from 2018 to 2021 (Lontoh et al. In Prep, Figure 24) and current abundances are higher than what is predicted based on a 6% per year decline.

The number of leatherback sea turtles from the East Pacific population that are likely to be captured in any given year in the DSLL fishery ranges from a low of one to a high of three with zero to one mortality (assuming mean mortality rates). Based on the maximum 5-year running average of 18.4 leatherbacks captured in the DSLL fishery, annually we expect one of them to be from the East Pacific population and none would die from their capture in an average year. Based on the maximum 5-year running sum of 92 leatherback turtles that would be captured in the DSLL fishery, we expect 184 leatherback turtles would be captured over 10 years. Out of 184 leatherback turtles captured in ten years, 11.5 would be from the East Pacific population and we expect 4 of those to die. Over a 40 year time horizon, if captures remain consistent with historical numbers we would anticipate a cumulative total of 46 will be from the East Pacific population and we would anticipate that 16 will die as a result of their interactions.

Based on our estimate that 44% of leatherbacks captured by the DSLL are adults, we estimate that an average of 5 East Pacific adults may be captured over 10 years and that one of those will die. Over 40 years, we anticipate that the fishery will interact with 20 East Pacific adult leatherback turtles resulting in the mortality of 5 adults. The mean number of leatherback sea turtles from the West Pacific population that are likely to be captured in the DSLL fishery in any given year is 16 (95th percentile: 40) with 6 (95th percentile: 14) mortalities assuming mean mortality rates. Based on the maximum 5-year running average of 18.4 leatherbacks captured in the DSLL fishery, annually we expect 17.3 of them to be from the West Pacific population and 6.0 would die from their capture. Based on the maximum 5-year running sum of 92 leatherback turtles that would be captured in the DSLL fishery, we expect 184 leatherback turtles would be captured over 10 years. Out of 184 leatherback turtles captured in ten years, 172.5 would be from the West Pacific population and we expect 60 of those to die. Over a 20 year time horizon, if captures remain consistent with historical numbers we would anticipate a cumulative total of 368 captures, of those 345 would be from the West Pacific population and we would anticipate that 121 would die as a result of their interactions. We anticipate that approximately 44% of leatherback sea turtles interacting with the fishery are adults and that 24% of those will die as a result of their interaction. We estimate that an average of 1.8 West Pacific adults may be killed by the fishery in a year. Applying the sex ratio of 0.73, these adults equate to 1.2 adult females. Over 10 years, we anticipate that the fishery will interact with 76 adult leatherback turtles resulting in the mortality of 18 adults. Over 20 years, we anticipate that the fishery will interact with 152 West Pacific adult leatherback turtles resulting in the mortality of 36 adults.

Summary of Risk Analysis: East Pacific Leatherback Sea Turtles

Based on the Eastern Pacific population decline of 8.1% per year, we estimate that there were 1,083 adults in the population in 2022. At that abundance, the capturing and killing of one adult East Pacific leatherback every 10 years constitutes the capture of 0.047% and the death of

0.011% of the adult population per year. If captures in the DSLL remain the same as the population declines, the fishery would incidentally capture and kill 0.287% of the adult population in 40 years, which we consider the most conservative reasonable scenario as capture rates are likely to decline if the population declines by 96.3% over this time period as predicted from a 8.1% per year decline. The population declines include the historic impact of the DSLL. If we excluded mortalities from the fishery moving forward, results in population declines of between 8.02% to 8.10% with the lower rate of decline representative of the most conservative reasonable scenario described above. Current nesting beach trends for the East Pacific leatherback range from -15.5% to 9.5% and therefore the anticipated population effect (highest of 0.287%) of the fishery is well within our uncertainty regarding the current trend for this population. A power analysis indicated that almost 150,000 years of data would be required to detect this level of difference.

As noted, at the estimated rate of decline of 8.1% per year, in 40 years the population would be at 3.9% of 2022 abundance (96.1% decline). At the population decline of 8.02% per year predicted for 2062 if the impact of the DSLL is removed, this abundance level would be reached in 40.5 years (ln [3.9/100]/[8.02/100]), or 0.5 years later. This assumes fishery capture rates remain the same when the adult population size is reduced to 42 individuals, which is not likely to occur. We would expect fishery captures to decline, and therefore the proportion of the population experiencing mortalities attributable to the DSLL would also decline. This would reduce the difference in time to reach 3.9% of 2022 abundances to much less than 0.5 years. Thus, we do not anticipate that the trajectory of the population will be appreciably changed by interactions with the DSLL. Because the trajectory of East Pacific leatherback sea turtle abundance will not be appreciably impacted by the proposed action, we are reasonably certain it will not cause material changes having biological consequences to the species' numbers, reproduction, or distribution. In accordance with Section 1.2.2 (Jeopardy Analyses) above, NMFS does not anticipate the DSLL fishery will reduce appreciably the likelihood of the survival or recovery of East Pacific leatherback sea turtles in the wild by reducing their reproduction, numbers, or distribution.

Summary of Risk Analysis: West Pacific Leatherback Sea Turtles

The primary nesting beaches for the West Pacific leatherback population are Jamursba-Medi and Wermon beaches in Indonesia, representing 50 to 75% of the population's nesting (NMFS and FWS 2020a). Conservation efforts at these beaches have increased nest and nesting female protection and hatchling production (Pakiding et al. 2020), and it is possible these conservation efforts have alleviated the precipitous decline at these locations. We recognize that it is imperative that these efforts continue as well as new efforts initiated at other nesting beaches to fully understand the status of this population and protect and enhance its productivity. For the purpose of this analysis, we acknowledge the uncertainty regarding the current status of this population as detailed in the Status of Listed Resources section and noted by Martin et al. (2020a, 2020b) given the 95% credible interval for their estimate of the population's growth rate. Given the new information provided in Lontoh et al. (In Prep; Figure 24), we assume the population size estimated by Martin et al. (2020a) to be representative of the population size in 2022. We present information on the impact of the DSLL if the mean 6% per year decline occurs after 2022.

Based on NMFS' PVA model (Martin et al. 2020a; Siders et al. 2023), leatherback sea turtles in the West Pacific population are declining at about 6% per year (95% CI: -23.8% to 12.2%), and the population as indicated by the index beaches is at risk of falling to less than half of its 2017 abundance in 7.3 years (95% CI, 1 to 19 years) from 2021, or by about 2018, range 2022 to 2040. Based on Martin et al. (2020a), we estimate that there were approximately 1,054 adult females in the West Pacific population in 2017 or 1,442 adults based on a 73% female biased sex ratio and we further assume this population sizes are representative of current (2022) abundances.

At that abundance, the capturing and killing of 1.8 adult West Pacific leatherbacks each year constitutes the death of 0.13% of the adult population. If captures in the DSLL remain the same as the population declines, and if the population declines at a mean rate of 6% per year after 2022, the fishery would incidentally capture and kill 0.42% of the adult population in 20 years which we consider the most conservative reasonable scenario as capture rates are likely to decline if the population declines by 70% over this time period as predicted from a 6% per year decline. The population declines include the historic impact of the DSLL. Removing the fishery mortalities would result in population declines of between 5.75% and 5.87% with the lower rate of decline representative of the most conservative reasonable scenario described above. These anticipated population declines without the impact of the fishery are well within our uncertainty regarding the current trend for this population (95% CI: -23.8% to 12.2%) and a power analysis indicates we would not have the ability to detect this difference over a 20 year timeframe. Furthermore, a population declining at 5.75% per year would reach 12.5% of the 2022 population abundance less than one year later than a population declining at 6% per year (14.7 yr compared to 15.3 yr). This difference assumes fishery captures remain the same while the population declines of 434 adults which we consider unlikely.

Over shorter time periods, the impact of the DSLL has a lower impact on population growth rate because keeping DSLL captures the same as the population declines results in fishery captures having a greater proportional impact as the population continues to decline (i.e. 0.13% of the adult population is killed in 2022 but 0.42% of the adult population is killed in 2042 if fishery captures remain the same). A population declining at a rate of 5.87% per year will reach 50% of 2022 abundance levels 0.26 years (or about 3 months) later than a population declining at a rate of 6% per year (11.55 yr compared to 11.81 yr). The analysis above is predicated on the population declining at a rate of 6% per year after 2022. As of 2022, we anticipate the fishery kills approximately 0.13% of the adult population per year. This proportion of the population would not have an appreciable impact on the population growth rates, and is well within the uncertainty for the trend of this population (95% CI: -23.8% to 12.2%).

In summary, the West Pacific leatherback population declined by 78.3% from the 1980's to 2011 (Tapilatu et al. 2013) due to a combination of factors that include adult female harvest, low hatchling production due to egg harvest, predation, inundation and beach erosion (Tapilatu and Tiwari 2007; Tapilatu et al. 2013; Tapilatu 2014), and fishery bycatch (Lewison et al. 2014). The primary index beaches of Jamursba Medi and Wermon have been the focus of intense conservation efforts since the early 2000's and nest protection and relocation efforts have increased hatchling production from 21,966 per year between 2005 to 2013 to a minimum of 32,000 to 50,000 per year from 2017 to 2019. These efforts have likely slowed the decline of this species and stabilized trends (Lontoh et al. In Prep), although continuation of this potential stabilization will rely on continued efforts at these index beaches and the identification and protection of additional nesting beaches. The recent stabilization of nesting beach trends have occurred in conjunction with the continued operation of the DSLL capturing a mean of 16 West Pacific leatherbacks per year, suggesting the fishery is not impeding the potential for this population to recover or impede the goals set forth in the Leatherback Recovery Plan. At 2017 abundance levels, the DSLL removed 0.13% of the adult population which we described as not having an appreciable impact on population growth rates. Thus, we do not anticipate that the trajectory of the population will be appreciably changed by interactions with the DSLL. Because West Pacific leatherback sea turtle abundance will not be appreciably impacted by the proposed action, we are reasonably certain it will not cause material changes having biological consequences to the population's numbers, reproduction, or distribution.

We roll the effects on the two Pacific leatherback populations up to the species level, in accordance with Section 1.2.2 (Jeopardy Analyses) above. When we add the effects of the action and cumulative effects to the environmental baseline and in light of the status of the species we find the proposed action will not appreciably reduce the likelihood of both the survival and recovery of leatherback sea turtles in the wild by reducing their reproduction, numbers, or distribution. Our task in a section 7 consultation is to consider whether the nature and magnitude of the proposed action's effects, when considered together with the species status and all other threats acting on it, are consequential enough to appreciably reduce the species' likelihood of survival and recovery. Our analysis suggests that the proposed action's effects, which include interacting with a mean of 17 leatherback sea turtles and killing 6 leatherback sea turtles in any given year, would have inconsequential impacts on the species' overall reproduction, numbers and distribution in the wild. That is, our no-jeopardy determination merely concludes that the action's impacts, when considered together with the leatherback's baseline status and all other threats acting upon the species, is not the cause of some new peril or material change that exacerbates the species' decline.

Although we conclude the DSLL fishery does not jeopardize leatherback sea turtles, we identify measures in Sections 7.3 and 7.4 necessary or appropriate to minimize the impacts from takings in the fishery.

5.11 North Pacific Loggerhead Sea Turtle

As discussed in the Status of Listed Resources section, all loggerhead sea turtles caught in this fishery are North Pacific loggerhead sea turtles; which is comprised of at least three subpopulations (Matsuzawa et al. 2016). We do not know with certainty what portion of the loggerhead sea turtles that the fishery interacts with would be from the three different subpopulations but it is likely the fishery interacts with all three. While there has been an increase on Yakushima nesting beaches over the last decade, some beaches on mainland Japan have increased slightly while others have decreased (Matsuzawa et al. 2016).

Recent abundance estimates suggest that there are approximately 340,800 North Pacific loggerhead sea turtles (T. Jones pers. comm. 2019; Martin et al. 2020a). A recent PVA indicates that the adult female portion of the Yakushima subpopulation is increasing at a rate of approximately 2.3% per year and is comprised of 4,538 (95% CI: 4,077-5,064) adult females

(Martin et al. 2020a) at the index beaches or 8,726 adult females for the DPS assuming the index beaches represent 52% of all nesting for the DPS. This analysis found essentially no difference in the probabilities of the subpopulation reaching 50%, 25% and 12.5% of current abundance in population projections conducted with and without the fishery (Table 46; Martin et al. 2020a, 2020b).

Fisheries bycatch from pelagic longlining and artisanal coastal fisheries are the greatest threat to individual fitness and the DPS as described in the Status of Listed Species section. There is a great deal of uncertainty regarding which subpopulations face the greatest threat from fisheries.

As described in the Environmental Baseline, effects from international and United States fisheries have resulted in interactions with the North Pacific loggerhead sea turtle in the Action Area. These activities are reasonably likely to continue and may increase over time due to the effects of increased human population, and increased human consumption of fish products. The SSLL ITS authorized 36 captures with an anticipated 6 mortalities for North Pacific loggerhead sea turtles

Climate change may be affecting the DPS already but will likely increase in the future. Global average sea levels are expected to continue to rise by at least several inches in the next 15 years, and by one to four feet by 2100 (Wuebbles et al. 2017). This could affect migrations and feeding patterns by changing ocean circulation but will also reduce the beach access for nesting due to significant beach armoring in Japan. Increased sand temperatures can also cause decrease egg survival and an increase in female turtles skewing sex ratios and affecting the reproductive capacity of the population.

Loggerhead sea turtles are exposed to the stressors of hooking and entanglement in the Action Area. Since 2004, 16 loggerhead sea turtles were observed as captured in the DSLL fishery. When these reports are adjusted to account for the percentage of observer coverage, about 79 loggerhead sea turtles are likely to have been captured in the fishery in the 18-year period from 2004 to 2021 (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a). We anticipate that over 10 years, up to 86 North Pacific loggerhead sea turtles will be captured and of those 48 will be killed. Over 40 years, assuming captures remain consistent with historical capture rates, up to 344 North Pacific loggerhead sea turtles will be captured and 193 of those will be killed. As described in the Risk Analysis, the action results in the capture of an average of nine North Pacific loggerhead sea turtles with the mortality of five of those. The maximum 5-year running average is 8.6 turtles with 4.8 mortalities. At the estimated abundance level (340,797 individuals), incidentally capturing 8.4 and killing 4.8 North Pacific loggerhead sea turtles per year constitutes the death of 0.0014% of the population. As this DPS has been experiencing increasing trends, this level of impact will not change the DPS' overall trajectory.

As previously discussed, results of the PVA model suggest that the adult female portion of the North Pacific loggerhead sea turtle population is increasing at a mean rate of 2.3% per year (95% CI: -1.1 to 15.6%), and these rates are inclusive of the historic impact of the DSLL. If mortalities from the DSLL were to cease, the mean rate of increase would be 2.30015% per year, a difference that is much smaller than our confidence in the underlying data.

Therefore, we are reasonably certain the population trend will continue to be positive with the continued authorization of the DSLL fishery as currently managed. Because North Pacific loggerhead sea turtle abundance will continue to grow with the proposed action, we are reasonably certain it will not cause material changes having biological consequences to the DPS' numbers, reproduction, or distribution. In accordance with Section 1.2.2 (Jeopardy Analyses) above, NMFS does not anticipate the DSLL fishery will reduce appreciably the likelihood of the survival or recovery of North Pacific loggerhead sea turtles in the wild by reducing their reproduction, numbers, or distribution

5.12 Olive Ridley Sea Turtle

As discussed in the Status of Listed Resources section, there are two listed DPSs that occur in the Action Area, the endangered breeding colony populations on the Pacific coast of Mexico and the threatened populations from everywhere but the coast of Mexico endangered population. While the threatened olive ridley species is listed globally, there is genetic distinction between East and Indo-West Pacific olive ridleys (NMFS and FWS 2014) and 26.7% of the olive ridley sea turtles interacting with the DSLL fishery are likely from Indo-West Pacific beaches (Dutton pers. comm. June 29, 2018). The remaining 73.3% of captures are from the East Pacific and may either be from the endangered Mexico population or from East Pacific beaches of the threatened global species.

As described in the Environmental Baseline, effects from fisheries bycatch are the most significant threat for the species. There were 762 olive ridley sea turtles reported captured by 16 countries in the western and central Pacific Ocean from 1989 to 2015. Of these, 206 were a result of United States fishery interactions. When extrapolated from 5% observer coverage the estimate for all fishery interactions results in 15,240 olive ridley sea turtles caught in the western and central Pacific Ocean from 1989-2015. These activities are reasonably likely to continue and may increase over time due to the effects of increased human population, increased human consumption of fish products, and increased effort by fleets to catch fish.

No significant climate change-related impacts to olive ridley turtle populations have been observed to date. However, climate change-related impacts will likely influence biological trajectories in the future on a century scale (Paremsan and Yohe 2003). However, olive ridleys are migratory and are expected to adapt to changing conditions, which may minimize the impacts of climate change (Plotkin 2010 in NMFS and FWS 2014).

As discussed in the Exposure section, hooking is the most significant stressor that olive ridley sea turtles are exposed to directly from the action. During the 18-year time interval assessed, fishery observers reported that 212 olive ridleys were captured in the DSLL fishery. When these reports are adjusted to account for the percentage of observer coverage, about 1,055 olive ridley turtles are likely to have been captured in the fishery in the 18-year period from 2004 to 2021 (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021a, 2022a). We anticipate that over 10 years, up to 1,184 olive ridley sea turtles will be captured and of those 1,090 will be killed. Over 40 years, assuming captures remain consistent with historical capture rates, up to 4,736 olive ridley sea turtles will be captured and 4,361 of those will be killed. Below we summarize our analysis for eastern Pacific and Indo-West Pacific nesting beaches.

Eastern Pacific Populations. As noted in the Status of Listed Resources section, the endangered Mexico population of olive ridley sea turtles is comprised of more than 1,000,000 nesting females (Abreu-Grobois and Plotkin 2008; Valverde et al. 2012; NMFS and FWS 2014). The portion of the threatened population that nests in the East Pacific counts of adult olive ridley sea turtles on eastern Pacific nesting beaches similarly number at more than 1,000,000 nesting females on beaches from Costa Rica and Nicaragua (Chaves et al. 2005 as cited in NMFS and FWS 2014; Valverde et al. 2012; NMFS and FWS 2014).

We anticipate that over 10 years, up to 868 olive ridley sea turtles from East Pacific beaches will be captured and of those 799 will be killed. Over 40 years, assuming captures remain consistent with historical capture rates, up to 3,471 olive ridley sea turtles from East Pacific beaches will be captured, and of those 3,197 will be killed. Even when we considered the potential effects of these mortalities on adult abundance, which would overestimate their effect, these mortalities are not likely to appreciably affect the abundance of adult females on these populations in any given year or accumulated over 40 years.

Western Indo-Pacific Populations. As noted in the Status of Listed Resources section, we estimate a minimum population size of 316,883 juveniles and adults in the Western Pacific. We consider this a minimum estimate because it only represents two nesting beaches. There are numerous nesting aggregations throughout the West Pacific that do not have census data to allow us to include them in our estimate (NMFS and FWS 2014).

We anticipate that over 10 years, up to 316 olive ridley sea turtles from Indo-West Pacific beaches will be captured and of those 291 will be killed. Over 40 years, assuming captures remain consistent with historical capture rates, up to 1,264 olive ridley sea turtles from Indo-West Pacific beaches will be captured, and of those 1,164 will be killed.

As described in the Risk Analysis, we used these data to assess the percent of the population that may be impacted by the fishery assuming a population size of approximately 316,883 turtles. We anticipate that annual captures will represent no more than 0.010% of the population being impacted or no more that 0.0092 % of the population being killed. As this population is likely stable with some beaches experiencing increasing trends with the historic effect of the DSLL, we do not expect the continued operation of the fishery to change the species' overall trajectory.

In summary, we have concluded that the DSLL fishery is not likely to cause measurable change in the pattern of population growth or decline over time for either the endangered Mexico olive ridley sea turtle or the threatened global olive ridley sea turtle. Therefore, we do not expect the action to change the number of populations that comprise the species, the spatial distribution of those populations, or their expected patterns of growth and decline over time. As a result, in accordance with Section 1.2.2 (Jeopardy Analyses) above, NMFS does not anticipate the DSLL fishery will reduce appreciably the likelihood of the survival or recovery of either the endangered or threatened populations of olive ridley sea turtles' in the wild by reducing their reproduction, numbers, or distribution.

5.13 Sperm Whale

As discussed in the Status of Listed Resources section, while they are the most abundant of the large whale species, the sperm whale is listed as endangered. The most recent estimate indicated a global population of between 300,000 and 450,000 individuals (Whitehead 2002; NMFS 2015c). In the North Pacific, Barlow and Taylor (2005) estimate that there are between 26,300 and 32,100 sperm whales in the temperate Northeast portion of the Pacific Ocean. There are no trend assessments for sperm whales in the Pacific but Moore and Barlow (2017) detected a non-significant increasing trend for eastern North Pacific sperm whales. Three human activities are known to threaten sperm whales: whaling, entanglement in fishing gear, and shipping. Historically, whaling represented the greatest threat to every population of sperm whale and was ultimately responsible for listing sperm whales as an endangered species.

As described in the Environmental Baseline, effects of climate change on marine species include alterations in reproductive seasons and locations, shifts in migration patterns, reduced distribution and abundance of prey, and changes in the abundance of competitors or predators. However, information specific to the response to climate change by this species in the Action Area is uncertain at this time. Many effects from the stress regimes created by the Environmental Baseline are unknown for this particular species although some acoustic stressors may result in behavioral disturbances (temporarily disrupted foraging, avoidance, cessation of vocal behavior) but is not quantifiable in the Action Area at this time. To date, there has been 1 interaction with a sperm whale in United States fisheries that occur within the Action Area during the time period considered in this analysis. Overall, two individuals have been captured historically. It's estimated that a total of 6 interactions accounting for observer coverage (NMFS 2018b) may have occurred from 2004 to 2021 based on the observation in 2011.

As discussed in the Exposure section, only one interaction has been observed in the DSLL, which occurred in 2011, with a supplemental record of one additional interaction from a NMFS gear testing cruise in 2002. Based on the evidence available, an average of one every other year, but up to 3 sperm whales may be entangled in longline gear associated with the DSLL fishery each year. As discussed in the Response section, the evidence available leads us to conclude that the overwhelming majority of these whales are likely to free themselves from entangling line, although some of these whales may swim away with trailing line. Again, we recognize that the sample size is extremely low. While sperm whales often depredate other longline fisheries, the information to date reveals only entanglements have occurred in this fishery. In a small percentage of captures, sperm whales may need human help to be disentangled from gear. The evidence available does not indicate that sperm whales might die or experience changes in their migratory, reproductive, or other social behavior as a result of being entangled by the DSLL.

Given their size we consider it very unlikely any sperm whales will die as a result of their interaction with the DSLL. Muto et al. (2021) suggests no M/SI for sperm whales in Hawaii based fisheries, given the reported interaction with the fishery from 2011 and we agree that no mortalities are anticipated for this fishery. As discussed in the Risk section, up to 3 captures per year with no expected mortalities would not change the status or trend of North Pacific Ocean sperm whales with a minimum abundance estimate of 26,300 individuals over the next 40 years.

Based on our analyses, the DSLL fishery may interact with an average of one sperm whale every other year and up to three in a year, and would be expected to result in entanglements in longline gear associated with this fishery. However, with rare exceptions, behavior modifications as a result of an interaction is unlikely to reduce the longevity, reproductive success, or social ecology of those whales given their size and ability to free themselves from entanglement. Because of this, and that we are reasonably certain no sperm whales will die from the fishery, the DSLL will not cause material changes having biological consequences to the species' numbers, reproduction, or distribution. As a result, NMFS does not anticipate the DSLL fishery will reduce appreciably the likelihood of the survival or recovery of sperm whales in the wild by reducing their reproduction, numbers, or distribution.

5.14 Main Hawaiian Island Insular False Killer Whale

As discussed in the Status of Listed Resources section, NMFS currently recognizes five Pacific Islands Region management populations including the MHI insular DPS which is listed as endangered (77 FR 70915, 28 November, 2012). Badger et al. (in review) estimated the population to be 138 (95% CRI=120–160) with a declining annual population growth rate of -3.51 (95% CRI: -8.40, 2.04) over the entire time series considered (1999 to 2021) and -5.53, (95% CRI = -9.91, -1.61) over the past 10 years. Badger et al. (in review) also found that abundance appeared to vary by cluster, with the largest subset of the population being members of cluster 1 and fewest in cluster 4. Hooking, entanglement, or intentional harm caused by fishermen, competition for food with commercial fisheries (Boggs and Ito 1993; Reeves et al. 2009), exposure to environmental contaminants (Ylitalo et al. 2009), and small population size are identified as the most substantial threats to the population (Carretta et al. 2019).

The implementation of the FKWTRP since 2013 has benefitted the MHI IFKW. Specifically, the elimination of the seasonal contraction of the MHI Longline Fishing Prohibited Area effectively protects MHI IFKW from being exposed to the majority of the DSLL fishery. Currently 5.4% of the DPS' range overlaps with the DSLL fishery, accounting for approximately 4,200 square miles where interactions with the DPS may still occur.

As described in the Environmental Baseline, effects of climate change on marine species include alterations in reproductive seasons and locations, shifts in migration patterns, reduced distribution and abundance of prey, and changes in the abundance of competitors or predators. However, information specific to the response to climate change by this DPS in the Action Area is uncertain at this time. Many effects from the stress regimes created by the Environmental Baseline are unknown for this particular DPS although some acoustic stressors may result in behavioral disturbances (temporarily disrupted foraging, avoidance, cessation of vocal behavior) but is not quantifiable in the Action Area at this time.

To date, while interactions occur with other MMPA stocks of false killer whales (not ESA listed) in both the SSLL and DSLL fisheries within the Action Area, no observed interactions have occurred with the MHI IFKW in either fishery. However, the MHI insular population has a high rate of dorsal fin disfigurements and mouthline injuries consistent with injuries from unidentified fishing lines (Baird and Gorgone 2005; Baird et al. 2014, 2017; Beach et al. 2015; Baird 2019). We note, the fisheries which are responsible for these injuries (whether commercial or recreational) are unknown. The rate of scarring combined with the location density map (Figure 28) and other evidence like hook types that have been recovered in the stomach of a small number of whales suggest that state fisheries are the predominant cause of these injuries.

As described in the Exposure section, from 2013 to 2021, approximately 8% of set in the zone of overlap were observed (NMFS unpublished data). Our estimates suggest that the annual probability of capturing a MHI IFKW is small, and these small probabilities accumulate enough to suggest that on average, 1 capture in 12 to 23 years is expected. Capture may be more frequent as indicated by the 95th percentile.

As described in the Response section, those interactions are expected to result in the mortality and serious injury of some MHI IFKW. We would expect one MHI IFKW's would die or be seriously injured approximately every 14 to 30 years but it may be more frequently as indicated by the 95th percentile

As described in the Risk section, the small numbers that comprise this DPS place it at a high risk from just a small number of mortalities. Based on the evidence available, one MHI IFKW may be entangled or hooked in longline gear associated with the DSLL fishery an average of every 12 years based on the maximum 5-yr running average. The evidence available leads us to conclude that some of the captured whales are likely to free themselves from entangling line, although some of these whales may swim away with trailing line. In a small percentage of captures, MHI IFKW will need human help to be disentangled from gear. The evidence available allows us to estimate that one serious injury or mortality may occur as a result of being entangled and/or hooked on average every 14 years based on the maximum 5-yr running average.

The mortality of 0.073 individuals per year represents 0.025% of the population. Assuming the current population trend includes mortalities from the DSLL, if we remove these mortalities, from 2022 to 2023, the rate of population decline would be reduced from 5.53% per year to 5.47% per year. From 2022 to 2062 the rate of population decline would be reduced to 5.33% per year. Given the estimated standard deviation for the population growth rate (0.018), and based on a power analysis for detecting differences in means with the same variance (Cohen 1988), more than 1,400 years of data would be required to detect a statistically significant difference in these values. While the MHI IFKW are declining, incidental interactions with the DSLL fishery are unlikely to change the overall trajectory of the population and are likely to be within the variability of any calculated trend and would not be statistically detectable with our predictive capabilities. Therefore, we are reasonably certain the DSLL will not cause material changes having biological consequences to the DPS' numbers, reproduction, or distribution. In accordance with Section 1.2.2 (Jeopardy Analyses) above. NMFS does not anticipate the DSLL fishery will reduce appreciably the likelihood of the survival or recovery of MHI IFKWs in the wild by reducing their reproduction, numbers, or distribution. However, we highlight the limited information on potential interactions in the DSLL with this fishery due to the low observer coverage and emphasize the need to fill this data gap.

6 CONCLUSION

After reviewing the Status of Listed Resources, the Environmental Baseline for the Action Area, the effects of the proposed action, and the cumulative effects, it is NMFS' biological opinion that SFD's authorization to operate the DSLL fishery as currently managed is *not likely* to jeopardize the continued existence of the following:

Threatened giant manta ray, threatened Indo-West Pacific DPS scalloped hammerhead shark; threatened oceanic whitetip shark; threatened Central North Pacific, East Indian-West Pacific, East Pacific, and Southwest Pacific DPSs of green sea turtles; endangered Central West Pacific and Central South Pacific DPSs of green sea turtles; endangered leatherback sea turtles and

North Pacific DPS loggerhead sea turtles; threatened and endangered populations of olive ridley sea turtles, endangered sperm whale, and endangered main Hawaiian Islands insular DPS false killer whale.

7 INCIDENTAL TAKE STATEMENT

Section 9 of the ESA and protective regulations pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species without a special exemption. "Incidental take" is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. 50 CFR 402.02. Under the terms of section 7(b)(4) and section 7(o)(2), taking that is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the ESA provided that such taking is in compliance with the reasonable and prudent measures and terms and conditions of the Incidental Take Statement.

The proposed action results in the incidental take of threatened giant manta ray; threatened Indo-West Pacific scalloped hammerhead shark; threatened oceanic whitetip shark; threatened East Pacific, Central North Pacific, East Indian-West Pacific, and Southwest Pacific green sea turtles; endangered Central West Pacific and Central South Pacific green sea turtles; endangered leatherback sea turtles, endangered North Pacific loggerhead sea turtles; threatened olive ridley sea turtles; endangered sperm whales, and main Hawaiian Island insular false killer whales.

We have not promulgated an ESA section 4(d) rule prohibiting take of threatened oceanic white tip sharks, Indo-West Pacific scalloped hammerhead sharks, or giant manta ray, so an exemption from the take prohibitions of section 9 of the ESA is neither necessary nor appropriate for these species. Consistent with the decision in *Center for Biological Diversity v. Salazar*, 695 F.3d 893 (9th Cir. 2012), we have included an incidental take statement to serve as a check on the no-jeopardy conclusion by providing a reinitiation trigger if the level of take analyzed in the biological opinion is exceeded. In addition, 50 CFR 402.14(i)(3) provides that in order to monitor the impacts of incidental take, "the Federal agency or any applicant must report the progress of the action and its impact on the species to the Service as specified in the incidental take statement."

The measures described below are nondiscretionary, and must be undertaken by NMFS for the exemption in section 7(o)(2) to apply. NMFS has a continuing duty to regulate the activity covered by this incidental take statement. If NMFS fails to assume and implement the terms and conditions, the protective coverage of section 7(o)(2) may lapse. In order to monitor the impact of incidental take, NMFS must monitor the progress of the action and its impact on the species as specified in the incidental take statement (50 CFR 402.14(i)(3)).

7.1 MMPA Authorization

A marine mammal species or population stock that is listed as threatened or endangered under the ESA is, by definition, also considered depleted under the MMPA. The ESA allows takings of threatened and endangered marine mammals only if authorized by section 101(a)(5) of the MMPA. Section 101(a)(5)(E) of the MMPA, 16 U.S.C. 1361 *et seq.*, has provisions for NOAA's NMFS, as delegated by the Secretary of Commerce, to issue permits for the taking of marine mammals designated as depleted because of their listing under the ESA, 16 U.S.C. 1531 *et seq.*, by U.S. vessels and those vessels which have valid fishing permits issued by the Secretary in accordance with section 204(b) of the Magnuson-Stevens Fishery Conservation and Management Act, 16 U.S.C. 1824(b), for a period of up to three years. NMFS may issue the authorization to take Endangered and threatened marine mammals incidental to these commercial fisheries only after the agency has determined, after notice and opportunity for public comment, that:

(1) the incidental mortality and serious injury from commercial fisheries will have a negligible impact on the affected species or stock;

(2) a recovery plan has been developed or is being developed for such species or stock under the ESA; and

(3) where required under section 118 of the MMPA, a monitoring program has been established, vessels engaged in such fisheries are registered in accordance with section 118 of the MMPA, and a take reduction plan has been developed or is being developed for such species or stock.

Further, when an action will result in incidental take of endangered and threatened marine mammals, ESA section 7(b)(4)(C) requires that such taking be authorized pursuant to section 101(a)(5) of the MMPA. Section 7(b)(4)(C)(iii) requires the ESA incidental take statement to specify those measures that are necessary to comply with section 101(a)(5) of the MMPA. Section 7(b)(4) and section 7(o)(2) provide that taking that is incidental to an otherwise lawful agency action is not considered to be prohibited taking under the ESA if that action is performed in compliance with the terms and conditions of this incidental take statement, including those specified as necessary to comply with the MMPA, section 101(a)(5). NMFS determined the DSLL fishery will have a negligible impact on ESA-listed marine mammal species for the purposes of issuing a permit under MMPA (Docket 2020-0096). Correspondingly, NMFS issued an MMPA permit covering the DSLL effective for a three-year period beginning May 6, 2021(86 FR 24384).

7.2 Amount or Extent of Take

The following levels of incidental take may be expected to result from the proposed action. The reasonable and prudent measures, with their implementing terms and conditions, are designed to minimize the impact of incidental take that might otherwise result from the proposed action. NMFS uses causal inference to determine if individual threatened and endangered species, or their designated critical habitat, would likely be taken by harassing, harming, pursuing, hunting, shooting, wounding, killing, trapping, capturing, or collecting or attempting to engage in any such conduct. If take is anticipated to occur then the Services must describe the amount or extent of such anticipated take and the reasonable and prudent measures, and terms and conditions necessary to minimize the impacts of incidental take (FWS and NMFS 1998). If, during the course of the action, any level of incidental take is exceeded for any of the species as listed, NMFS SFD must immediately reinitiate formal consultation with NMFS PRD pursuant to the section 7 regulations (50 CFR 402.16). NMFS PRD anticipates that the following ESA-listed species and DPS' could be taken as a result of the proposed action by capturing, harming, wounding, and/or killing: giant manta ray, Indo-West Pacific scalloped hammerhead shark, oceanic whitetip shark, Central North Pacific green sea turtle, Central South Pacific green sea turtle, Central West Pacific green sea turtle, East Pacific green sea turtle, East Indian-West Pacific green sea turtle, Southwest Pacific green sea turtle, leatherback sea turtle, North Pacific loggerhead sea turtle, olive ridley sea turtle (both endangered and threatened populations), sperm whale, and Main Hawaiian Islands Insular false killer whale.

We anticipate the amount of take from the DSLL fishery in Table 54 below. The number of takes occurring annually is highly variable due to fluctuations in fishery target-species distribution, ESA-listed species distribution and abundance, fishing effort, sea surface temperatures, ocean currents, and other factors that are difficult to predict. As a result, using the estimated mean annual take levels as reinitiation triggers is not reasonable or practical. Warden et al. (2015) state "when the population is large compared to the incidental mortality, frequent (e.g., annual) monitoring is not likely to produce results that are substantially different from the previous assessment. Less frequent but more comprehensive assessments, which explicitly address uncertainty, may provide more reliable information." For these reasons, and based on our experience monitoring fisheries, we believe that the maximum 5-year running sum is most appropriate metric for meaningful tracking of take with respect to the ITS.

Year to year variation in capture numbers is expected, and managing the incidental take by the 5-year running sum accounts for this annual variation, allowing for years with higher than average captures and years with lower than average captures. Exceeding the maximum 5-year running sum (Table 54) over any five consecutive years is a reinitiation trigger. This does not imply we will wait five years to assess take. Observed interactions are documented in real time and uploaded to our internal database. The database is continually monitored by NMFS SFD and PRD. Therefore, we will know of any exceedance of the numbers regardless of when it happens during the 5-year period. As an example, we will trigger reinitiation if 15 Indo-West Pacific scalloped hammerhead sharks are caught within say a 2 or 3-year period.

Table 54. Anticipated take for the DSLL fishery. These are the estimated number of captures, expanded from the observed number of incidental captures and inclusive of prorated numbers based on unidentified captures.

Species	Maximum 5-Year Running Sum
Giant manta ray	181
Indo-West Pacific scalloped hammerhead sharks	14
Oceanic whitetip shark	10,589

Species	Maximum 5-Year Running Sum
Green sea turtle (all species)	77
Leatherback sea turtle	92
North Pacific Loggerhead	433
Olive ridley sea turtle (all populations)	592
Sperm whale	6
Main Hawaiian insular false killer whale	0.427 (1 observed)

7.3 Reasonable and Prudent Measures

"Reasonable and prudent measures" are nondiscretionary measures that are necessary or appropriate to minimize the impact of the amount or extent of incidental take (50 CFR 402.02). NMFS has determined that the following reasonable and prudent measures, as implemented by the terms and conditions that follow, are necessary and appropriate to minimize the impacts of the Hawaii DSLL fishery, as described in the proposed action, on threatened and endangered species and to monitor the level and nature of any incidental takes. These measures are non-discretionary—they must be undertaken by NMFS for the exemption in ESA section 7(o)(2) to apply.

1. NMFS shall require that ESA-listed species incidentally caught alive be released from fishing gear in a manner that minimizes injury and the likelihood of further gear entanglement or entrapment to increase their post-release survivorship.
2. NMFS shall ensure that the Hawaii DSLL fishery has a monitoring and reporting program sufficient to confirm that extent of take is not exceeded, and that the terms and conditions in this incidental take statement are effective in minimizing incidental take.

7.4 Terms and Conditions

NMFS SFD and its applicant, the Hawaii Longline Association, shall undertake and comply with the following terms and conditions to implement the reasonable and prudent measures identified in Section 8.3 above. These terms and conditions are non-discretionary, and if NMFS and HLA fail to adhere to these terms and conditions, or if NMFS fails to implement measures requiring the Applicant to comply with these terms and conditions, the protective coverage of section 7(o)(2) may lapse.

1. The following terms and conditions implement Reasonable and Prudent Measure No. 1:
    a. NMFS SFD shall implement measures to minimize the amount of trailing gear left on ESA-listed species to the maximum extent practicable to reduce the amount of injury and harm, the likelihood of further gear entanglement or entrapment, and improve the post-release mortality of ESA-listed species. This may include using new technologies once proven, such as fighting lines (i.e., additional gear that reduces tension on the branchline), line cutting devices, and other gear modifications.
    b. To reduce post-release mortality, within two years NMFS SFD shall require species handling training for crew members and at a minimum have one trained person on deck who directs and oversees activities of the vessel when retrieving fishing gear. Training shall include best practices identified in 1.a above.
2. The following terms and conditions implement Reasonable and Prudent Measure No. 2:
    a. NMFS SFD shall maintain observer coverage at levels reliable for estimating protected species interaction rates onboard Hawaii DSLL vessels. This may include electronic monitoring.
        i. NMFS SFD shall collect standardized information regarding the incidental capture, injury, and mortality of ESA-listed marine species for each interaction by species, gear, and set information, as well as the presence or absence of tags on these species.
        ii. NMFS SFD shall improve length estimates of leatherback sea turtles that are not boarded. These lengths will be recorded in a specified field rather than only noted in the comment field.
        iii. To the maximum extent practicable, observers shall identify the hooking location for every interaction, and estimate the length of any trailing gear left on ESA-listed species at release when those species cannot be boarded. These data are intended to allow NMFS to improve estimates of harm, injury, and mortalities within this fishery.
        iv. The observers shall document the method or technology used to release all ESA-listed species.
        v. To improve NMFS' understanding of and estimates of interactions with MHI IFKW in the overlap area, within one year NMFS will determine the minimum level of observer coverage reliable for estimating MHI FKW interactions with the DSLL vessels. If the current level of observer coverage is below this level, within two years provide observer coverage at the level determined reliable.
    b. NMFS SFD shall collaborate with NMFS PRD to develop annual data products to be provided to NMFS PRD detailing levels of captures, updates to interaction

        mortality rates and locations of fishing effort and subsequent captures. These reports will be provided to NMFS PRD once all data from the previous year have been finalized but not later than July of the following year. The report from the first year will include all data for calendar year 2023, as well as the period from the date this opinion was signed to the end of the year. Only the take that occurs post-signature is exempted by this opinion, but the calendar year data will be used to ensure incidental captures are within the expectations of our analysis.

    c. Every five years after signing of this opinion, NMFS SFD shall collaborate with NMFS PRD to evaluate the data:
        i. Use the data from 2.a.i to explore ecosystem and area based management techniques with potential to reduce interactions with ESA-listed species in the future. Examples include time area closures, rotational zone management, dynamic closures, hotspot analyses, and move-on methods, etc. and should include review of the SEZ as currently designed.
        ii. Evaluate the effectiveness of methods designed to reduce post release mortality in ESA-listed species and make improvements or incorporate changes as warranted.

7.5 Conservation Recommendations

Section 7(a)(1) of the Act directs Federal agencies to utilize their authorities to further the purposes of the Act by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information.

In order to keep NMFS PRD informed of actions minimizing or avoiding adverse effects or benefitting listed species or their habitats, NMFS SFD should notify NMFS PRD in writing upon initiating any of these conservation recommendations in their final action.

1. NMFS should continue to improve its understanding of the effect of recent gear changes in the Hawaii DSLL fishery on the capture, injury, harm and mortality of ESA-listed species. Specifically, NMFS should:
    a. Monitor the amount of trailing gear that is left on ESA-listed species, as well as species used to calculate at-vessel mortality estimates of ESA-listed species (e.g. pelagic false killer whales and unidentified sharks and rays) to understand fishery performance with this management measure and improve understanding of the effects of the Hawaii DSLL fishery on these species.
    b. NMFS SFD should implement measures for best handling practices for release methods to enhance survivorship for giant manta ray as recommended by Hutchinson et al. (2017) and WCPFC (2017), and as new information becomes available.
    c. Implement best handling practices for release methods to enhance survivorship.

2. NMFS SFD should develop and implement a tagging program to examine the genetic profile of giant manta rays captured in the fishery to better inform management and recovery goals for these species. This should explore site use and residency patterns of giant manta rays released alive in the fishery and to examine post release mortality metrics pertinent to this specific fishery. This data would help to clarify what ray species is being captured (*M. alfredi* vs. *M. birostris*), and where these species are being captured in the fishery in order to avoid these areas by reducing potential interactions in the future.
3. NMFS should promote studies on ecology, habitat use, fecundity, genetics, and post interaction survivability of ESA-listed marine species.
4. NMFS should explore how climate change, including ocean warming, may affect habitat quality, prey abundance and distribution, and the physiological ecology (e.g., thermal tolerance) of the ESA-listed species considered herein.
5. NMFS should encourage RFMOs to require reporting of oceanic whitetip catches and discards, and for Parties to increase reporting of oceanic whitetip catch and disposition to improve data quality and quantify the impact of fishing on the species.
6. NMFS should enhance bilateral cooperation and engagement with key countries that have large international longline fleets to promote conservation and recovery of leatherback sea turtles, oceanic whitetip sharks, and other ESA-listed species considered herein.
7. NMFS should enhance capacity building among the international fishing community for increasing data collection and information sharing.
8. NMFS should develop an outreach and education campaign for the public to increase awareness of the status and importance of ESA-listed species, while incorporating cultural insights and perspectives from various regions/locations of the species' range.
9. NMFS should explore implementing an Electronic Monitoring program in the fishery to improve our understanding of interactions with ESA-listed species and reduce uncertainty when those interactions occur. At minimum, NMFS should consider a voluntary EM selection pool with a random strata selection that maintains fleet-wide observer coverage in accordance with term and condition 2a.
    a. Explore cost share and assistance programs for EM installations, explore feasibility, etc.
    b. Consider reviewing and updating an Annual Deployment Plan for the Region including an EM selection pool or fleet wide deployment.
    c. Continue to coordinate with other NMFS Regions where EM technologies were successfully implemented for comparable fisheries and vessel class.

In order to keep NMFS' PRD informed of actions minimizing or avoiding adverse effects or benefitting listed species or their habitats, SFD should notify the PRD in writing upon initiating any of these conservation recommendations in their final action.

7.6 Reinitiation Notice

This concludes formal consultation on the continued operation of the Hawaii DSLL fishery. As provided in 50 CFR 402.16, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained or is authorized by law, and if:

1. The amount or extent of incidental take for any species is exceeded;
2. New information reveals effects of the agency action that may affect listed species or critical habitat in a manner or to an extent not considered in this opinion;
3. The agency action is subsequently modified in a manner that may affect listed species or critical habitat to an extent in a way not considered in this opinion; or
4. A new species is listed or critical habitat designated that may be affected by the action.

8 LITERATURE CITED

Abreu-Grobois A, and P. Plotkin. 2008. IUCN SSC Marine Turtle Specialist Group. *Lepidochelys olivacea*. 2008:e. T11534A3292503. http://dx.doi.org/10.2305/IUCN.UK.2008.RLTS.T11534A3292503.en.

Acevedo-Whitehouse, K., and A. L. Duffus. 2009. Effects of environmental change on wildlife health. Philosophical Transactions of the Royal Society B: Biological Sciences. 364(1534):3429-3438.

Acevedo-Gutierrez, A., B. Brennan, P. Rodrigues, and M. Thomas 1997. Resightings and behavior of false killer whales (*Pseudorca crassidens*) in Costa Rica. Marine Mammal Science. 13(2):307-314.

Adams, D. H., and E. Amesbury. 1998. Occurrence of the manta ray, *Manta birostris*, in the Indian River Lagoon, Florida. Florida Scientist.7-9.

Adams, J., M. Sweezey, and P. V. Hodson. 2014. Oil and oil dispersant do not cause synergistic toxicity to fish embryos. Environmental Toxicology and Chemistry. 33(1):107-114.

Agamy, E. 2013a. Impact of laboratory exposure to light Arabian crude oil, dispersed oil and dispersant on the gills of the juvenile brown spotted grouper (*Epinephelus chlorostigma*): a histopathological study. Marine Environment Research. 86:46-55.

Agamy, E. 2013b. Sub chronic exposure to crude oil, dispersed oil and dispersant induces histopathological alterations in the gills of the juvenile rabbit fish (*Siganus canaliculatus*). Ecotoxicology and Environmental Safety. 92:180-190.

Aguilar, R., J. Mas, and X. Pastor. 1995. Impact of Spanish swordfish longline fisheries on the loggerhead sea turtle *Caretta caretta* population in the western Mediterranean. In: Richardson, J. I., T. H. Richardson, editors. Proceedings of the Twelfth Annual Workshop on Sea Turtle Biology and Conservation, 1995. p. 1-6. NOAA Tech Memo NMFS-SEFSC-361.

Aguirre, A.A., T.R. Spraker, A. Chaves, L. DuToit, W. Eure, and G.H. Balazs. 1999. Pathology of fibropapillomatosis in olive ridley sea turtles, Lepidochelys olivacea, nesting in Costa Rica. Journal of Aquatic Animal Health 11:283-289.

Alefaio, S.T., Alefaio, A.R., and A. Resture. 2006. Turtle monitoring on Funafuti, Tuvalu, 4th December-14th December Suva. p. 8.

Alfaro-Shigueto, J., Dutton, P.H., Van Bressem, M.F., and J. Mangel. 2007. Interactions between leatherback turtles and Peruvian artisanal fisheries. Chelonian Conservation and Biology. 6(1):129-134.

Alfaro-Shigueto, J., Mangel, J.C., Bernedo, F., Dutton, P.H., Seminoff, J.A., and B.J. Godley. 2011. Small-scale fisheries of Peru: a major sink for marine turtles in the Pacific. Journal of Applied Ecology. 48(6):1432-1440.

Alfaro-Shigueto, J., Mangel, J.C., and P.H. Dutton. 2006. Kinan I, editor. Loggerhead turtle bycatch in Peru. Proceedings of the Second Western Pacific Sea Turtle Cooperative

Research & Management Workshop; 2006: Western Pacific Regional Fishery Management Council, Honolulu, Hawaii.

Allen, C.D., Lemons, G.E., Eguchi, T., LeRoux, R.A., Fahy, C.C,. Dutton, P.H., Peckham, S.H., and J.A. Seminoff. 2013. Stable isotope analysis reveals migratory origin of loggerhead turtles in the Southern California Bight. Marine Ecology Progress Series. 472:275-285.

Allen, C.D., Robbins, M.N., Eguchi, T., Owens, D.W., Meylan, A.B., Meylan, P.A., Kellar, N.M., Schwenter, J.A., Nollens, H.H., LeRoux, R.A. *et al*. 2015. First Assessment of the Sex Ratio for an East Pacific Green Sea Turtle Foraging Aggregation: Validation and Application of a Testosterone ELISA. PLoS One. 10(10):e0138861.

Abercrombie, D. L., S. C. Clarke, and M. S. Shivji. 2005. Global-scale genetic identification of hammerhead sharks: Application to assessment of the international fin trade and law enforcement. Conservation Genetics. 6(5):775-788.

Amano, M., and M. Yoshioka. 2003. Sperm whale diving behavior monitored using a suction-cup-attached TDR tag. Marine Ecology Progress Series. 258:291-295.

Anderson, R.C., Adam, M.S., and J. I. Goes. 2011. From monsoons to mantas: seasonal distribution of *Manta alfredi* in the Maldives. Fisheries Oceanography. 20(2):104-113.

Aoki, K., M. Amano, M. Yoshioka, K. Mori, D. Tokuda, and N. Miyazaki. 2007. Diel diving behavior of sperm whales off Japan. Marine Ecology Progress Series. 349:277-287.

Arenas, P., and M. Hall. 1992. The association of sea turtles and other pelagic fauna with floating objects in the eastern tropical Pacific Ocean. In: Salmon M, Wyneken J, editors. Proceedings of the Eleventh Annual Workshop on Sea Turtle Biology and Conservation. US Dep. Comm., NOAA Tech. Memo. NMFS-SEFSC-302. p. 7-10. https://repository.library.noaa.gov/view/noaa/6067

Arendt, M. D., J. A. Schwenter, B. E. Witherington, A. B. Meylan, and V. S. Saba. 2013. Historical versus Contemporary Climate Forcing on the Annual Nesting Variability of Loggerhead Sea Turtles in the Northwest Atlantic Ocean. PLoS One. 8(12).

Ashford, J. R., P. S. Rubilar, and A. R. Martin. 1996. Interactions between cetaceans and longline fishery operations around South Georgia. Marine Mammal Science. 12(3):452-457.

Avens, L., Goshe, L.R., Zug, G.R., Balazs, G.H., Benson, S.R., and H. Harris. 2020. Regional comparison of leatherback sea turtle maturation attributes and reproductive longevity. Marine Biology. 167(1):4.

Azanza-Ricardo, J., Martín, M.E.I., Sansón, G.G., Harrison, E., Cruz, Y.M., and F. Bretos. 2017. Possible Effect of Global Climate Change on *Caretta caretta* (Testudines, Cheloniidae) Nesting Ecology at Guanahacabibes Peninsula, Cuba. Chelonian Conservation and Biology. 16(1):12-19.

Backus, R.H., Springer, S., and E.L. Arnold Jr. 1956. A contribution to the natural history of the white-tip shark, *Pterolamiops longimanus* (Poey). Deep Sea Research (1953). 3(3):178-188.

Balcıoglu, E. B. 2016. Potential effects of polycyclic aromatic hydrocarbons (PAHs) in marine foods on human health: a critical review. Toxin Reviews. 35(3-4):98-105.

Baird, R. W. 2009. A review of false killer whales in Hawaiian waters: biology, status, and risk factors. U.S. Marine Mammal Commission. 41 p.

Baird, R. W. 2019. Evidence of fisheries interactions with Hawaiian insular odontocetes. Cascadia Research Collective. 31 p.

Baird, R. W., and A. M. Gorgone. 2005. False killer whale dorsal fin disfigurements as a possible indicator of long-line fishery interactions in Hawaiian waters. Pacific Science. 59(4):593-601.

Baird, R. W., A. M. Gorgone, D. L. Webster, D. J. McSweeney, J. W. Durban, A. D. Ligon, D. R. Salden, and M. H. Deakos. 2005. False killer whales around the main Hawaiian Islands: an assessment of inter-island movements and population size using individual photo-identification. Contract Report JJ133F04SE0120 prepared for the Pacific Islands Fisheries Science Center, National Marine Fisheries Service 2570. 24 p.

Baird, R. W., D. Cholewiak, D. L. Webster, G. S. Schorr, S. D. Mahaffy, C. Curtice, J. Harrison, and S. M. Van Parijs. 2015. 5. Biologically Important Areas for Cetaceans Within U.S. Waters – Hawai‘i Region. Aquatic Mammals. 41(1):54-64.

Baird, R. W., A. M. Gorgone, D. J. McSweeney, D. L. Webster, D. R. Salden, M. H. Deakos, A. D. Ligon, G. S. Schorr, J. Barlow, and S. D. Mahaffy. 2008. False killer whales (*Pseudorca crassidens*) around the main Hawaiian Islands: Long-term site fidelity, inter-island movements, and association patterns. Marine Mammal Science. 24(3):591-612.

Baird, R. W., Hanson, M. B., Schorr, G. S., Webster, D. L., McSweeney, D. J., Gorgone, A. M., Mahaffy, S. D., Holzer, D. M., Oleson, E. M., and R. D. Andrews. 2012. Range and primary habitats of Hawaiian insular false killer whales: informing determination of critical habitat. Endangered Species Research. 18(1):47-61.

Baird, R. W., S. D. Mahaffy, A. M. Gorgone, T. Cullins, D. J. McSweeney, E. M. Oleson, A. L. Bradford, J. Barlow, and D. L. Webster. 2014. False killer whales and fisheries interactions in Hawaiian waters: Evidence for sex bias and variation among populations and social groups. Marine Mammal Science. 31(2):579-590.

Baird, R. W., S. D. Mahaffy, A. M. Gorgone, K. A. Beach, T. Cullins, D. J. McSweeney, D. S. Verbeck, and D. L. Webster. 2017. Updated evidence of interactions between false killer whales and fisheries around the main Hawaiian Islands: assessment of mouthline and dorsal fin injuries. PSRG-2017-16. 8 p.

Baird, R. W., Schorr, G. S., Webster, D. L., McSweeney, D. J., Hanson, M. B., and R. D. Andrews. 2010. Movements and habitat use of satellite-tagged false killer whales around the main Hawaiian Islands. Endangered Species Research. 10:107-121.

Baird, R. W., C. J. Cornforth, S. D. Mahaffy, J. K. Lerma, A. E. Harnish, and M. A. Kratofil. 2023. Field studies and analyses from 2020 through 2022 to support the cooperative conservation and long-term management of main Hawaiian Islands insular false killer whales. Final report under State of Hawai‘i contract 68819 issued to Cascadia Research

Collective 218 W 4th Avenue Olympia, WA 98506. 43 p.Baker, J. D., Littnan, C. L., and D. W. Johnston. 2006. Potential effects of sea level rise on the terrestrial habitats of endangered and endemic megafauna in the Northwestern Hawaiian Islands. Endangered Species Research. 2:21-30.

Balazs, G. H. 1975. Green turtle’s uncertain future. Defenders. 50(6):521-523.

Balazs, G. H. 1976. Green turtle migrations in the Hawaiian Archipelago. Biological Conservation. 9(2):125-140.

Balazs, G. H. 1983. Sea turtles and their traditional usage in Tokelau. Atoll Research Bulletin. 279.

Balazs, G. H. 1985. Impact of ocean debris on marine turtles: Entanglement and Ingestion. Honolulu Hawaii: U.S. Department of Commerce. p. 387-429.

Balazs, G. H., and M. Chaloupka. 2004. Thirty-year recovery trend in the once depleted Hawaiian green sea turtle stock. Biological Conservation. 117(5):491-498.

Balazs, G. H., Forsyth, R. G., and A. K. H. Kam. 1987. Preliminary assessment of habitat utilization by Hawaiian green turtles in their resident foraging pastures. NOAA Technical memorandum, NOAA-TM-NMFS-SWFC-71. p. 116. https://repository.library.noaa.gov/view/noaa/5758

Balazs, G. H., Pooley, S. G., and S. K. K. Murakawa. 1995a. Guidelines for handling marine turtles hooked or entangled in the Hawaii longline fishery: Results of an expert workshop held in Honolulu, Hawaii, March 15-17, 1995. US Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center. p. 46.

Balazs, G. H., Siu, P., and J.-P. Landret. 1995b. Ecological aspects of green turtles nesting at Scilly Atoll in French Polynesia. In: Richardson, JI and TH Richardson (compilers), Proceedings of the Twelfth Workshop on Sea Turtle Biology and Conservation. NOAA Tech. Memo. NMFS-SEFSC-361. p. 7-10. https://repository.library.noaa.gov/view/noaa/8713

Balazs, G. H., Van Houtan, K. S., Hargrove, S. A., Brunson, S. M., and S. K. K. Murakawa. 2015. A Review of the Demographic Features of Hawaiian Green Turtles (*Chelonia mydas*). Chelonian Conservation and Biology. 14(2):119-129.

Bannister, J. L., and E. Mitchell. 1980. North Pacific sperm whale stock identity: distributional evidence from Maury and Townsend charts. Reports of the International Whaling Commission Special Issue. 2:219-230.

Barlow, H, and B. L. Taylor. 2005. Estimates of sperm whale abundance in the Northeastern temperate Pacific from a combined acoustic and visual survey. Marine Mammal Science 21:429-445.

Barlow, J., and S. Rankin. 2007. False killer whale abundance and density: Preliminary estimates for the PICEAS study area south of Hawaii and new estimates for the US EEZ around Hawaii.

Barr, J. M. 2006. Community based Sea Turtle Monitoring and Management at Helen Reef, Hatohobei State, Republic of Palau. Oregon State University. p. 216.

Barraza, A. D., L. M. Komoroske, C. D. Allen, T. Eguchi, R. Gossett, E. Holland, D. D. Lawson, R. A. LeRoux, V. Lorenzi, and J. A. J. M. p. b. Seminoff. 2020. Persistent organic pollutants in green sea turtles (*Chelonia mydas*) inhabiting two urbanized Southern California habitats. 153:110979.

Baulch, S., and C. Perry. 2014. Evaluating the impacts of marine debris on cetaceans. Marine Pollution Bulletin. 80(1-2):210-221.

Baum, J., Medina, E., Musick, J. A., and M. Smale. 2015. *Carcharhinus longimanus*. The IUCN Red List of Threatened Species 2015: e.T39374A85699641. (Downloaded on 30 June 2018). http://dx.doi.org/10.2305/IUCN.UK.2015.RLTS.T39374A85699641.en.

Baum, J. K., and R. A. Myers. 2004. Shifting baselines and the decline of pelagic sharks in the Gulf of Mexico. Ecology Letters. 7(2):135-145.

Bayless, A.R., E.M. Oleson, S. Baumann-Pickering, A.E. Simonis, J. Marchetti, S. Martin, and S.M. Wiggins. 2017. Acoustically monitoring the Hawai'i longline fishery for interactions with false killer whales. Fisheries research. 190: 122-131.

Beach, K. A. 2015. Mouthline injuries as an indicator of fisheries interactions in Hawaiian Odontocetes. Evergreen State College. p. 57.

Beale, C. S., J. D. Stewart, E. Setyawan, A. B. Sianipar, M. V. Erdmann, and C. Embling. 2019. Population dynamics of oceanic manta rays (*Mobula birostris*) in the Raja Ampat Archipelago, West Papua, Indonesia, and the impacts of the El Nino–Southern Oscillation on their movement ecology. Diversity and Distributions. 25(9):1472-1487.

Beavers, S. C., and E. R. Cassano. 1996. Movements and dive behavior of a male sea turtle (*Lepidochelys olivacea*) in the eastern tropical Pacific. Journal of Herpetology. 30(1):97-104.

Bechmann, R. K., B. K. Larsen, I. C. Taban, L. I. Hellgren, P. Møller, and S. Sanni. 2010. Chronic exposure of adults and embryos of *Pandalus borealis* to oil causes PAH accumulation, initiation of biomarker responses and an increase in larval mortality. Marine Pollution Bulletin. 60(11):2087-2098.

Beerkircher, L. R., Cortés, E., and M. Shivji. 2008. Case study: Elasmobranch bycatch in the pelagic longline fishery off the southeastern United States, 1992–1997. In: Camhi M, Pikitch EK, Babcock EA, editors. Sharks of the Open Ocean: Biology, Fisheries Conservation. Blackwell Publishing Ltd. p. 242-246.

Bejarano-Alvarez, M., F. Galvan-Magana, and R. I. Ochoa-Baez. 2011. Reproductive biology of the scalloped hammerhead shark *Sphyrna lewini* (Chondrichthyes: Sphyrnidae) off southwest Mexico. International Journal of Ichthyology. 17(1):11-22.

Bell, C. D., J. M. Blumenthal, A. C. Broderick, and B. J. Godley. 2010. Investigating potential for depensation in marine turtles: how low can you go? Conservation Biology. 24(1):226-235.

Bellagio Steering Committee. 2008. Strategic planning for long-term financing of Pacific leatherback conservation and recovery: Proceedings of the Bellagio Sea Turtle Conservation Initiative, Terengganu, Malaysia; July 2007. WorldFish Center Conference Proceedings.

Bembenek-Bailey, S. A., J. N. Niemuth, P. D. McClellan-Green, M. H. Godfrey, C. A. Harms, H. Gracz, and M. K. Stoskopf. 2019. NMR Metabolomic Analysis of Skeletal Muscle, Heart, and Liver of Hatchling Loggerhead Sea Turtles (*Caretta caretta*) Experimentally Exposed to Crude Oil and/or Corexit. Metabolites. 9(2).

Benson, S. R., Dutton, P. H., Hitipeuw, C., Samber, B., Bakarbessy, J., and D. Parker. 2007a. Post-nesting migrations of leatherback turtles (*Dermochelys coriacea*) from Jamursba-Medi, Bird's Head Peninsula, Indonesia. Chelonian Conservation and Biology. 6(1):150-154.

Benson, S. R., P. H. Dutton, C. Hitipeuw, B. Samber, R. F. Tapilatu, V. Rei, L. Ambio, J. Pita, P. Ramohia, and J. Horoku. 2012. A Tri-National Aerial Survey of Leatherback Nesting Acivity in New Guinea and The Solomon Islands.

Benson SR, R. V., Hitipeuw C, Samber B, Tapilatu R, Pita J, Ramohia P, Pakacha P, Horoku J, Wurlanty B, *et al.* 2018. A tri-national aerial survey of leatherback nesting activity in New Guinea and the Solomon Islands. 38th Annual Symposium on Sea Turtle Biology and Conservation, Kobe, Japan.

Benson, S. R., Eguchi, T., Foley, D. G., Forney, K. A., Bailey, H., Hitipeuw, C., Samber, B. P., Tapilatu, R. F., Rei, V., Ramohia, P. *et al.* 2011. Large-scale movements and high-use areas of western Pacific leatherback turtles, *Dermochelys coriacea*. Ecosphere. 2(7):art84.

Benson, S. R., Forney, K. A., Harvey, J. T., Carretta, J. V., and P. H. Dutton. 2007b. Abundance, distribution, and habitat of leatherback turtles (*Dermochelys coriacea*) off California, 1990– 2003. Fishery Bulletin. 105(3):337-347.

Benson, S.R., K.A. Forney, J.E. Moore, E.L. LaCasella, J.T. Harvey, and J.V. Caretta. 2020. A long-term decline in the abundance of endangered leatherback turtles, *Dermochelys coriacea*, at a foraging ground in the California Current Ecosystem. Global Ecology and Conservation 24: e01371.

Benson, S. R., R. F. Tapilatu, N. Pilcher, P. S. Tomillo, and L. S. Martínez. 2015. Leatherback turtle populations in the Pacific Ocean. Biology and conservation of leatherback turtles. p. 110-122. John Hopkins University Press, Baltimore.

Bermudez, R., Winder, M., Stuhr, A., Almen, A. K., Engstrom-Ost, J., and U. Riebesell. 2016. Effect of ocean acidification on the structure and fatty acid composition of a natural plankton community in the Baltic Sea. Biogeosciences. 13(24):6625-6635.

Bernardo, J., and P. T. Plotkin. 2007. An evolutionary perspective on the arribada phenomenon and reproductive behavioral polymorphism of olive ridley sea turtles (*Lepidochelys olivacea*). In: Plotkin, P.T., editor. Biology and conservation of ridley sea turtles. p. 59-87.

Berzin, A. A., and A. A. Rovnin. 1966. The distribution and migrations of whales in the northeastern part of the Pacific, Chukchi and Bering Seas. Izvestiya Tikhookeanskogo Nauchno-Issledovatel'skogo Institut Rybnogo Khozyaistva I Okeanografii. 58:179-207.

Bessudo, S., G. A. Soler, A. P. Klimley, J. T. Ketchum, A. Hearn, and R. Arauz. 2011. Residency of the scalloped hammerhead shark (*Sphyrna lewini*) at Malpelo Island and evidence of migration to other islands in the Eastern Tropical Pacific. Environmental Biology of Fishes. 91(2):165-176.

Best, P.B. 1983. Sperm whale stock assessments and the relevance of historical whaling records. Reports of the International Whaling Commission (Special Issue 5): 41–55.

Beverly, S., and L. Chapman. 2007. Interactions between sea turtles and pelagic longline fisheries. Third Regular Session of the Scientific Committee of the Western and Central Pacific Fisheries Commission. p. 13-24.

Bigelow, K. and F. Carvalho. 2021. Statistical and Monte Carlo analysis of the Hawaii deep-set longline fishery with emphasis on take and mortality of Oceanic Whitetip Shark. NOAA Fisheries, Pacific Islands Fisheries Science Center. 12 p.

Bigelow, H. B., and W. C. Schroeder. 1953. Sawfishes, guitarfishes, skates and rays. Fishes of the Western North Atlantic. Memoirs of Sears Foundation for Marine Research. 1:514.

Bigelow, K., J. Rice, and F. Carvalho. 2022. Future Stock Projections of Oceanic Whitetip Sharks in the Western and Central Pacific Ocean (Update on Project 101). Scientific Committeee Eighteenth Regular Session. WCPFC-SC18-2022/EB-WP-02. 19 p.

Bignami, S., Sponaugle, S., and R. K. Cowen. 2013. Response to ocean acidification in larvae of a large tropical marine fish, *Rachycentron canadum*. Global Change Biology. 19(4):996-1006.

Binckley, C. A., J. R. Spotila, K. S. Wilson, and F. V. Paladino. 1998. Sex determination and sex ratios of Pacific leatherback turtles, Dermochelys coriacea. Copeia 2(291-300).

Blanco ,G. S. 2010. Movements and Behavior of the East Pacific Green Turtle (*Chelonia mydas*) from Costa Rica. Drexel University.

Bleckmann, H., and M. H. Hofmann. 1999. Special senses. In: Hamlett WC, editor. Sharks, skates, rays: The biology of elasmobranch fishes. The Johns Hopkins University Press, Baltimore, Maryland. p. 300-328.

Boggs, C. H. 1992. Depth, capture time, and hooked longevity of longline-caught pelagic fish: timing bites of fish with chips. Fishery Bulletin. 90:642-658.

Boggs, C. H., D. P. Gonzales, and R. M. Kokubun. 2015. Marine Mammals Reported under Catch Lost to Predators on Fisherman's Commercial Catch Reports to the State of Hawaii, 2003-2014. Pacific Islands Fisheries Science Center, PIFSC Data Report, DR-15-006. 14 p. https://repository.library.noaa.gov/view/noaa/5011

Boggs, C. H., and R. Y. Ito. 1993. Hawaii's pelagic fisheries. Marine Fisheries Review. 55(2):69-82.

Bonfil, R., S. Clarke, H. Nakano, M. D. Camhi, E. K. Pikitch, and E. A. Babcock. 2008. The biology and ecology of the oceanic whitetip shark, *Carcharhinus longimanus*. Sharks of the open ocean: Biology, Fisheries and Conservation.128-139.

Booth, H., U. Mardhiah, H. Siregar, J. Hunter, M. I. H. Putra, J. Marlow, A. Cahyana, A. Y. L. Demoor, S. Lewis, D. J. C. S. Adhiasto et al. 2021. An integrated approach to tackling wildlife crime: Impact and lessons learned from the world's largest targeted manta ray fishery. 3(2):e314.

Borenstein, M. 2009. Effects sizes for continuous data. Pages: 221-236. In: *The handbook of research synthesis and meta-analysis. Second Edition.* Edited by H. Cooper, L.V. Hedges and J.C. Valentine. Russell Sage Foundation; New York, New York.

Bowen, B. W., A. M. Clark, F. A. Abreu-Grobois, A. Chaves, H. A. Reichart, and R. J. Ferl. 1998. Global phylogeography of the ridley sea turtles (*Lepidochelys spp.*) as inferred from mitochondrial DNA sequences. Genetica. 101(3):179-189.

Bradford, A. L., E. M. Oleson, R. W. Baird, C. H. Boggs, K. A. Forney, and N. C. Young. 2015. Revised stock boundaries for false killer whales (*Pseudorca crassidens*) in Hawaiian waters. U.S Dep. Commerc. NOAA Tech Memo., NOAA-TM-NMFS-PIFSC-47. 29 p. https://repository.library.noaa.gov/view/noaa/5049

Bradford, A. L., and K. A. Forney. 2013. Injury determinations for cetaceans observed interacting with Hawaii and American Samoa longline fisheries during 2007-2011. PIFSC Working Paper WP-13-002. https://repository.library.noaa.gov/view/noaa/4604

Bradford, A. L., K. A. Forney, E. M. Oleson, and J. Barlow. 2017. Abundance estimates of cetaceans from a line-transect survey within the U.S. Hawaiian Islands Exclusive Economic Zone. Fishery Bulletin. 115(2):129-142.

Bradford, A. L., R. W. Baird, S. D. Mahaffy, A. M. Gorgone, D. J. McSweeney, T. Cullins, D. L. Webster, and A. N. Zerbini. 2018. Abundance estimates for management of endangered false killer whales in the main Hawaiian Islands. Endangered Species Research. 36:297-313.

Branstetter, B. K. 1987. Age, growth and reproductive biology of the silky shark, Carcharhinus falciformis, and the scalloped hammerhead, *Sphyrna lewini*, from the northwestern Gulf of Mexico. Environmental Biology of Fishes. 19(3):161-173.

Branstetter, S. 1990. Early life-history implications of selected carcharhinoid and lamnoid sharks of the northwest Atlantic. In: Pratt, H. J., S. Gruber, T. Taniuchi, editors. Elasmobranchs as living resources: advances in the biology, ecology, systematics and the status of the fisheries.

Briscoe, D. K., D. M. Parker, G. H. Balazs, M. Kurita, T. Saito, H. Okamoto, M. Rice, J. J. Polovina, and L. B. Crowder. 2016a. Active dispersal in loggerhead sea turtles (*Caretta caretta*) during the 'lost years'. Proceedings of the Royal Society B: Biological Sciences. 283(1832):20160690.

Briscoe, D. K., D. M. Parker, S. Bograd, E. Hazen, K. Scales, G. H. Balazs, M. Kurita, T. Saito, H. Okamoto, M. Rice et al. 2016b. Multi-year tracking reveals extensive pelagic phase of juvenile loggerhead sea turtles in the North Pacific. Movement Ecology. 4(1):23.

Broderick, D. 1998. Subsistence harvesting of marine turtles in the Solomon Islands. Patterns of resource use in Kia, Wagina, and Katupika communities, Isabel and Choiseul Provinces. Brisbane, Australia. p. 12.

Brodziak, J., W. A. Walsh, and R. Hilborn. 2013. Model selection and multimodel inference for standardizing catch rates of bycatch species: a case study of oceanic whitetip shark in the Hawaii-based longline fishery. Canadian Journal of Fisheries and Aquatic Sciences. 70(12):1723-1740.

Brownell, R. L., P. J. Clapham, T. Miyashita, and T. Kasuya. 2001. Conservation status of North Pacific right whales. Journal of Cetacean Research and Management.269-286.

Bureau of Marine Resources. 2005. Palau marine turtle conservation and management project, September 2004 to September 2005. Republic of Palau.

Burgess, K. B. 2017. Feeding ecology and habitat use of the giant manta ray Manta birostris at a key aggregation site off mainland Ecuador. The University of Queensland, Queensland. p. 174.

Busch, D. S., C. J. Harvey, and P. McElhany. 2013. Potential impacts of ocean acidification on the Puget Sound food web. ICES Journal of Marine Science. 70(4):823-833.

Camacho, M., L. D. Boada, J. Oros, P. Calabuig, M. Zumbado, and O. P. Luzardo. 2012. Comparative study of polycyclic aromatic hydrocarbons (PAHs) in plasma of Eastern Atlantic juvenile and adult nesting loggerhead sea turtles (*Caretta caretta*). Marine Pollution Bulletin. 64(9):1974-1980.

Camacho, M., O. P. Luzardo, L. D. Boada, L. F. L. Jurado, M. Medina, M. Zumbado, and J. Oros. 2013. Potential adverse health effects of persistent organic pollutants on sea turtles: evidences from a cross-sectional study on Cape Verde loggerhead sea turtles. Science of the Total Environment. 458:283-289.

Camargo, S. M., Coelho, R., Chapman, D., Howey-Jordan, L., Brooks, E. J., Fernando, D., Mendes, N. J., Hazin, F.H., Oliveira, C., Santos, M.N. *et al.* 2016. Structure and Genetic Variability of the Oceanic Whitetip Shark, *Carcharhinus longimanus,* Determined Using Mitochondrial DNA. PLoS One. 11(5):e0155623.

Campana, S. E., W. Joyce, and M. J. Manning. 2009. Bycatch and discard mortality in commercially caught blue sharks *Prionace glauca* assessed using archival satellite pop-up tags. Marine Ecology Progress Series. 387:241-253.

Carey, F. G., and B. H. Robinson. 1981. Daily patterns in the activities of swordfish, *Xiphias gladius,* observed by acoustic telemetry [California]. Fishery Bulletin. 79(2).

Carey, J. R., and D. A. Roach. 2020. Biodemography: An Introduction to Concepts and Methods. Princeton University Press.

Carr, A. 1987. New perspectives on the pelagic stage of sea turtle development. Conservation Biology. 1(2):103-121.

Carr, A. F., M. H. Carr, and A. B. Meylan. 1978. The ecology and migrations of sea turtles. 7, The West Caribbean green turtle colony. Bulletin of the AMNH; v. 162, article 1. 52 p.

Carretta, J. V. 2022. Estimates of marine mammal, sea turtle, and seabird bycatch from the California large-mesh drift gillnet fishery: 1990-2020. NOAA-TM-NMFS-SWFSC-666. https://repository.library.noaa.gov/view/noaa/45012

Carretta, J. V., V. T. Helker, M. M. Muto, J. Greenman, K. Wilkinson, D. Lawson, J. Viezbicke, and J. Jannot. 2018. Sources of Human-Related Injury and Mortality for U.S. Pacific West Coast Marine Mammal Stock Assessments, 2012-2016. US Department of Commerce. NOAA Technical Memorandum NMFS-SWFSC601. https://repository.library.noaa.gov/view/noaa/18081

Carretta, J. V., J. E. Moore, and K. A. Forney. 2017. Regression tree and ratio estimates of marine mammal, sea turtle, and seabird bycatch in the California drift gillnet fishery, 1990-2015. U.S. Department of Commerce, NOAA Technical Memorandum, NOAA-TM-NMFS-SWFSC-568. p. 89. https://repository.library.noaa.gov/view/noaa/13141

Carretta, J. V., K. A. Forney, E. M. Oleson, D. W. Weller, A. R. Lang, J. Baker, M. M. Muto, M. B. Hanson, A. J. Orr, H. Huber *et al.* 2019. U.S. Pacific Marine Mammal Stock Assessments: 2018. NOAA-TM-NMFS-SWFSC-617. https://repository.library.noaa.gov/view/noaa/20266

Carretta, J. V., J. E. Moore, and K. A. Forney. 2019. Estimates of marine mammal, sea turtle, and seabird bycatch from the California large-mesh drift gillnet fishery: 1990-2017. NOAA-TM-NMFS-SWFSC-619. https://repository.library.noaa.gov/view/noaa/20693

Carretta, J. V., B. Delean, V. Helker, M. M. Muto, J. Greenman, K. Wilkinson, D. Lawson, J. Viezbicke, and J. Jannot. 2020. Sources of Human-Related Injury and Mortality for U.S. Pacific West Coast Marine Mammal Stock Assessments, 2014-2018. NOAA-TM-NMFS-SWFSC-631. 147 p. https://repository.library.noaa.gov/view/noaa/25230

Carretta, J. V., E. M. Oleson, J. Baker, D. W. Weller, A. R. Lang, K. A. Forney, M. M. Muto, B. Hanson, A. J. Orr, H. Huber et al. 2022. False Killer Whale (*Pseudorca crassidens*): Hawaiian Islands Stock Complex – Main Hawaiian Islands Insular, Northwestern Hawaiian Islands, and Hawaii Pelagic Stocks. 12 p. In: U. S. Pacific Marine Mammal Stock Assessments: 2021. Updated 4/15/22. Available online at: https://media.fisheries.noaa.gov/2022-08/2021-FALSE%20KILLER%20WHALE-Hawaiian%20Islands%20Stock%20Complex.pdf

Caseale, P., M. Affronte, G. Insacco, D. Freggi, C. Vallini, P. P. d'Astore, R. Basso, G. Paolillo, G. Abbate, and R. Argano. 2010. Sea turtle strandings reveal high anthropogenic mortality in Italian waters. Aquatic Conservation-Marine and Freshwater Ecosystems. 20(6):611-620.

Caut, S., E. Guirlet, and M. Girondot. 2010. Effect of tidal overwash on the embryonic development of leatherback turtles in French Guiana. Marine Environment Restoration. 69(4):254-261.

Chaloupka, M., and G. Balazs. 2007. Using Bayesian state-space modelling to assess the recovery and harvest potential of the Hawaiian green sea turtle stock. Ecological modelling. 205(1-2):93-109.

Chaloupka, M., K. A. Bjorndal, G. H. Balazs, A. B. Bolten, L. M. Ehrhart, C. J. Limpus, H. Suganuma, S. Troeng, and M. Yamaguchi. 2008a. Encouraging outlook for recovery of a once severely exploited marine megaherbivore. Global Ecology and Biogeography. 17(2):297-304.

Chaloupka, M., and C. Limpus. 2005. Estimates of sex-and age-class-specific survival probabilities for a southern Great Barrier Reef green sea turtle population. Marine Biology. 146(6):1251-1261.

Chaloupka, M. Y. and N. J. Pilcher. 2019. *Chelonia mydas (Hawaiian subpopulation)*. The IUCN Red List of Threatened Species 2019: e.T16285718A142098300.

Chaloupka, M., T. M. Work, G. H. Balazs, S. K. K. Murakawa, and R. Morris. 2008b. Cause-specific temporal and spatial trends in green sea turtle strandings in the Hawaiian Archipelago (1982–2003). Marine Biology. 154(5):887-898.

Chan, F., J. A. Barth, C. A. Blanchette, R. H. Byrne, F. Chavez, O. Cheriton, R. A. Feely, G. Friederich, B. Gaylord, T. Gouhier, *et al.* 2017. Persistent spatial structuring of coastal ocean acidification in the California Current System. Science Reports. 7(1):2526.

Chan, S.K.-F., I.-J. Cheng, T. Zhou, H.-J. Wang, H.-X. Gu, and X.-J. Song. 2007. A comprehensive overview of the population and conservation status of sea turtles in China. Chelonian Conservation and Biology. 6(2):185-198.

Chen, C. T., T. Leu, and S. J. Joung. 1988. Notes on reproduction in the scalloped hammerhead, *Sphyrna lewini*, in Taiwan waters. Fisheries Bulletin. 86:389-393.

Cheng, I. J., P. H. Dutton, C. L. Chen, H. C. Chen, Y. H. Chen, and J. W. Shea. 2008. Comparison of the genetics and nesting ecology of two green turtle rookeries. Journal of Zoology. 276(4):375-384.

Chin, A., P. M. Kyne, T. I. Walker, and R. B. McAuley. 2010. An integrated risk assessment for climate change: analysing the vulnerability of sharks and rays on Australia's Great Barrier Reef. Global Change Biology. 16(7):1936-1953.

Chivers, S. J., R. W. Baird, K. M. Martien, B. L. Taylor, E. Archer, A. M. Gorgone, B. L. Hancock, N. M. Hedrick, D. Matilla, and D. J. McSweeney. 2010. Evidence of genetic differentiation for Hawaii insular false killer whales (*Pseudorca crassidens*). NOAA TM-NMFS-SWFSC-458. Cascadia Research Collective. Olympia, WA. 49 p. https://repository.library.noaa.gov/view/noaa/3729

CITES. 2010. Consideration of proposals for amendment of Appendices I and II. Convention on International Trade in Endangered Species of Wild Fauna and Flora. Fifteenth meeting of the Conference of the Parties Doha (Qatar), 13-25 March 2010. p. 28.

CITES. 2013. Consideration of proposals for amendment of Appendices I and II: Manta Rays. Bangkok, Thailand, March 3-14.

Clapham, P. J., S. B. Young, and R. L. Brownell Jr. 1999. Baleen Whales: Conservation Issues and The Status Of The Most Endangered Populations. Mammal Review. 29(1):37-62.

Clapham, P. J., C. P. Good, S. E. Quinn, R. R. Reeves, J. E. Scarff, and R. L. Brownell Jr. 2004. Distribution of North Pacific right whales (*Eubalaena japonica*) as shown by 19th and 20th century whaling catch and sighting records

Clapp, R. B., and W. Wirtz. 1975. The natural history of Lisianski Island, Northwestern Hawaiian Islands. Atoll Research Bulletin.

Clark, J.M. and D. J. Agnew. 2010. Estimating the impact of depredation by killer whales and sperm whales on longline fishing for toothfish (*Dissostichus eleginoides*) around South Georgia. CCAMLR Science 17:163-178.

Clark, T. B. 2010. Abundance, home range, and movement patterns of manta rays (*Manta alfredi, M. birostris*) in Hawai'i. [Honolulu]: University of Hawaii at Manoa.

Clarke, S. 2017. Joint analysis of sea turtle mitigation effectiveness. Western and Central Pacific Fisheries Commission.

Clarke, T. A. 1971. The Ecology of the Scalloped Hammerhead Shark, *Sphyrna lewini*, in Hawaii.

Clarke, S. 2013. Towards an Integrated Shark Conservation and Management Measure for the Western and Central Pacific Ocean. Western and Central Pacific Fisheries Commission Scientific Committee Ninth Regular Session. WCPFC-SC9-2013/ EB-WP-08. 36 pp.

Clarke, S., S. Harley, S. Hoyle, and J. Rice. 2011a. An indicator-based analysis of key shark species based on data held by SPC-OFP. WCPFC-SC7-2011/EB-WP-01. p. 88.

Clarke, S., K. Yokawa, H. Matsunaga, and H. Nakano. 2011b. Analysis of North Pacific Shark Data from Japanese Commercial Longline and Research/Training Vessel Records. Pohnpei, Federated States of Micronesia. p. 89.

Clarke, S.C., S. J. Harley, S. D. Hoyle, and J. S. Rice. 2012. Population trends in Pacific Oceanic sharks and the utility of regulations on shark finning. Conservation Biology. 27(1):197-209.

Clarke, S. C., J. E. Magnussen, D. L. Abercrombie, M. K. McAllister, and M. S. Shivji. 2006a. Identification of Shark Species Composition and Proportion in the Hong Kong Shark Fin Market Based on Molecular Genetics and Trade Records. Conservation Biology. 20(1):201-211.

Clarke, S. C., M. K. McAllister, E. J. Milner-Gulland, G. P. Kirkwood, C. G. Michielsens, D. J. Agnew, E. K. Pikitch, H. Nakano, and M. S. Shivji. 2006b. Global estimates of shark catches using trade records from commercial markets. Ecology Letters. 9(10):1115-1126.

CMS. 2014. Proposal for the inclusion of the reef manta ray (*Manta alfredi*) in CMS Appendix I and II. 18th Meeting of the Scientific Council, UNEP/CMS/ScC18/Doc.7.2.9. p. 17.

Coelho, R., F. H. V. Hazin, M. Rego, M. Tambourgi, P. Oliveira, P. Travassos, F. Carvalho, and G. Burgess. 2009. Notes on the reproduction of the oceanic whitetip shark, *Carcharhinus longimanus,* in the southwestern Equatorial Atlantic Ocean. Collective Volume of Scientific Papers ICCAT. 64(5):1734-1740.

Coelho, R., P. G. Lino, and M. N. Santos. 2011. At-haulback mortality of elasmobranchs caught on the Portuguese longline swordfish fishery in the Indian Ocean. IOTC–2011–WPEB07–31.

Cohen, J. 1988. Power analysis for the behavioral sciences. 2nd Edition. Lawrence Erlbaum Associates, Hilldale, NJ 579 pp.

Collen, B., L. McRae, S. Deinet, A. De Palma, T. Carranza, N. Cooper, J. Loh, and J. E. Baillie. 2011. Predicting how populations decline to extinction. Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences. 366(1577):2577-2586.

Collins, J. F., J. P. Brown, G. V. Alexeeff, and A. G. Salmon. 1998. Potency equivalency factors for some polycyclic aromatic hydrocarbons and polycyclic aromatic hydrocarbon derivatives. Regulatory Toxicology Pharmacology. 28(1):45-54.

Commonwealth of Australia. 2017. Recovery Plan for Marine Turtles in Australia. Australian Government, Department of the Environment and Energy. 154 p.

Compagno, L. J. V. 1984. FAO species catalogue Vol. 4, part 2 sharks of the world: An annotated and illustrated catalogue of shark species known to date. Food and Agriculture Organization of the United Nations.

Conant, T. A., P. H. Dutton, T. Eguchi, E. P. Epperly, C.C. Fahy, M. H. Godfrey, S. L. MacPherson, E. E. Possardt, B. A. Schroeder, J. A. Seminoff, *et al.* 2009. Loggerhead sea turtle (*Caretta caretta*) 2009 status review under the U.S. Endangered Species Act. Report of the Loggerhead Biological Review Team to the National Marine Fisheries Service, August 2009. 222 pages.

Connor, R. C., J. Mann, P. L. Tyack, and H. Whitehead. 1998. Social evolution in toothed whales. Trends in Ecology Evolution. 13(6):228-232.

Conservation International Pacific Islands Program. 2013. Biodiversity Conservation Lessons Learned Technical Series 11: Double programme de recherche sur les tortues marines de l'Archipel de la Société, Polynésie française [Dual Research Program on Sea Turtles of the Society Archipelago – French Polynesia]. p. 96.

Cooper, H. L., Potts, D. C., and A. Paytan. 2016. Metabolic responses of the North Pacific krill, *Euphausia pacifica,* to short- and long-term pCO2 exposure. Marine Biology. 163(10).

Cornelius, S. E. 1986. The sea turtles of Santa Rosa National Park. Fundacion Parques Nacionales, San José, Costa Rica.

Cornelius, S. E., and D. C. Robinson-Clark. 1986. Post-nesting movements of female olive ridley turtles tagged in Costa Rica. Vida Silvestre Neotropical. 1(1):12-23.

Cornelius, S. E., M. A. Ulloa, J. C. Castro, M. M. Del Valle, and D. C. Robinson. 1991. Management of olive ridley sea turtles *Lepidochelys olivacea* nesting at Playas Nancite and Ostional, Costa Rica. In: Robinson JG, Redford KH, editors. Neotropical Wildlife Use and Conservation. The University of Chicago Press, Chicago, Illinois. p. 111-135.

Cornelius, S.E., R. Arauz, J. Fretey, M.H. Godfrey, R. Márquez-M., and K. Shanker. 2007. Effect of land-based harvest of Lepidochelys. Pages 231-251 in Plotkin, P.T. (editor). Biology and Conservation of Ridley Sea Turtles. Johns Hopkins University Press, Baltimore, Maryland.

Cortes, E. 2002. Incorporating uncertainty into demographic modeling: application to shark populations and their conservation. Conservation Biology. 16(4):1048-1062.

Cortes, E., C. A. Brown, and L. Beerkircher. 2007. Relative abundance of pelagic sharks in the western North Atlantic Ocean, including the Gulf of Mexico and Caribbean Sea. Gulf and Caribbean Research. 19(2):37-52.

Couturier, L. I., F.R. Jaine, and T. Kashiwagi. 2015. First photographic records of the giant manta ray *Manta birostris* off eastern Australia. PeerJ. 3:e742.

Couturier, L. I. E., A. D. Marshall, F. R. Jaine, T. Kashiwagi, S. J. Pierce, K. A. Townsend, S. J. Weeks, M. B. Bennett, and A. J. Richardson. 2012. Biology, ecology and conservation of the Mobulidae. Journal of fish biology. 80(5):1075-1119.

Coyne, M.C. 2000. Population sex ratio of the Kemp's ridley sea turtle (*Lepidochelys kempii*): Problems in population modeling. Dissertation, Texas A&M University, May 2000. 136pp

Crespo, E. A., S. N. Pedraza, S. L. Dans, M. Koen Alonso, L. M. Reyes, N. A. García, M. Coscarella, and A. C. Schiavini. 1997. Direct and indirect effects of the highseas fisheries on the marine mammal populations in the northern and central Patagonian coast. Journal of Northwest Atlantic Fishery Science. 22:189-207.

Croll, D. A., H. Dewar, N. K. Dulvy, D. Fernando, M. P. Francis, F. Galvan-Magana, M. Hall, S. Heinrichs, A. Marshall, K. M. Newton, *et al.* 016. Vulnerabilities and fisheries impacts: the uncertain future of manta and devil rays. Aquatic Conservation: Marine and Freshwater Ecosystems. 26(3):562-575.

Crouse, D.T. 1999. The consequences of delayed maturity in the human dominated world. In: J.A. Musick (ed.) Life in the slow lane: Ecology and conservation of long-live marine animals. American Fisheries Society Symposium 23, pp 195-202.

Cruce, J. A. 2009. Monitoring of Nesting Green Turtles (*Chelonia mydas*) on Loosiep Island, Ulithi Atoll, Yap, Federated States of Micronesia 2008 Technical Report Prepared for JIMAR. p. 20.

Cummings, W. C., and J. F. Fish. 1971. A synopsis of marine animal underwater sounds in eight geographic areas. Naval Undersea Research and Development Center. Pasadena, CA. 100 p

da Silva, A.C.C.D., J.C. de Castilhos, G.G. Lopez, and P.C.R. Barata. 2007. Nesting biology and conservation of the olive ridley sea turtle (*Lepidochelys olivacea*) in Brazil, 991/1992 to 2002/2003. Journal of the Marine Biological Association of the United Kingdom. 87:1047¬-1056.

Dagorn, L., K. N. Holland, V. Restrepo, and G. Moreno. 2013. Is it good or bad to fish with FADs? What are the real impacts of the use of drifting FADs on pelagic marine ecosystems? Fish and Fisheries. 14(3):391-415.

Dapp, D. R., T. I. Walker, C. Huveneers, and R. D. Reina. 2016. Respiratory mode and gear type are important determinants of elasmobranch immediate and post-release mortality. Fish and Fisheries. 17(2):507-524.

Danion, M., S. Le Floch, F. Lamour, J. Guyomarch, C. J. E. Quentel, and E. Safety. 2011. Bioconcentration and immunotoxicity of an experimental oil spill in European sea bass (*Dicentrarchus labrax L.*). 74(8):2167-2174.

Davis, R. W., N. Jaquet, D. Gendron, U. Markaida, G. Bazzino, and W. Gilly. 2007. Diving behavior of sperm whales in relation to behavior of a major prey species, the jumbo squid, in the Gulf of California, Mexico. Marine Ecology Progress Series. 333:291-302.

De Andres, E., B. Gomara, D. Gonzalez-Paredes, J. Ruiz-Martin, and A. Marco. 2016. Persistent organic pollutant levels in eggs of leatherback turtles (*Dermochelys coriacea*) point to a decrease in hatching success. Chemosphere. 146:354-361.

Deakos, M. H., J. D. Baker, and L. Bejder. 2011. Characteristics of a manta ray *Manta alfredi* -population off Maui, Hawaii, and implications for management. Marine Ecology Progress Series. 429:245-260.

Delgado-Trejo, C., and J. Alvarado-Diaz. 2012. Current conservation status of the black sea turtle in Michoacan, Mexico. Sea turtles of the Eastern Pacific: advances in research and conservation. University of Arizona Press, Tucson. p. 263-278.

Delgado, S., and W. J. Nichols. 2005. Saving sea turtles from the ground up: awakening sea turtle conservation in northwestern Mexico. Maritime Studies. 4:89-104.

Denkinger, J., M. Parra, J. P. Munoz, C. Carrasco, J. C. Murillo, E. Espinosa, F. Rubianes, and V. Koch. 2013. Are boat strikes a threat to sea turtles in the Galapagos Marine Reserve? Ocean and Coastal Management. 80:29-35.

Dethmers, K. E., D. Broderick, C. Moritz, N. N. Fitzsimmons, C. J. Limpus, S. Lavery, S. Whiting, M. Guinea, R. I. Prince, and R. Kennett. 2006. The genetic structure of Australasian green turtles (*Chelonia mydas*): exploring the geographical scale of genetic exchange. Molecular Ecology. 15(13):3931-3946.

Dewar, H., P. Mous, M. Domeier, A. Muljadi, J. Pet, and J. Whitty. 2008. Movements and site fidelity of the giant manta ray, *Manta birostris*, in the Komodo Marine Park, Indonesia. Marine Biology. 155(2):121-133.

Diemer, K. M., B. Q. Mann, and N. E. Hussey. 2011. Distribution and movement of scalloped hammerhead *Sphyrna lewini* and smooth hammerhead *Sphyrna zygaena* sharks along the east coast of southern Africa. African Journal of Marine Science. 33(2):229-238.

Domingo, A., M. Pons, S. Jimenez, P. Miller, C. Barcelo, and Y. Swimmer. 2012. Circle Hook Performance in the Uruguayan Pelagic Longline Fishery. Bulletin of Marine Science. 88(3):499-511.

Doney, S. C., M. Ruckelshaus, J. E. Duffy, J. P. Barry, F. Chan, C. A. English, H. M. Galindo, J. M. Grebmeier, A. B. Hollowed, N. Knowlton *et al.* 2012. Climate change impacts on marine ecosystems. Annual Review of Marine Science. 4:11-37.

Donogue, M., R. R. Reeves, and G. S. Stone. 2002. Report on the workshop on interactions between cetaceans and longline fisheries. New England Aquarium Aquatic Forum Series Report. p. 1-44.

Donoso, M., and P. H. Dutton. 2010. Sea turtle bycatch in the Chilean pelagic longline fishery in the southeastern Pacific: Opportunities for conservation. Biological Conservation. 143(11):2672-2684.

Donovan, G. P. 1991. A review of IWC stock boundaries. Reports International Whaling Commission. 13:39-68.

Dornfeld, T. C., and F. V. Paladino. 2012. Benefits and impacts of solitary nesting for olive ridleys, *Lepidochelys olivacea*. In: Jones TT, Wallace BP, editors. Proceedings of the Thirty-First Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-631. p. 199-200. https://repository.library.noaa.gov/view/noaa/4405

Driggers, W. B., J. K. Carlson, E. Cortes, and G. W. Ingram Jr. 2011. Effects of wire leader use and species-specific distributions on shark catch rates off the southeastern United States. IOTC-2011-SC14 –INF08.

Dudley, S. F. J., and C. A. Simpfendorfer. 2006. Population status of 14 shark species caught in the protective gillnets off KwaZulu–Natal beaches, South Africa, 1978–2003. Marine Freshwater Research. 57(2):225-240.

Duffy, C. A. J., and D. Abbott. 2003. Sightings of mobulid rays from northern New Zealand, with confirmation of the occurrence of *Manta birostris* in New Zealand waters. New Zealand Journal of Marine and Freshwater Research. 37(4):715-721.

Dulvy, N. K., S. A. Pardo, C. A. Simpfendorfer, and J. K. Carlson. 2014. Diagnosing the dangerous demography of manta rays using life history theory. PeerJ. 2:e400.

Duncan, K. M., and K. N. Holland. 2006. Habitat use, growth rates and dispersal patterns of juvenile scalloped hammerhead sharks *Sphyrna lewini* in a nursery habitat. Marine Ecology Progress Series. 312:211-221.

Duncan, K. M., A. P. Martin, B. W. Bowen, and H. G. De Couet. 2006. Global phylogeography of the scalloped hammerhead shark (*Sphyrna lewini*). Molecular Ecology. (15):2239-2251.

Dunn, M.R.; Finucci, B.; Pinkerton, M.H.; Sutton, P. 2022. Review of commercial fishing interactions with marine reptiles. NIWA Client Report 2022147WN. 78 p

Dutton, P. H., B. W. Bowen, D. W. Owens, A. Barragan, and S. K. Davis. 1999. Global phylogeography of the leatherback turtle (*Dermochelys coriacea*). Journal of Zoology. 248(3):397-409.

Dutton, P. H., C. Hitipeuw, M. Zein, S. R. Benson, G. Petro, J. Pita, V. Rei, L. Ambio, and J. Bakarbessy. 2007. Status and genetic structure of nesting populations of leatherback turtles (*Dermochelys coriacea*) in the western Pacific. Chelonian Conservation and Biology. 6(1):47-53.

Dutton, P. H., B. W. Bowen, D. W. Owens, A. Barragan, and S. K. Davis. 1999. Global phylogeography of the leatherback turtle (*Dermochelys coriacea*). Journal of Zoology 248(3):397-409.

Eckert, K. L. 1993. The biology and population status of marine turtles in the North Pacific Ocean. NOAA-TM-NMFS-SWFSC-186. p. 166. https://repository.library.noaa.gov/view/noaa/6133

Eckert, K. L., B. P. Wallace, J. G. Frazier, S. A. Eckert, and P. C. H. Pritchard. 2012. Synopsis of the biological data on the leatherback sea turtle (*Dermochelys coriacea*). CreateSpace.

Eckert, S. A., K. L. Eckert, P. J. Ponganis, and G. L. Kooyman. 1989. Diving and foraging behavior of leatherback sea turtles (*Dermochelys coriacea*). Canadian Journal of Zoology. 67(11):2834-2840.

Eckert, S. A., D. W. Nellis, K. L. Eckert, and G. L. Kooyman. 1986. Diving patterns of two leatherback sea turtles (*Dermochelys coriacea*) during internesting intervals at Sandy Point, St. Croix, USVI. Herpetologica 42(3):381-388.

Eckert, S. A., and L. Sarti. 1997. Distant fisheries implicated in the loss of the world's largest leatherback nesting population. Marine Turtle Newsletter. 78:2.

Eguchi, T., J. A. Seminoff, R. A. LeRoux, D. Prosperi, D. L. Dutton, and P. H. Dutton. 2012. Morphology and growth rates of the green sea turtle (*Chelonia mydas*) in a northern-most temperate foraging ground. Herpetologica. 68(1):76-87.

Ehsanpour, M., M. Afkhami, R. Khoshnood, K. J. Reich. 2014. Determination and maternal transfer of heavy metals (Cd, Cu, Zn, Pb and Hg) in the Hawksbill sea turtle (*Eretmochelys imbricata*) from a nesting colony of Qeshm Island, Iran. Bulletin of Environmental Contamination and Toxicology. 92(6):667-673.

Ellis, J. R., S. R. McCully Phillips, and F. Poisson. 2017. A review of capture and post-release mortality of elasmobranchs. Journal of fish biology. 90(3):653-722.

Erdmann, M. 2014. New MMAF-CI-SEAA manta tagging program launched by Mark Erdmann.

ESRI. 2018. How Kernal Density works. 2019 (February 4) https://pro.arcgis.com/en/pro-app/tool-reference/spatial-analyst/how-kernel-density-works.htm.

FAO. 2012. Report of the fourth FAO expert advisory panel for the assessment of proposals to amend Appendices I and II of CITES concerning commercially-exploited aquatic species. In: FAO Fisheries and Aquaculture Report No. 1032. Rome. p. 169.

Fernando, D. and G. Stevens. 2011. A study of Sri Lanka's manta and mobula ray fishery. The Manta Trust.

Ferreira, I. M. 2008. Growth and reproduction in false killer whales (*Pseudorca crassidens* Owens, 1846) [Masters Thesis]. University of Pretoria. p. 152.

Ferretti, F., B. Worm, G.L. Britten, M.R. Heithaus, and H.K. Lotze. 2010. Patterns and ecosystem consequences of shark declines in the ocean. Ecology Letters 13: 1055-1071.

FitzSimmons, N. N., C. Moritz, C. J. Limpus, J. D. Miller, C. J. Parmenter, and R. I. Prince. 1996. Comparative genetic structure of green, loggerhead, and flatback populations in Australia based on variable mtDNA and nDNA regions. In: Proceedings of the International Symposium on Sea Turtle Conservation Genetics. Miami, Florida: NOAA Technical memorandum NMFS-SEFSC-396. p. 25-32. https://repository.library.noaa.gov/view/noaa/8453

Foltz, K. M., R. W. Baird, G. M. Ylitalo, and B. A. Jensen. 2014. Cytochrome P4501A1 expression in blubber biopsies of endangered false killer whales (*Pseudorca crassidens*) and nine other odontocete species from Hawai'i. Ecotoxicology. 23(9):1607-1618.

Fonseca, L. G., Villachica, W. N., Matarrita, E. R., Arguello, Y., Orrego, C. M., Quiros, W., Seminoff, J. A. and R. A. Valverde. 2013. Preliminary data on the olive ridley tagging program at Nancite Beach, Costa Rica. In: Tucker T, Belskis L, Panagopoulou A, Rees AF, Frick M, Williams F, LeRoux R, Stewart K, editors. Proceedings of the Thirty Third Annual Symposium of Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC 645. https://repository.library.noaa.gov/view/noaa/4403

Forney, K. A., J. Barlow, M. M. Muto, M. Lowry, J. Baker, G. Cameron, J. Mobley, C. Stinchcomb, and J. V. Carretta. 2000. US Pacific marine mammal stock assessments, 2000. NOAA-TM-NMFS-SWFSC-300. 281 p. https://repository.library.noaa.gov/view/noaa/3166

Forney, K. A., and D. R. Kobayashi. 2007. Updated Estimates of Mortality and injury of cetaceans in the Hawaii-based longline fishery, 1994-2005. U.S. Department of Commerce, NOAA-TM-NMFS-SWFSC-412. 35 p. https://repository.library.noaa.gov/view/noaa/3621

Forney, K. A., R. W. Baird, and E. M. Oleson. 2010. Rationale for the 2010 revision of stock boundaries for the Hawai'i insular and pelagic stocks of false killer whales, Pseudorca crassidens. NOAA Technical Memorandum NMFS-SWFSC-471. 7 p. https://repository.library.noaa.gov/view/noaa/3796

Fossi, M. C., Baini, M., Panti, C., Galli, M., Jimenez, B., Munoz-Arnanz, J., Marsili, L., Finoia, M. G., and D. Ramirez-Macias. 2017. Are whale sharks exposed to persistent organic pollutants and plastic pollution in the Gulf of California (Mexico)? First ecotoxicological investigation using skin biopsies. Comparative Biochemistry and Physiology C-Toxicology & Pharmacology. 199:48-58.

Fossi, M. C., Coppola, D., Baini, M., Giannetti, M., Guerranti, C., Marsili, L., Panti, C., de Sabata, E., and S. Clo. 2014. Large filter feeding marine organisms as indicators of microplastic in the pelagic environment: the case studies of the Mediterranean basking shark (*Cetorhinus maximus*) and fin whale (*Balaenoptera physalus*). Marine Environmental Research. 100:17-24.

Fossi, M. C., Panti, C., Guerranti, C., Coppola, D., Giannetti, M., Marsili, L., and R. Minutoli. 2012. Are baleen whales exposed to the threat of microplastics? A case study of the Mediterranean fin whale (*Balaenoptera ptzysalus*). Marine Pollution Bulletin. 64(11):2374-2379.

Francke, D. L., S. A. Hargrove, E. W. Vetter, C. D. Winn, G. H. Balazs, and K. D. Hyrenbach. 2013. Behavior of juvenile green turtles in a coastal neritic habitat: Validating time–depth–temperature records using visual observations. Journal of experimental marine biology ecology. 444:55-65.

Francke, D. L., and TRP Staff. 2015. Marine Turtle Strandings in the Hawaiian Islands January – December 2014. PIFSC Internal Report, IR-15-008. 21 p.

Frankham, R., C. J. A. Bradshaw, and B. W. Brook. 2014. Genetics in conservation management: Revised recommendations for the 50/500 rules, Red List criteria and population viability analysis. Biological Conservation 170:56-63.

Frazer, N. B., and R. C. Ladner. 1986. A growth curve for green sea turtles, *Chelonia mydas*, in the US Virgin Islands, 1913-14. Copeia. 1986(3):798-802.

Freedman, R., and S. S. Roy. 2012. Spatial patterning of *Manta birostris* in United States east coast offshore habitat. Applied Geography. 32(2):652-659.

Fuentes, M. M. P. B., M. Hamann, and C. J. Limpus. 2010a. Past, current and future thermal profiles of green turtle nesting grounds: Implications from climate change. Journal of Experimental Marine Biology and Ecology. 383(1):56-64.

Fuentes, M. M. P. B., C. J. Limpus, and M. Hamann. 2011. Vulnerability of sea turtle nesting grounds to climate change. Global Change Biology. 17(1):140-153.

Fuentes, M. M. P. B., C. J. Limpus, M. Hamann, and J. Dawson. 2010b. Potential impacts of projected sea-level rise on sea turtle rookeries. Aquatic Conservation: Marine and Freshwater Ecosystems. 20(2):132-139.

Fuentes, M. M. P. B., J. A. Maynard, M. Guinea, I. P. Bell, P. J. Werdell, and M. Hamann. 2009. Proxy indicators of sand temperature help project impacts of global warming on sea turtles in northern Australia. Endangered Species Research. 9:33-40.

Futerman, A. M. 2018. At the Intersection of Science & Policy: International Shark Conservation & Management. Duke Environmental Law & Policy Forum. 28:259-306.

Gambell, R. 1985. Sei whale, *Balaenoptera borealis* Lesson, 1828. Handbook of Marine Mammals, Vol. 3. The Sirenians and Baleen Whales. Academic Press Inc.

Gaona, O., and A. Barragán. 2016. Las tortugas marinas en México: Logros y perspectivas para su conservación. Ciudad de México, México.

Gaos, A. R., I. L. Yanez, and R. M. Arauz. 2006. Sea turtle conservation and research on the Pacific coast of Costa Rica. Technical Report. Programa Restauración de Tortugas Marinas.

Gaspar, P., S. R. Benson, P. H. Dutton, A. Reveillere, G. Jacob, C. Meetoo, A. Dehecq, and S. Fossette. 2012. Oceanic dispersal of juvenile leatherback turtles: going beyond passive drift modeling. Marine Ecology Progress Series. 457:265-284.

Gaspar, P., and M. Lalire. 2017. A model for simulating the active dispersal of juvenile sea turtles with a case study on western Pacific leatherback turtles. PLoS One. 12(7):e0181595.

Gaston, K. J., T. M. Blackburn, J. D. Greenwood, G. D. Gregory, R. M. Quinn, and J. H. Lawton. 2000. Abundance-occupancy relationships. Journal of Applied Ecology. 37:39-59.

Gennip, S. J. V., E. E. Popova, A. Yool, G. T. Pecl, A. J. Hobday, and C. J. B. Sorte. 2017. Going with the flow: the role of ocean circulation in global marine ecosystems under a changing climate. Global Change Biology. 23(7):2602-2617.

Germanov, E. S. 2015a. From manta rays to mass spectrometry. Accessible at: https://elitzagermanov.wordpress.com/2015/08/22/from-manta-rays-to-mass-spectrometry/

Germanov, E. S. 2015b. Microplastics & Megafauna. Accessible at: https://elitzagermanov.wordpress.com

Germanov, E. S., A. D. Marshall, L. Bejder, M. C. Fossi, and N. R. Loneragan. 2018. Microplastics: No small problem for filter-feeding megafauna. Trends in Ecology & Evolution. 33(4):227-232.

Gero, S., M. Milligan, C. Rinaldi, P. Francis, J. Gordon, C. Carlson, A. Steffen, P. Tyack, P. Evans, and H. Whitehead. 2014. Behavior and social structure of the sperm whales of Dominica, West Indies. Marine Mammal Science. 30(3):905-922.

Gero, S., D. Engelhaupt, L. Rendell, and H. Whitehead. 2009. Who cares? Between-group variation in alloparental caregiving in sperm whales. Behavioral Ecology. 20(4):838-843.

Gilman, E., D. Kobayashi, T. Swenarton, N. Brothers, P. Dalzell, and I. Kinan-Kelly. 2007a. Reducing sea turtle interactions in the Hawaii-based longline swordfish fishery. Biological Conservation. 139(1-2):19-28.

Gilman, E., T. Moth-Poulsen, and G. Bianchi. 2007b. Review of measures taken by intergovernmental organizations to address sea turtle and seabird interactions in marine capture fisheries. FAO Fisheries Circular. No. 1025. Rome, FAO. 42p.

Gilman, E., P. Suuronen, M. Hall, and S. Kennelly. 2013. Causes and methods to estimate cryptic sources of fishing mortality. Journal of Fish Biology. 83(4):766-803.

Gjertsen, H., and F. Pakiding. 2011. Socioeconomic research and capacity-building to strengthen conservation of Western Pacific leatherback turtles in Bird’s Head, Papua Barat, Indonesia. Honolulu: Western Pacific Regional Fishery Management Council.

Goetz, S., M. Laporta, J. M. Portela, M. B. Santos, and G. J. Pierce. 2010. Experimental fishing with an “umbrella-and-stones” system to reduce interactions of sperm whales (*Physeter macrocephalus*) and seabirds with bottom-set longlines for Patagonian toothfish (*Dissostichus eleginoides*) in the Southwest Atlantic. ICES Journal of Marine Science doi:10.1093/icesjms/fsq161.

Graham, R. T., M. J. Witt, D. W. Castellanos, F. Remolina, S. Maxwell, B. J. Godley, and L. A. Hawkes. 2012. Satellite tracking of manta rays highlights challenges to their conservation. PLoS One. 7(5):e36834.

Green, D. and F. Ortiz-Crespo. 1982. Status of sea turtle populations in the central eastern Pacific. Pages 221-233 in Bjorndal, K.A. (editor). Biology and Conservation of Sea Turtles. Smithsonian Institution Press, Washington, D.C.

Griffiths, S. P., N. J. A. C. M. Lezama-Ochoa, and F. Ecosystems. 2021. A 40-year chronology of the vulnerability of spinetail devil ray (*Mobula mobular*) to eastern Pacific tuna fisheries and options for future conservation and management. 31(10):2910-2925.

Gulko, D., and K. L. Eckert. 2004. Sea turtles: an ecological guide. Honolulu, HI: Mutual Publishing. p. 121.

Hahn, A. T., M. Jensen, B. Bowen, A. Abreu-Grobois, J. C. de Castilhos, B. de Thoisy, M. da Costa Gadegaste, C. Limpus, S. Whiting, N. FitzSimmons *et al.* 2012. Preliminary results on global genetic structure and phylogeography of olive ridleys 63 (*Lepidochelys olivacea*). In: Jones TT, Wallace BP, editors. Proceedings of the Thirty-First Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-631.

Haigh, R., D. Ianson, C. A. Holt, H. E. Neate, and A. M. Edwards. 2015. Effects of ocean acidification on temperate coastal marine ecosystems and fisheries in the northeast Pacific. PLoS One. 10(2):e0117533.

Hall, M. A., and M. Roman. 2013. Bycatch and non-tuna catch in the tropical tuna purse seine fisheries of the world. FAO Fisheries and Aquaculture Technical Paper. (568):244.

Hamabata, T., S. Nishida, N. I. Kamezaki, and H. Koike. 2009. Genetic structure of populations of the green turtle (*Chelonia mydas*) in Japan using mtDNA control region sequences. 35-50 p.

Hamer, D.J., S.J. Childerhouse, and N.J. Gales. 2012. Odontocete bycatch and depredation in longline fisheries: a review of available literature and of potential solutions. Marine Mammal Science 28:E345-E374.

Hanni, K. D., D. J. Long, R. E. Jones, P. Pyle, and L. E. Morgan. 1997. Sightings and strandings of Guadalupe fur seals in central and northern California, 1988–1995. Journal of Mammalogy. 78(2):684-690.

Hanselman, D. H., B. J. Pyper, and M. J. Peterson. 2018. Sperm whale depredation on longline surveys and implications for the assessment of Alaska sablefish. Fisheries Research. 200:75-83.

Harfush, M., Y. O. Mendez, E. M. Lopez-Reyes, E. A. Padilla, and F. A. Abreu-Grobois. 2008. Evaluation of the olive ridley sea turtle (*Lepidochelys olivacea*) size and fecundity and hatch success percentage from their eggs transferred to a protection corral on Escobilla beach, Oaxaca. In: Kalb H, Rohde AS, Gayheart K, Shanker K, editors. Proceedings of the Twenty-Fifth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-582. https://repository.library.noaa.gov/view/noaa/4414

Harley, S., B. Caneco, C. Donovan, L. Tremblay-Boyer, and S. Brouwer. 2015. Monte Carlo simulation modelling of possible measures to reduce impacts of longlining on oceanic whitetip and silky sharks. Working Paper EBIP-02. Eleventh Meeting of the Scientific Committee, WCPFC, 5–13 August 2015, Pohnpei, Federated States of Micronesia. 30 p.

Harms, C. A., P. McClellan-Green, M. Godfrey, E. Christiansen, H. Broadhurst, and C. Godard-Codding. 2014. Clinical pathology effects of crude oil and dispersant on hatchling loggerhead sea turtles (*Caretta caretta*). Proceedings of the 45th Annual International Association for Aquatic Animal Medicine, Gold Coast, Australia.17-22.

Harms, C. A., P. McClellan-Green, M. H. Godfrey, E. F. Christiansen, H. J. Broadhurst, and C. A. J. Godard-Codding. 2019. Crude Oil and Dispersant Cause Acute Clinicopathological Abnormalities in Hatchling Loggerhead Sea Turtles (*Caretta caretta*). Frontiers in Veterinary Science. 6:344.

Harrison, A.-L., D. P. Costa, A. J. Winship, S. R. Benson, S. J. Bograd, M. Antolos, A. B. Carlisle, H. Dewar, P. H. Dutton, and S. J. Jorgensen. 2018. The political biogeography of migratory marine predators. Nature Ecology & Evolution. 2(10):1571-1578.

Hasarangi, D. G. N., R. Maldeniya, and S. S. K. Haputhantri. 2012. A review on shark fishery resources in Sri Lanka. IOTC–2012–WPEB08–15 Rev_1. p. 15.

Hatase, H., Y. Matsuzawa, K. Sato, T. Bando, and K. Goto. 2004. Remigration and growth of loggerhead turtles (*Caretta caretta*) nesting on Senri Beach in Minabe, Japan: life-history polymorphism in a sea turtle population. Marine Biology. 144(4):807-811.

Hatase, H., K. Omuta, and K. Tsukamoto. 2013. A mechanism that maintains alternative life histories in a loggerhead sea turtle population. Ecology. 94(11):2583-2594.

Hatase, H., K. Sato, M. Yamaguchi, K. Takahashi, and K. Tsukamoto K. 2006. Individual variation in feeding habitat use by adult female green sea turtles (*Chelonia mydas*): are they obligately neritic herbivores? Oecologia. 149(1):52-64.

Hawaii Department of Transportation (HIDOT). 2013. Statewide Fuel Facilities Development Plan: Proposed Fuel Facilities in Hawaii’s Commercial Harbors. Available from:

https://hidot.hawaii.gov/harbors/files/2013/01/Statewide-Fuel-Facilities-Development-Plan-Part-1.pdf

Hawaii Longline Association. 2020. RE: Hawaii-Based Commercial Longline Deep-Set Fishery Section 7 Consultation. 7 p.

Hawkes, L. A., A. C. Broderick, M. H. Godfrey, B. Godley, and M. J. Witt. 2014. The impacts of climate change on marine turtle reproductive success. Coastal Conservation. 19:287.

Hawkes, L. A., A. C. Broderick, M. H. Godfrey, and B. J. Godley. 2009. Climate change and marine turtles. Endangered Species Research. 7:137-154.

Hays, G. C., A. Taxonera, B. Renom, K. Fairweather, A. Lopes, J. Cozens, J.-O. Laloe. 2022 Changes in mean body size in an expanding population of a threatened species. Proc. R. Soc. B 289: 20220696. https://royalsocietypublishing.org/doi/10.1098/rspb.2022.0696

Hays, G. C., J. D. R. Houghton, C. Isaacs, R. S. King, C. Lloyd, and P. Lovell. 2004. First records of oceanic dive profiles for leatherback turtles, *Dermochelys coriacea,* indicate behavioural plasticity associated with long-distance migration. Animal Behaviour. 67(4):733-743.

Hazen, E. L., S. J. Jorgensen, R. R. Rykaczewski, S. J. Bograd, D. G. Foley, I. D. Jonsen, S. A. Shaffer, J. P. Dunne, D. P. Costa, L. B. Crowder *et al.* 2012. Predicted habitat shifts of Pacific top predators in a changing climate. Nature Climate Change. 3(3):234-238.

Hazin, F. H., H. G. Hazin, and P. Travassos. 2007. CPUE and catch trends of shark species caught by Brazilian longliners in the Southwestern Atlantic Ocean. Collective Volume of Scientific Papers ICCAT. 60(2):636-647.

Hearn, A., J. Ketchum, A. P. Klimley, E. Espinoza, and C. Penaherrera. 2010. Hotspots within hotspots? Hammerhead shark movements around Wolf Island, Galapagos Marine Reserve. Marine Biology. 157(9):1899-1915.

Hearn, A. R., D. Acuna, J. T. Ketchum, C. Penaherrera, J. Green, A. Marshall, M. Guerrero, and G. Shillinger. 2014. Elasmobranchs of the Galapagos marine reserve. The Galapagos Marine Reserve. Springer. p. 23-59.

Heinrichs, S., M. O'Malley, H. Medd, and P. Hilton. 2011. The global threat to Manta and Mobula Rays. Manta Ray of Hope, WildAid, San Francisco, California.

Heppell, S. S., D. Crouse, L. Crowder, S. Epperly, W. Gabriel, T. Henwood, R. Marquez, and N. Thompson. 2005. A population model to estimate recovery time, population size, and management impacts on Kemp's ridleys. Chelonian Conservation and Biology 4:767–773.

Herman, L., C. Baker, P. Forestell, and R. Antinoja. 1980. Right whale, *Balaena glacialis*, sightings near Hawaii: a clue to the wintering grounds. Marine Ecology Progress Series. 2:271-275.

Hernandez-Echeagaray, O. E., R. Hernandez-Cornejo, M. Harfush-Melendez, and A. J. Garcia-Gasca. 2012. Evaluation of sex ratios of the olive ridley sea turtle (*Lepidochelys olivacea*) on the arribada nesting beach, La Escobilla, Mexico. 133:12-16.

Heupel, M. and R. McAuley. 2007. Sharks and Rays (Chondrichthyans) in the North-west Marine Region. Department of the Environment and Water Resources, National Oceans Office Branch, Canberra.

Hildebrand, J. A. 2004. Impacts of anthropogenic sound on cetaceans. Unpublished paper submitted to the International Whaling Commission Scientific Committee SC/56 E. p. 13.

Hildebrand, J. A. 2009. Anthropogenic and natural sources of ambient noise in the ocean. Marine Ecology Progress Series. 395:5-20.

Hill, P. S., and D. P. DeMaster. 1999. Pacific Marine Mammal Stock Assessments, 1999.

Hinojosa-Alvarez, S., R. P. Walter, P. Diaz-Jaimes, F. Galván-Magana, and E. M. Paig-Tran. 2016. A potential third manta ray species near the Yucatán Peninsula? Evidence for a recently diverged and novel genetic *Manta* group from the Gulf of Mexico. PeerJ 4:e2586

Hirth, H. F., and D. L. Rohovit. 1992. Marketing patterns of green and hawksbill turtles in Port Moresby, Papua New Guinea. Oryx. 26(1):39-42.

Hitipeuw, C., and J. Maturbongs. 2002. Marine turtle conservation program Jamurba-Medi nesting beach, north coast of the Bird’s Head Peninsula, Papua. Proceedings of the Western Pacific Sea Turtle Cooperative Research and Management Workshop. February 5–8, 2002; Honolulu, Hawaii, USA. Western Pacific Regional Fishery Management Council. p. 161-175.

Hochscheid, S. 2014. Why we mind sea turtles' underwater business: A review on the study of diving behavior. Journal of Experimental Marine Biology and Ecology. 450:118-136.

Hoenig, J. M., and S. H. Gruber. 1990. Life-History Patterns in the Elasmobranchs: Implications for Fisheries Management. In: Pratt J, H. L., Gruber SH, Taniuch T, editors. Elasmobranchs as Living Resources: Advances in the Biology, Ecology, Systematics, and the Status of the Fisheries. NOAA Technical Report NMFS 90. p. 528.

Holland, K. N., B. M. Wetherbee, J. D. Peterson, and C. G. Lowe. 1993. Movements and distribution of hammerhead shark pups on their natal grounds. Copeia.495-502.

Homma, K., T. Maruyama, T. Itoh, H. Ishihara, and S. Uchida. 1999. Biology of the manta ray, *Manta birostris Walbaum*, in the Indo-Pacific. Proceedings of the 5th Indo-Pacific Fish Conference. p. 209.

Horikoshi, K., H. Suganuma, H. Tachikawa, F. Sato, and M. Yamaguchi. 1994. Decline of Ogasawara green turtle population in Japan. Proceedings of the Fourteenth Annual Symposium on Sea Turtle Biology and Conservation. Miami, FL: US Department of Commerce. NOAA Technical Memorandum NMFS-SEFSC-351. p. 235-236. https://repository.library.noaa.gov/view/noaa/6185

Horwood, J. 1987. The sei whale: Population biology, ecology & management. Routledge.

Howell, E. A., P. H. Dutton, J. J. Polovina, H. Bailey, D. M. Parker, and G. H. Balazs. 2010. Oceanographic influences on the dive behavior of juvenile loggerhead turtles (*Caretta caretta*) in the North Pacific Ocean. Marine Biology. 157(5):1011-1026.

Howell, E. A., P. H. Dutton, J. J. Polovina, H. Bailey, D. M. Parker, and G. H. Balazs. 2015. Enhancing the TurtleWatch product for leatherback sea turtles, a dynamic habitat model for ecosystem-based management. Fisheries Oceanography. 24(1):57-68.

Howell, E. A., D. R. Kobayashi, D. M. Parker, G. H. Balazs, and A. Polovina. 2008. TurtleWatch: a tool to aid in the bycatch reduction of loggerhead turtles *Caretta caretta* in the Hawaii-based pelagic longline fishery. Endangered Species Research. 5:267-278.

Howey-Jordan, L. A., E. J. Brooks, D. L. Abercrombie, L. K. Jordan, A. Brooks, S. Williams, E. Gospodarczyk, and D. D. Chapman. 2013. Complex movements, philopatry and expanded depth range of a severely threatened pelagic shark, the oceanic whitetip (*Carcharhinus longimanus*) in the western North Atlantic. PLoS One. 8(2):e56588.

Howey, L. A., E. R. Tolentino, Y. P. Papastamatiou, E. J. Brooks, D. L. Abercrombie, Y. Y. Watanabe, S. Williams, A. Brooks, D. D. Chapman, and L. K. B. Jordan. 2016. Into the deep: the functionality of mesopelagic excursions by an oceanic apex predator. Ecology and Evolution. 6(15):5290-5304.

Humber, F., B. J. Godley, and A. C. Broderick. 2014. So excellent a fishe: a global overview of legal marine turtle fisheries. Diversity and Distributions. 20(5):579-590.

Hutchinson, J., and M. Simmonds. 1992. Escalation of threats on marine turtles. Oryx. 26(2):95-102.

Hutchinson, M. R., D. G. Itano, J. A. Muir, and K. M. Holland. 2015. Post-release survival of juvenile silky sharks captured in a tropical tuna purse seine fishery. Marine Ecology Progress Series. 521:143-154.

Hutchinson, M., F. Poisson, and Y. Swimmer. 2017. Developing best handling practice guidelines to safely release mantas and mobulids captured in commercial fisheries. 13th Regular Session of the Scientific Committee. 9–17 August 2017, Rarotonga, Cook Islands. Information papers. WCPFC-SC13-2017/SA-IP-08.

Hutchinson, M., and K. Bigelow. 2019. Quantifying Post Release Mortality Rates of Sharks Incidentally Captured in Pacific Tuna Longline Fisheries and Identifying Handling Practices to Improve Survivorship. Scientific Committee Fifteenth Regular Session. Pohnpei, Federated States of Micronesia. WCPFC-SC15-2019/EB-WP-04 (Rev.01). 26 p.

Hutchinson, M., Z. Siders, J. Stahl, and K. Bigelow. 2021. Quantitative estimates of post-release survival rates of sharks captured in Pacific tuna longline fisheries reveal handling and discard practices that improve survivorship. PIFSC Data Report, DR-21-001, 56 p.

Hussar, E., S. Richards, Z.-Q. Lin, R. P. Dixon, K. A. J. W. Johnson, Air,, and S. Pollution. 2012. Human health risk assessment of 16 priority polycyclic aromatic hydrocarbons in soils of Chattanooga, Tennessee, USA. 223(9):5535-5548.

IATTC (Inter-American Tropical Tuna Commission). 2012. Conservation status and habitat use of sea turtles in the eastern Pacific Ocean. p. 28.

IATTC. 2021. Scientific Advisory Committee 12th Meeting, 10-14 May 2021. SAC-12-09. Improving species and catch data reporting (Resolution C-03-05). 49 p.

Ikonomopoulou, M. P., H. Olszowy, R. Francis, K. Ibrahim, and J. Whittier. 2013. Accumulation of trace metals in the embryos and hatchlings of *Chelonia mydas* from peninsular Malaysia incubated at different temperatures. Science of the Total Environment. 450-451:301-306.

Incardona, J. P., T. K. Collier, and N. L. Scholz. 2004. Defects in cardiac function precede morphological abnormalities in fish embryos exposed to polycyclic aromatic hydrocarbons. Toxicology Applied Pharmacology. 196(2):191-205.

IOTC (Indian Ocean Tuna Commission). 2011. Report of the Fourteenth Session of the IOTC Scientific Committee. IOTC–2011–SC14–R[E]. 259 p.

IOTC. 2015. Status of the Indian Ocean oceanic whitetip shark (OCS: *Carcharhinus longimanus*). IOTC–2015–SC18–ES18[E]. p. 7.

IOTC. 2016. Status of the Indian Ocean Scalloped Hammerhead Shark (SPL: *Sphyrna lewini*). IOTC–2016–SC19–ES19[E].

IPCC (Intergovernmental Panel on Climate Change). 2014. Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (eds.)]. Geneva, Switzerland. p. 151.

IPCC. 2018. Summary for Policymakers. In: Masson-Delmotte V, Zhai P, Portner H-O, Roberts D, Skea J, Shukla PR, Pirani A, Moufouma-Okia W, Pean C, Pidcock R *et al.* editors. Global Warming of 1.5°C. An IPCC Special Report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty. World Meteorological Organization, Geneva, Switzerland: 32

IPCC. 2021. Climate Change 2021 The Physical Science Bases: Working Group I Contribution to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press. Cambridge University Press, Cambridge, UK and New York, NY, USA, 2409 pp., doi:10.1017/9781009157896.

Isaac, J. L. 2009. Effects of climate change on life history: implications for extinction risk in mammals. Endangered Species Research. 7(2):115-123

Ishizaki, A. 2015. Protected Species Conservation by the Western Pacific Regional Fishery Mangagement Council. Pacific Islands Fishery Monographs No. 4, p. 24

Irvine, L., D. M. Palacios, J. Urban, and B. Mate. 2017. Sperm whale dive behavior characteristics derived from intermediate-duration archival tag data. Ecology Evolution. 7(19):7822-7837.

Ishihara, T., N. Kamezaki, Y. Matsuzawa, and A. Ishizaki. 2014. Assessing the status of Japanese coastal fisheries and sea turtle bycatch. Wildlife and Human Society. 2:23-35.

Islam, M. Z. 2001. Notes on the trade in marine turtle products in Bangladesh. Marine Turtle Newsletter. 94(10).

IWC (International Whaling Commission). 2021. Catches Taken: Special Permit. Available at: https://iwc.int/table_permit

Jambeck, J. R., R. Geyer, C. Wilcox, T. R. Siegler, M. Perryman, A. Andrady, R. Narayan, and K. L. Law. 2015. Plastic waste inputs from land into the ocean. Science. 347(6223):768-771.

Jaine, F., C. Rohner, S. Weeks, L. Couturier, M. Bennett, K. Townsend, and A. Richardson. 2014. Movements and habitat use of reef manta rays off eastern Australia: Offshore excursions, deep diving and eddy affinity revealed by satellite telemetry. Marine Ecology Progress Series 510: 73-86

Jensen, M. P. 2010. Assessing the composition of green turtle (*Chelonia mydas*) foraging grounds in Australasia using mixed stock analyses. Institute for Applied Ecology, University of Canberra. p. 142.

Jensen, M. P., F. A. Abreu-Grobois, J. Frydenberg, and V. Loeschcke. 2006. Microsatellites provide insight into contrasting mating patterns in arribada vs. non-arribada olive ridley sea turtle rookeries. Molecular Ecology. 15(9):2567-2575.

Jensen, M. P., C. D. Allen, T. Eguchi, I. P. Bell, E. L. LaCasella, W. A. Hilton, C. A. M. Hof, and P. H. Dutton. 2018. Environmental Warming and Feminization of One of the Largest Sea Turtle Populations in the World. Current Biology. 28(1):154-159 e154.

Jensen, M. P., C. J. Limpus, S. D. Whiting, M. Guinea, R. I. T. Prince, K. E. M. Dethmers, I. B. W. Adnyana, R. Kennett, and N. N. FitzSimmons. 2013. Defining olive ridley turtle *Lepidochelys olivacea* management units in Australia and assessing the potential impact of mortality in ghost nets. Endangered Species Research. 21(3):241-253.

Jino, N., H. Judge, O. Revoh, V. Pulekera, A. Grinham, S. Albert, and H. Jino. 2018. Community-based Conservation of Leatherback Turtles in Solomon Islands: Local Responses to Global Pressures. Conservation & Society. 16(4):459-466.

Jones, T. T., B. L. Bostrom, M. D. Hastings, K. S. Van Houtan, D. Pauly, and D. R. Jones. 2012. Resource requirements of the Pacific leatherback turtle population. PLoS One. 7(10).

Jones, T. T., S. Martin, T. Eguchi, B. Langseth, J. Baker, and A. Yau. 2018. Review of draft response to PRD’s request for information to support ESA section 7 consultation on the effects of Hawaii-based longline fisheries on ESA listed species. NMFS Pacific Islands Fisheries Science Center, Honolulu, HI. p. 35.

Joung, S. J., N. F. Chen, H. H. Hsu, and K. M. Liu. 2016. Estimates of life history parameters of the oceanic whitetip shark, *Carcharhinus longimanus*, in the Western North Pacific Ocean. Marine Biology Research. 12(7):758-768.

Jouppi, D. 2015. Jet ski fishing mahimahi (dorado) false killer whale attack Hawaii Feeding Whales Dusan Jouppi. Available at: https://m.youtube.com/watch?v=ugXggt5n-V4

Kabua, E., M. Langidrik, K. Boktok, D. M. Parker, and K. P. Frutchey. 2012. Marine turtle-fisheries interaction mitigation training, data collection activities and satellite telemetry in the Republic of the Marshall Islands. In: Belskis L, Frick M, Panagopoulou A, Rees AF, Williams K, editors. Proceedings of the Twenty-Ninth Annual Symposium on Sea Turtle Biology and Conservation. Brisbane, Australia: OAA Technical Memorandum NOAA NMFS-SEFSC-630. p. 120. https://repository.library.noaa.gov/view/noaa/4406

Kajiura, S. M. 2001. Head morphology and electrosensory pore distribution of carcharhinid and sphyrnid sharks. Environmental Biology of Fishes. 61(2):125-133.

Kaplan, I. C. 2005. A risk assessment for Pacific leatherback turtles (*Dermochelys coriacea*). Canadian Journal of Fisheries and Aquatic Sciences 62(8):1710-1719.

Kalb, H. J. 1999. Behavior and physiology of solitary and arribada nesting olive ridley sea turtles (*Lepidochelys olivacea*) during the internesting period. Texas A & M University.

Kamezaki, N., Y. Matsuzama, O. Abe, H. Asakawa, T. Fujii, K. Goto, S. Hagino, M. Hayami, M. Ishii, T. Iwamoto *et al.* 2003. Loggerhead turtles nesting in Japan. In: Bolten, A.B., B. E. Witherington, editors. Loggerhead sea turtles. Smithsonian Institution: Washington, DC.

Kaschner, K., D. P. Tittensor, J. Ready, T. Gerrodette, and B. Worm. 2011. Current and future patterns of global marine mammal biodiversity. PLoS One. 6(5):e19653.

Karl, T. R., J. M. Melillo, T. C. Peterson, and S. J. Hassol. 2009. Global climate change impacts in the United States. Cambridge University Press.

Kashiwagi, T., T. Ito, and F. Sato. 2010. Occurences of reef manta ray, *Manta alfredi*, and giant manta ray, *M. birostris*, in Japan, examined by photographic records. Report of Japanese Society for Elasmobranch Studies. 46:20-27.

Kaschner, K., D. P. Tittensor, J. Ready, T. Gerrodette, and B. Worm. 2011. Current and future patterns of global marine mammal biodiversity. PLoS One. 6(5):e19653.

Kasuya, T. 1986. False Killer Whales. In: Tamura, T., S. Ohsumi, S. Arai, editors. Report of investigation in search of solution for dolphin-fishery conflict in the Iki Island area. p. 23. Japan Fisheries Agency.

Kasuya, T. 1991. Density dependent growth in North Pacific sperm whales. Marine Mammal Science. 7(3):230-257.

Kelez, S., X. Velez-Zuazo, and C. Manrique. 2003. New evidence on the loggerhead sea turtle *Caretta caretta* (Linnaeus 1758) in Peru. Ecologia Aplicada. 2(1):141-142.

Keller, B. D., D. F. Gleason, E. McLeod, C. M. Woodley, S. Airame, B. D. Causey, A. M. Friedlander, R. Grober-Dunsmore, J. E. Johnson, S. L. Miller *et al.* 2009. Climate change, coral reef ecosystems, and management options for marine protected areas. Environmental Management. 44(6):1069-1088.

Kelly, I. K. 2020. A Review of vessel collision threat to sea turtles in Hawaii. NOAA-NMFS-PIRO Report, 18 pp.

Kiessling, I. 2003. Finding solutions: derelict fishing gear and other marine debris in northern Australia. National Oceans Office. p. 58.

Kittinger, J. N., J. M. Pandolfi, J. H. Blodgett, T. L. Hunt, H. Jiang, K. Maly, L. E. McClenachan, J. K. Schultz, and B. A. Wilcox. 2011. Historical reconstruction reveals recovery in Hawaiian coral reefs. PLoS One. 6(10):e25460.

Kittinger, J. N., K. S. Van Houtan, L. E. McClenachan, and A. L. Lawrence. 2013. Using historical data to assess the biogeography of population recovery. Ecography. 36(8):868-872.

Kinch, J. 2006. Socio-economic assessment study for the Huon Coast: Final Technical Report. Western Pacific Regional Fishery Management Council, Honolulu, Hawaii.

Kinch J, Anderson P, and A. K. 2009. Assessment of leatherback turtle nesting and consumptive use in the autonomous region of Bougainville, Papua New Guinea. Western Pacific Regional Fisheries Management Council.

Klain, S., J. Eberdong, A. Kitalong, Y. Yalap, E. Matthews, A. Eledui, M. Morris, W. Andrew, D. Albis, and P. Kemesong. 2007. Linking Micronesia and Southeast Asia: Palau sea turtle satellite tracking and flipper tag returns. Marine Turtle Newsletter. 118:9-11.

Kobayashi, D. R., I. J. Cheng, D. M. Parker, J. J. Polovina, N. Kamezaki, and G. H. Balazs. 2011. Loggerhead turtle (*Caretta caretta*) movement off the coast of Taiwan: characterization of a hotspot in the East China Sea and investigation of mesoscale eddies. ICES Journal of Marine Science. 68(4):707-718.

Kobayashi, D. R., J. J. Polovina, D. M. Parker, N. Kamezaki, I. J. Cheng, I. Uchida, P. H. Dutton, and G. H. Balazs. 2008. Pelagic habitat characterization of loggerhead sea turtles, *Caretta caretta*, in the North Pacific Ocean (1997–2006): Insights from satellite tag tracking and remotely sensed data. Journal of Experimental Marine Biology and Ecology. 356(1-2):96-114.

Kock, K.-H., M. G. Purves, and G. Duhamel. 2006. Interactions between cetacean and fisheries in the Southern Ocean. Polar Biology. 29(5):379-388.

Kohler, N. E., and P. A. Turner. 2001. Shark tagging: a review of conventional methods and studies. Environmental Biology of Fishes. 60:191-223.

Konrad, C. M., S. Gero, T. Frasier, and H. Whitehead. 2018. Kinship influences sperm whale social organization within, but generally not among, social units. Royal Society Open Science. 5(8):180914.

Kubota, M. 2004. A Mechanism for the Accumulation of Floating Marine Debris North of Hawaii. Journal of Physical Oceanography. 24(5):1059-1064.

Kuljis, A. 2014. Improving Wildlife Agency Decisions by Acknowledging and Explaining Policy Choices Embedded in Agency Science. Ecology Law Quarterly.377-401.

Kurniawan, N., and A. Gitayana. 2020. Why did the population of olive ridley turtle Lepidochelys olivacea (Eschscholtz, 1829) increase in Alas Purwo National Park's Beach, East Java, Indonesia. Russian Journal of Marine Biology 46: 338-345.

Laist, D. W., J. M. Coe, and K. O'Hara. 1999. Marine Debris Pollution. In: Twiss JR, Reeves RR, editors. Conservation and Management of Marine Mammals. Washington and London: Smithsonian Institution Press. p. 342-366.

Lammers, M. O., A. A. Pack, E. G. Lyman, and L. Espiritu. 2013. Trends in collisions between vessels and North Pacific humback whales (*Megaptera novaeangliae*) in Hawaiian waters (1975 – 2011). Journal of Cetacean Research and Management 13:73-80.

Last, P. R., and J. D. Stevens. 2009. Sharks and rays of Australia. CSIRO Publishing. p. 656.

Lawson, J. M., S. V. Fordham, M. P. O’Malley, L. N. Davidson, R. H. Walls, M. R. Heupel, G. Stevens, D. Fernando, A. Budziak, and C. A. Simpfendorfer. 2017. Sympathy for the devil: a conservation strategy for devil and manta rays. PeerJ. 5:e3027.

Lawson, T. 2011. Estimation of Catch Rates and Catches of Key Shark Species in Tuna Fisheries of the Western and Central Pacific Ocean Using Observer Data. Information Paper EB IP–02. Seventh Regular Session of the Scientific Committee of the WCPFC. Pohnpei, FSM. 9th–17th August. p. 52.

Learmonth, J. A., C. D. MacLeod, M. B. Santos, G. J. Pierce, H. Q. P. Crick, and R. A. Robinson. 2006. Potential effects of climate change on marine mammals. Oceanography and Marine Biology - an Annual Review. 44:431-464.

Leatherwood, S., R. R. Reeves, and W. F. Perrin. 1988. Whales, dolphins and porpoises of the Eastern North Pacific and adjacent Arctic waters. NOAA Technical Report, NMFS Circular. 444:245.

Lee, T. S. 1993. Summary of cetacean survey data collected between the years of 1974 and 1985. 184 p.

Lessa, R., R. Paglerani, and F. Santana. 1999a. Biology and morphometry of the oceanic whitetip shark, *Carcharhinus longimanus* (Carcharhinidae), off North-Eastern Brazil. Cybium: international journal of ichthyology. 23(4):353-368.

Lessa, R., F. M. Santana, and R. Paglerani. 1999b. Age, growth and stock structure of the oceanic whitetip shark, *Carcharhinus longimanus*, from the southwestern equatorial Atlantic. Fisheries Research. 42(1-2):21-30.

Lewis, S.A., N. Setiasih, F. Fahmi, D. Dharmadi, M. P. O'Malley, S. J. Campbell , M. Yusuf, and A. B. Sianipar. 2015. Assessing Indonesian manta and devil ray populations through historical landings and fishing community interviews. Peer J PrePrints 6:e1334v1.

Lewison, R. L., S. A. Freeman, and L. B. Crowder. 2004. Quantifying the effects of fisheries on threatened species: the impact of pelagic longlines on loggerhead and leatherback sea turtles. Ecology Letters. 7(3):221-231.

Lewison, R. L., and coauthors. 2014. Global patterns of marine mammal, seabird, and sea turtle bycatch reveal taxa-specific and cumulative megafauna hotspots. Proceedings of the National Academy of Sciences 111(14):5271-5276.

Limpus, C. J. 2009a. A biological review of Australian marine turtle species: Chapter 4, Olive Ridley. The State of Queensland, Environmental Protection Agency. p. 220-245.

Limpus, C. J. 2008. Olive Ridley turtle *Lepidochelys olivacea* (Eschscholtz). In: Fien L, editor. A biological review of Australian marine turtle species. The State of Queensland. Environmental Protection Agency.

Limpus, C. J. 2009b. A biological review of Australian marine turtles: Green turtle *Chelonia mydas* (Linnaeus). Queensland Environmental Protection Agency. p. 206.

Limpus, C. J., J. D. Miller, D. J. Limpus, and M. Hamann. 2002. The Raine Island Green Turtle Rookery: Y2K Update. In: Mosier A, Foley A, Brost B, editors. Proceedings of the Twentieth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-477. p. 132-134. https://repository.library.noaa.gov/view/noaa/3259

Liu, K. M., and C. Chen. 1999. Demographic analysis of the scalloped hammerhead, *Sphyrna lewini*, in northwestern Pacific. Fisheries Science. 65(2):218-223.

Liu, K,-M,, and W.-P. Tsai. 2011. Catch and life history parameters of pelagic sharks in the Northwestern Pacific. Keelung, Chinese Taipei, ISC Shark Working Group Workshop. p/ 12.

Lomolino, M. V., and R. Channell. 1995. Splendid isolation: patterns of range collapse in endangered mammals. Journal of Mammalogy 76: 335-347.

Lomolino, M. V., and R. Channel. 1998. Range collapse, re-introductions, and biogeographic guidelines for conservation. Conservation Biology. 12(2):481-484.

Lontoh, D. N. 2014. Variation in tissue stable isotopes, body size, and reproduction of Western Pacific leatherback turtles.

Luiz, O. J., A. P. Balboni, G. Kodja, M. Andrade, and H. Marum. 2009. Seasonal occurrences of *Manta birostris* (Chondrichthyes: Mobulidae) in southeastern Brazil. Ichthyological Research. 56(1):96-99.

Lutcavage, M.E., and P. L. Lutz . 1997. Diving Physiology. In: Lutz P, Musick J, editors. The Biology of Sea Turtles. Boca Raton, Florida: CRC Press. p. 277-295.

Lutcavage, M. E., P. Plotkin, P. Witherington, and P. L. Lutz. 1997. Human impacts on sea turtle survival. In: Lutz, P., J. Musick, editors. The Biology of Sea Turtles. p. 387-409. Boca Raton, Florida: CRC Press.

Lutcavage, M. E., P. L. Lutz, G. D. Bossart, and D. M. Hudson. 1995. Physiologic and clinicopathologic effects of crude oil on loggerhead sea turtles. Archives of Environmental Contamination Toxicology Applied Pharmacology. 28(4):417-422.

Lyons, K., A. Carlisle, A. Preti, C. Mull, M. Blasius, J. O'Sullivan, C. Winkler, and C. G. Lowe. 2013. Effects of trophic ecology and habitat use on maternal transfer of contaminants in four species of young of the year lamniform sharks. Marine Environtmental Research. 90:27-38.

Macbeth, W. G., P. T. Geraghty, V. M. Peddemors, and C. A. Gray. 2009. Observer-based study of targeted commercial fishing for large shark species in waters off northern New South

Wales. Northern Rivers Catchment Management Authority Project No. IS8-9-M-2. Industry and Investment NSW – Fisheries Final Report Series No. 114.

Macfadyen, G., T. Huntington, and R. Cappell. 2009. Abandoned, lost or otherwise discarded fishing gear. Food and Agriculture Organization of the United Nations (FAO). p. 21.

Mackas, D. L., R. Goldblatt, and A. G. Lewis. 1998. Interdecadal variation in developmental timing of *Neocalanus plumchrus* populations at Ocean Station P in the subarctic North Pacific. Canadian Journal of Fisheries and Aquatic Sciences. 55:1878–1893.

MacLeod, C. D. 2009. Global climate change, range changes and potential implications for the conservation of marine cetaceans: a review and synthesis. Endangered Species Research. 7:125-136.

Madigan DJ, Brooks EJ, Bond ME, Gelsleichter J, Howey LA, Abercrombie DL, Brooks A, and Chapman DD. 2015. Diet shift and site-fidelity of oceanic whitetip sharks *Carcharhinus longimanus* along the Great Bahama Bank. Marine Ecology Progress Series. 529:185-197.

Maison, K. A., I. K. Kelley, and K. P. Frutchey. 2010. Green Turtle Nesting Sites and Sea Turtle Legislation throughout Oceania. U.S. Dep. Commerce, NOAA Technical Memorandum. NMFS-F/SPO-110. p. 52. https://repository.library.noaa.gov/view/noaa/3961

Mancini, A., and V. Koch. 2009. Sea turtle consumption and black market trade in Baja California Sur, Mexico. Endangered Species Research. 7:1-10.

Mancini, A., V. Koch, J. A. Seminoff, and B. Madon. 2012. Small-scale gill-net fisheries cause massive green turtle *Chelonia mydas* mortality in Baja California Sur, Mexico. Oryx. 46(01):69-77.

MantaMatcher. 2016. Facebook: Manta Matcher - the Wildbook for Manta Rays. https://www.facebook.com/mantamatcher/

Mantua, N. J., S. R. Hare, Y. Zhang, J. M. Wallace, and R. C. Francis. 1997. A Pacific Interdecadal Climate Oscillation with Impacts on Salmon Production. Bulletin of the American Meteorological Society. 78(6):1069-1080.

Marquez, R., C. Penaflores, and J. Vasconcelos. 1996. Olive ridley turtles (*Lepidochelys olivacea*) show signs of recovery at La Escobilla, Oaxaca. Marine Turtle Newsletter. 73:5-7.

Marshall, A., and J. Conradie. 2014. Manta Fishery Solor. Marine Megafauna Foundation.

Marshall, A., T. Kashiwagi, M.B. Bennett, M. Deakos, G. Stevens, F. McGregor, T. Clark, H. Ishihara, and K. Sato. 2018. *Mobula alfredi* (amended version of 2011 assessment). The IUCN Red List of Threatened Species 2018: e.T195459A126665723.

Marshall, A.D., M. B. Bennett, G. Kodja, S. Hinojosa-Alvarez, F. Galvan-Magana, M. Harding, G. Stevens, and T. Kashiwagi. 2011. *Mobula birostris*. Available at: http://www.iucnredlist.org/details/198921/0

Marshall, A. D., and M. B. Bennett. 2010. Reproductive ecology of the reef manta ray *Manta alfredi* in southern Mozambique. Journal of fish biology. 77(1):169-190.

Marshall, A. D., L. J. V. Compagno, and M. B. Bennett. 2009. Redescription of the genus Manta with resurrection of *Manta alfredi* (Krefft, 1868)(Chondrichthyes; Myliobatoidei; Mobulidae). Zootaxa. 2301:1-28.

Martien, K. K., B. L. Taylor, S. J. Chivers, S. D. Mahaffy, A. M. Gorgone, and R. W. Baird. 2019. Fidelity to natal social groups and mating within and between social groups in an endangered false killer whale population. Endang Species Res 40:219-230.

Martien, K.K., S.J. Chivers, R.W. Baird, F.I. Archer, A.M. Gorgone, B.L. Hancock-Hanser, D. Mattila, D. J. McSweeney, E. M. Oleson, C. Palmer, V. L. Pease, K. M. Robertson, G. S. Schorr, M. B. Schultz, D. L. Webster, and B. L. Taylor, 2014. Nuclear and Mitochondrial Patterns of Population Structure in North Pacific False Killer Whales (*Pseudorca crassidens*), *Journal of Heredity*, Volume 105, Issue 5, September-October 2014, Pages 611–626, https://doi.org/10.1093/jhered/esu029

Martien, K, R.W. Baird, B.L. Taylor, E.M. Oleson, S.J. Chivers. 2011. Population structure and mechanisms of gene flow within island-associated false killer whales (*Pseudorca crassidens*) around the Hawaiian Archipelago. PSRG-11-14, 19pp.

Martin, S. L., S. M. Stohs, and J. E. Moore. 2015. Bayesian inference and assessment for rare-event bycatch in marine fisheries: a drift gillnet fishery case study. Ecological Applications. 25(2):416-429.

Martin, S.L., Siders, Z., Eguchi, T., Langseth. B., Ahrens, R., and T. T. Jones. 2020a. Assessing the population-level impacts of North Pacific loggerhead and western Pacific leatherback turtle interactions in the Hawaii-based shallow-set longline fishery. NOAA-TM-NMFS-PIFSC-95, 183 p. https://repository.library.noaa.gov/view/noaa/23037

Martin, S.L., Z. Siders, T. Eguchi, B. Langseth, A. Yau, J. Baker, R. Ahrens, TT. Jones. 2020b. Update to assessing the population-level impacts of North Pacific loggerhead and western Pacific leatherback turtle interactions: inclusion of the Hawaii-based deep-set and American Samoa-based longline fisheries. U.S. Dept. of Commerce, NOAA Technical Memorandum NOAA-TM-NMFS-PIFSC-101, 62 p. https://repository.library.noaa.gov/view/noaa/24251

Martínez, L. S., and coauthors. 2007. Conservation and biology of the leatherback turtle in the Mexican Pacific. Chelonian Conservation and Biology 6(1):70-78.

Mas, F., R. Forselledo, and A. Domingo. 2015. Mobulid ray by-catch in longline fisheries in the south-western Atlantic Ocean. Marine and Freshwater Research. 66(9):767-777.

Mastrangelo, G., E. Fadda, and V. Marzia. 1996. Polycyclic aromatic hydrocarbons and cancer in man. Environmental Health Perspectives. 104(11):1166-1170.

Matsuzawa, Y., N. Kamezaki, T. Ishihara, K. Omuta, H. Takeshita, K. Goto, T. Arata, H. Honda, K. Kameda, and Y. Kashima. 2016. Fine-scale genetic population structure of loggerhead turtles in the Northwest Pacific. Endangered Species Research. 30:83-93.

Maulany, R.I., G.S. Baxter, D.T. Booth, and R.J. Spencer. 2017. Population viability analysis (PVA) for olive ridley turtles (*Lepidochelys olivacea*) nesting in Alas Purwo National Park, Indonesia. The Malaysian Forester 80: 198-217.

Maximenko, N., J. Hafner, and P. Niiler. 2012. Pathways of marine debris derived from trajectories of Lagrangian drifters. Marine Pollution Bulletin. 65(1-3):51-62.

Mazaris, A. D., A. S. Kallimanis, S. P. Sgardelis, and J. D. Pantis. 2008. Do long-term changes in sea surface temperature at the breeding areas affect the breeding dates and reproduction performance of Mediterranean loggerhead turtles? Implications for climate change. Journal of Experimental Marine Biology and Ecology. 367(2):219-226.

McCarty, J. P. 2001. Ecological consequences of recent climate change. Conservation Biology. 15(2):320-331.

McCauley, R. D., J. Fewtrell, A. J. Duncan, C. Jenner, M. N. Jenner, J. D. Penrose, R. I. T. Prince, A. Adhitya, J. Murdoch, and K. McCabe. 2000. Marine seismic surveys—a study of environmental implications. The APPEA Journal. 40(1):692-708.

McCracken, M.L. 2000. Estimation of sea turtle take and mortality in the Hawaiian longline fisheries. National Marine Fisheries Service, Southwest Fisheries Science Center, Administrative Report H-00-06.29.

McCracken, M. L. 2016. Assessment of Incidental Interactions with Marine Mammals in the Hawaii Longline Deep and Shallow-set Fisheries from 2010 through 2014. 2 p.

McCracken, M. L. 2019a. Hawaii Permitted Deep-set Longline Fishery Estimated Anticipated Take Levels for Endangered Species Act Listed Species and Estimated Anticipated Dead or Serious Injury. PIFSC Data Report DR-19-011. p. 26.

McCracken, M. L. 2019b. American Samoa Longline Fishery Estimated Anticipated Take Levels for Endangered Species Act Listed Species. Pacific Island Fisheries Science Center. Honolulu, HI. 23 p.

McCracken, M. L. and B. Cooper. 2020a. Data Report Estimation of Bycatch with Bony Fish, Sharks, and Rays in the 2017, 2018, and 2019 Hawaii Permitted Deep-set Longline Fishery. PIFSC Data Report DR-20-023. Pacific Island Fisheries Science Center. Honolulu, HI. 1 p.

McCracken, M. L. and B. Cooper. 2020b. Data Report Hawaii Longline Fishery 2019 Seabird and Sea Turtle Bycatch for the Entire Fishing Grounds, Within the IATTC Convention Area, and Seabird Bycatch for above 23°N and 23°N–30°S. Pacific Island Fisheries Science Center. Honolulu, HI. 4 p.

McCracken, M. L. and B. Cooper. 2021a. Hawaii Longline Fishery 2020 Seabird and Sea Turtle Bycatch for the Entire Fishing Grounds, Within the IATTC Convention Area, and Seabird Bycatch to the north of 23°N and 23°N–30°S. Pacific Island Fisheries Science Center. Honolulu, HI. 11 p.

McCracken, M. L. and B. Cooper. 2021b. Estimation of Bycatch with Bony Fish, Sharks, and Rays in the 2020 Hawaii Permitted Deep-set Longline Fishery. Pacific Island Fisheries Science Center. Honolulu, HI. 4 p.

McCracken, M. L. and B. Cooper. 2022a. Hawaii Longline Fishery 2021 Seabird and Sea Turtle Bycatch for the Entire Fishing Grounds, Within the IATTC Convention Area, and Seabird Bycatch to the North of 23° N and 23 °N–30° S . Pacific Island Fisheries Science Center. Honolulu, HI. 4 p.

McCracken, M. L. and B. Cooper. 2022b. Estimation of Bycatch with Bony Fish, Sharks, and Rays in the 2021 Hawaii Permitted Deep-Set Longline Fishery. Pacific Island Fisheries Science Center. Honolulu, HI. 1 p.

McCracken, M. L., and K. A. Forney. 2010. Preliminary Assessment of Incidental Interactions with Marine Mammals in the Hawaii Longline Deep and Shallow Set Fisheries. NMFS. p. 27.

McDonald, M. A., and C. G. Fox. 1999. Passive acoustic methods applied to fin whale population density estimation. The Journal of the Acoustical Society of America. 105(5):2643-2651.

McKenna, M. F., D. Ross, S. M. Wiggins, and J. A. Hildebrand. 2012. Underwater radiated noise from modern commercial ships. Acoustical Society of America. 131(1):92-103.

McLaskey, A. K., J. E. Keister, P. McElhany, M. Brady Olson, D. Shallin Busch, M. Maher, and A. K. Winans. 2016. Development of *Euphausia pacifica* (krill) larvae is impaired under pCO2 levels currently observed in the Northeast Pacific. Marine Ecology Progress Series. 555:65-78.

McMahon, C. R., and G. C. Hays. 2006. Thermal niche, large-scale movements and implications of climate change for a critically endangered marine vertebrate. Global Change Biology. 12(7):1330-1338.

Medeiros, A. Mm, O. J. Luiz, and C. Domit. 2015. Occurrence and use of an estuarine habitat by giant manta ray *Manta birostris*. Journal of fish biology. 86(6):1830-1838.

Merchant-Larios, H., S. Ruiz-Ramirez, N. Moreno-Mendoza, and A. Marmolejo-Valencia. 1997. Correlation among thermosensitive period, estradiol response, and gonad differentiation in the sea turtle *Lepidochelys olivacea*. General Comparative Endocrinology. 107(3):373-385.

Methot Jr, R. and C. R. Wetzel. 2013. Stock synthesis: A biological and statistical framework forfish stock assessment and fishery management. *Fisheries Research*, *142*, 86–99.

Milessi, A. C., and M. C. Oddone . 2003. Primer registro de *Manta birostris* (Donndorff 1798)(Batoidea: Mobulidae) en el Rio de La Plata, Uruguay. Gayana (Concepción). 67(1):126-129.

Miller, M. H., J. Carlson, P. Cooper, D. Kobayashi, M. Nammack, and J. Wilson. 2014. Status review report: scalloped hammerhead shark (*Sphyrna lewini*). Final Report to National Marine Fisheries Service, Office of Protected Resources. March 2014. 133 p.

Miller, M. H. 2018. Endangered Species Act Status Review Report: Chambered Nautilus (*Nautilus pompilius*). Report to National Marine Fisheries Service, Office of Protected Resources, Silver Spring, MD. September 2018. 62 p.

Miller, M. H., and C. Klimovich. 2017. Endangered Species Act Status Review Report: Giant Manta Ray (*Manta birostris*) and Reef Manta Ray (*Manta alfredi*). Final report to National Marine Fisheries Service, Office of Protected Resources, Silver Spring, MD. September 2017. p. 128.

Milton, S. L., S. Leone-Kabler, A. A. Schulman, and P. L. Lutz. 1994. Effects of Hurricane Andrew on the Sea Turtle Nesting Beaches of South Florida. Bulletin of Marine Science. 54(3):974-981.

Mitchelmore, C. L., C. A. Bishop, and T. K. Collier. 2017. Toxicological estimation of mortality of oceanic sea turtles oiled during the Deepwater Horizon oil spill. Endangered Species Research. 33:39-50.

MMC. 2007. Marine Mammals and Noise: A Sound Approach to Research and Management. A Report to Congress from the Marine Mammal Commission. p. 370.

Mizroch, S. A., D. W. Rice, and J. M. Breiwick. 1984. The sei whale, *Balaenoptera borealis*. Marine Fisheries Review. 46(4):25-29.

Moazzam, M. 2018. Unprecedented decline in the catches of mobulids: an important component of tuna gillnet fisheries of the Northern Arabian Sea. IOTC-2018-WPEB14-30. 7 p.

Mobley Jr, J. R., S. S. Spitz, K. A. Forney, R. Grotefendt, and P. H. Forestell. 2000. Distribution and Abundance of Odontocete Species in Hawaiian Waters: Preliminary Results of 1993-98 Aerial Surveys. NMFS, Southwest Fisheries Science Center Administrative Report LJ-00-14C. 27 p.

Molony, B. 2007. Commonly Captured Sharks and Rays for Consideration by the Ecosystem and Bycatch SWG at SC3. In: Submitted at the 3rd Scientific Committee meeting of the Western and Central Pacific Fisheries Commission, EB-IP10. p. 14-23.

Moore, A. B. M. 2012. Records of poorly known batoid fishes from the north-western Indian Ocean (Chondrichthyes: Rhynchobatidae, Rhinobatidae, Dasyatidae, Mobulidae). African Journal of Marine Science. 34(2):297-301.

Moore, M., D. Livingstone, and J. Widdows. 1989. Hydrocarbons in marine mollusks: biological effects and ecological consequences. CRC Press Inc.: Boca Raton, FL.

Moore, J. E., and J. Barlow. 2017. Population abundance and trend estimates for Beaked whales and Sperm whales in the California Current from Ship-based visual line-transect survey data, 1991-2014. p. 16.

Morgan, A., and G. H. Burgess. 2007. At-vessel fishing mortality for six species of sharks caught in the Northwest Atlantic and Gulf of Mexico. Gulf Caribbean Research. 19(2):123-129.

Mourier, J. 2012. Manta rays in the Marquesas Islands: first records of *Manta birostris* in French Polynesia and most easterly location of *Manta alfredi* in the Pacific Ocean, with notes on their distribution. Journal of fish biology. 81(6):2053-2058.

Moyes, C. D., N. Fragoso, M. K. Musyl, and R. W. Brill. 2006. Predicting postrelease survival in large pelagic fish. Transactions of the American Fisheries Society. 135(5):1389-1397.

Mrosovsky, N. 1994. Sex ratios of sea turtles. The Journal of Experimental Zoology 270:16-27.

Munday, P. L., J. M. Leis, J. M. Lough, C. B. Paris, M. J. Kingsford, M. L. Berumen, and J. Lambrechts. 2009. Climate change and coral reef connectivity. Coral Reefs. 28(2):379-395.

Munoz, C. C., and P. Vermeiren. 2020. Maternal Transfer of Persistent Organic Pollutants to Sea Turtle Eggs: A Meta-Analysis Addressing Knowledge and Data Gaps Toward an Improved Synthesis of Research Outputs. Environmental Toxicology Chemistry. 39(1):9-29.

Munoz, C. C., A. J. Hendriks, A. M. J. Ragas, and P. Vermeiren. 2021. Internal and Maternal Distribution of Persistent Organic Pollutants in Sea Turtle Tissues: A Meta-Analysis. Environmental Science Technology. 55(14):10012-10024.

Murray, K. T. 2011. Interactions between sea turtles and dredge gear in the US sea scallop (*Placopecten magellanicus*) fishery, 2001–2008. Fisheries Research. 107(1-3):137-146.

Muto, M. M., V. T. Helker, R. P. Angliss, B. A. Allen, P. L. Boveng, J. M. Breiwick, M. F. Cameron, P. J. Clapham, S. P. Dahle, M. E. Dahlheim et al. 2021. Sperm whale (*Physeter macrocephalus*): North Pacific Stock. NMFS Alaska Fisheries Science Center. NOAA-TM-AFSC-421. 7 p.

Myers, R.A., G. Mertz, and P.S. Fowlow. 1997. Maximum population growth rates and recovery times for Atlantic cod, *Gadus morhua*. Fishery Bulletin 95: 762-772.

Nakano, H., M. Okazaki, and H. Okamota. 1997. Analysis of catch depth by species for tuna longline fishery based on catch by branch lines. Bulletin of the Natural Resources Institute, Far Seas Fishery. (34):43-62.

Narazaki, T., K. Sato, and N. Miyazaki. 2015. Summer migration to temperate foraging habitats and active winter diving of juvenile loggerhead turtles *Caretta caretta* in the western North Pacific. Marine Biology. 162(6):1251-1263.

National Oceanic and Atmospheric Administration Marine Debris Program. 2014. Entanglement of Marine Species in Marine Debris with an Emphasis on Species in the United States. Silver Spring, MD. p. 28.

Nellis, D. W., and S. E. Henke. 2000. Predation of Leatherback Turtle Hatchlings by Near Shore Aquatic Predators. In: Kalb HJ, Wibbels T, editors. Proceedings of the Nineteenth Annual Symposium on Sea Turtle Biology and Conservation. U.S. Dept. Commerce. NOAA Tech. Memo. NMFS-SEFSC-443. 291 p. https://repository.library.noaa.gov/view/noaa/4730

Newcombe, R. G. 1998. Two-sided confidence intervals for the single proportion: comparison of seven methods. Statistics in medicine. 17(8):857-872.

Newson, S. E., S. Mendes, H. Q. Crick, N. Dulvy, J. D. Houghton, G. C. Hays, A. M. Hutson, C. D. Macleod, G. J. Pierce, and R. A. Robinson. 2009. Indicators of the impact of climate change on migratory species. Endangered Species Research. 7:101-113.

Nicholson-Jack, A. 2020. A hitchhiker's guide to manta rays – Patterns of association between *Mobula alfredi* and *M. birostris* and their symbionts in the Maldives. University of Bristol. p. 53.

Nishizawa, H., O. Abe, J. Okuyama, M. N. Kobayashi, and N. Arai. 2011. Population genetic structure and implications for natal philopatry of nesting green turtles *Chelonia mydas* in the Yaeyama Islands, Japan. Endangered Species Research. 14(2):141-148.

Nitta, E. T., and J. R. Henderson. 1993. A review of interactions between Hawaii's fisheries and protected species. Marine Fisheries Review. 55(2):83-92.

NMFS (National Marine Fisheries Service). 1999. Biological Opinion. Interim final rule for the continued authorization of the United States tuna purse seine fishery in the eastern tropical Pacific Ocean under the Marine Mammal Protection Act and the Tuna Conventions Act as revised by the International Dolphin Conservation Program Act. NMFS, Southwest Regional Office, Protected Resources Division. p. 60.

NMFS. 2001. Final Environmental Impact Statement for Fishery Management Plan, Pelagic Fisheries of the Western Pacific Region. Pacific Islands Region.

NMFS. 2004a. Biological Opinion on Proposed Regulatory Amendments to the Fisheries Management Plan for the Pelagic Fisheries of the Western Pacific Region. p. 281.

NMFS. 2004b. Memorandum for the Record. Annual Observed Sea Turtle Mortality in the U.S. Tuna Purse Seine Fishery in the Eastern Tropical Pacific Ocean. NMFS, Southwest Regional Office, Protected Resources Division. p. 7.

NMFS. 2005. Biological opinion on the continued authorization of the Hawaii-based Pelagic, Deep-set, Tuna Longline Fishery based on the Fishery Management Plan for Pelagic Fisheries of the Western Pacific Region. Pacific Islands Region. p. 168.

NMFS. 2006. Endangered Species Act Section 7 Consultation Biological Opinion and Incidental Take Statement: Formal Consultation on the Continued Operation of the Diablo Canyon Nuclear Power Plant and San Onofre Nuclear Generating Station. NMFS, Southwest Regional Office, Protected Resources Division. p. 59.

NMFS. 2007. Recovery Plan for the Hawaiian Monk Seal (*Monachus schauinslandi*). Second Revision. National Marine Fisheries Service, Silver Spring, MD. 165 p.

NMFS. 2008. Biological Opinion on effects of Implementation of Bottomfish Fishing Regulations within Federal Waters of the Main Hawaiian Islands on ESA-listed marine species. Pacific Islands Region. 35 p.

NMFS. 2010. Final Recovery Plan for the Fin Whale (*Balaenoptera physalus*). Office of Protected Resources. Silver Spring, MD. 121 p.

NMFS. 2011. Endangered and Threatened Species: Determination of Nine Distinct Population Segments of Loggerhead Sea Turtles as Endangered or Threatened. 76 FR 58867. 86 p.

NMFS. 2012a. 2012 Stock Assessment and Fishery Evaluation report (SAFE) for Atlantic Highly Migratory Species. Atlantic Highly Migratory Species Management Division. NOAA Fisheries. U.S. Department of Commerce. p. 219.

NMFS. 2012b. Biological opinion on continued operation of the Hawaii-based Shallow-set Longline Swordfish Fishery – under Amendment 18 to the Fishery Management Plan for Pelagic Fisheries of the Western Pacific Region. p. 162.

NMFS. 2013. Biological Opinion on the continued management of the drift gillnet fishery under the Fishery Management Plan for U.S. West Coast Fisheries for Highly Migratory Species. 2012/03020:DDL. Southwest Region. p. 158.

NMFS. 2014. Endangered Species Act - Section 7 Consultation Biological Opinion on the continued operation of the Hawaii-based deep-set pelagic longline fishery. Pacific Island Region. p. 216.

NMFS. 2015a. Reinitiation: Deep- and shallow-set longline fishery effects on monk seal critical habitat and fin whales Final LOC. NMFS PIRO PRD. Honolulu, HI. p. 6.

NMFS. 2015b. Sperm Whale (*Physeter macrocephalus*) 5 year Review: Summary and Evaluation. 93 p.

NMFS. 2015c. Endangered Species Act Section 7 Consultation on the Continued Operation of the American Samoa Longline Fishery. NMFS, Pacific Island Regional Office. 6 p.NMFS. 2016a. Biological Opinion for the Continued Operation of the West Coast-based Deep-seet Longline Fishery managed under the Fishery Management Plan for U. S. West Coast Highly Migratory Species (WCR-2015-2948). p. 90.

NMFS. 2016b. Revised guidance for treatment of climate change in NMFS Endangered Species Act decisions. 8 p.

NMFS. 2016c. Biological Opinion on the Use of Dispersants and In-Situ Burning in the United States (U.S.) Caribbean National Marine Fisheries Service Office of Protected Resources. 139 p.

NMFS. 2017a. Biological Opinion on (1) U.S. Military Mariana Islands Training and Testing Activities; and (2) the NMFS's Promulgation of Regulations and Issuance of a Letter of Authorization Pursuant to the MMPA. Office of Protected Resources. Silver Spring, MD. p. 569.

NMFS. 2017b. Supplement to the 2014 Biological Opinion on the continued operation of the Hawaii-based deep-set pelagic longline fishery. NMFS Protected Resources. Honolulu, HI. 133 p.

NMFS. 2017c. North Pacific Right Whale (*Eubalaena japonica*) Five-Year Review: Summary and Evaluation. National Marine Fisheries Service Office of Protected Resources Alaska Region. 39 p.

NMFS. 2018a. Biological Evaluation on the Potential Effects of the Hawaii Shallow-set Pelagic Longline Fishery on Endangered Species Act Listed Species and their Designated Critical Habitat. Honolulu, Hawaii. p. 68.

NMFS. 2018b. Biological Evaluation: Potential Effects of the Hawaii Deep-set Pelagic Longline Fishery on Endangered Species Act Listed Species and their Designated Critical Habitat. p. 78.

NMFS. 2018c. Biological Opinion on the National Science Foundation-funded seismic survey in the North Pacific Ocean, and issuance of an Incidental Harassment Authorization pursuant to section 101(a)(5)(D) of the Marine Mammal Protection Act (MMPA). Office of Protected Resources. Silver Spring, MD.

NMFS. 2018d. Biological Opinion on U.S. Navy Hawaii-Southern California Training and Testing and the National Marine Fisheries Service's Promulgation of Regulations Pursuant to the Marine Mammal Protection Act for the Navy to "Take" Marine Mammals Incidental to Hawaii-Southern California Training and Testing. Office of Protected Resources. Silver Spring, MD. p. 683.

NMFS. 2018e. Endangered and Threatened Wildlife and Plants: Final Rulemaking To Designate Critical Habitat for the Main Hawaiian Islands Insular False Killer Whale Distinct Population Segment. 83 FR 35062. p. 34.

NMFS. 2018f. Pacific Islands Regional Observer Program Deep-set Quarterly Status Report from April 1, 2018-June 30, 2018. July 27, 2018. NMFS Pacific Islands Regional Office, Honolulu, HI.

NMFS. 2018g. U.S. West Coast Fisheries Summary–Tina Fahy. For Pacific Leatherback meeting May 9-10, 2018. 7 p.

NMFS. 2019a. Biological Opinion. Continued Authorization of the Hawaii Shallow-set Longline Fishery. NMFS, Pacific Island Regional Office, Honolulu, HI. 506 p.

NMFS. 2019b. Endangered Species Act (ESA) Section 7(a)(2) Biological Opinion and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Response and [Fish and Wildlife Coordination Act Recommendations]- Ammunition Pier and Turning Basin Construction Project at Naval Weapons Station Seal Beach. NMFS, West Coast Regional Office, Seattle, WA. 70 p.

NMFS. 2020a. Hawaii Pacific Longline Fishing: Regulation Summary. NOAA/NMFS/PIRO/SFD. https://media.fisheries.noaa.gov/dam-migration/hawaii-longline-reg-summary.pdf

NMFS 2020b. M. Tosatto Memo to the Record: Endangered Species Act Section 7 Consultation on the Continued Oepration of the American Samoa Pelagic Longline Fishery – Section 7(a)(2) and 7(d) Determinations; Likelihood of Jeopardy and Commitment of Resources during Consultation – Extension. 6 May 2020.

NMFS. 2020c. Continuing Operation of the Pacific Coast Groundfish Fishery (Reinitiation of consultation #NWR- 2012-876 – Humpback whale (*Megaptera novaeangliae*). Endangered Species Act (ESA) Section 7(a)(2) Biological Opinion and Section 7(a)(2) “Not Likely to Adversely Affect” Determination. National Marine Fisheries Service, West Coast Regional Office. WCRO-2018-01378. 83 p.

NMFS. 2021a. Biological Opinion on the Authorization of the United States Western and Central Pacific Ocean Purse Seine Fishery. National Marine Fisheries Service, Pacific Island Regional Office, Honolulu HI. 496 p.

NMFS. 2021b. Memorandum for A. Garrett from B. Harmen. Information about the effects of wire vs. nylon leaders on ESA-listed species in the Hawaii deep-set longline fishery. Pacific Islands Regional Office Sustainable Fisheries Division. 9 p.

NMFS. 2021c. Recovery Status Review for the Main Hawaiian Islands Insular False Killer Whale (*Pseudorca crassidens*) Distinct Population Segment. National Marine Fisheries Service, Pacific Island Regional Office, Honolulu HI. 117 p.

NMFS. 2021d. Biological Opinion on the reinitiation of Endangered Species Act (ESA) Section 7 Consultation on the Implementation of the Sea Turtle Conservation Regulations under the ESA and the Authorization of the Southeast U.S. Shrimp Fisheries in Federal Waters under the Magnuson-Stevens Fishery Management and Conservation Act (MSFMCA). National Marine Fisheries Service, Southeast Regional Office. SERO-2021-00087. 297 p.

NMFS. 2021e. Endangered Species Act Section 7 Consultation on the Atlantic Sea Scallop Fishery Management Plan. National Marine Fisheries Service, Greater Atlantic Regional Fisheries Office. GARFO-2020-00437. 310 p.

NMFS. 2022a. Main Hawaiian Islands Insular False Killer Whale (*Pseudorca crassidens*) Distinct Population Segment 5-Year Review: Summary and Evaluation. NMFS Pacific Islands Regional Office, Honolulu, Hawaii. 16 p.

NMFS. 2022b. Biological Opinion. Supplement to the Authorization of the Hawaii Deep-Set Longline Fishery; Effects to Oceanic Whitetip Sharks and Giant Manta Ray. NMFS, Pacific Island Regional Office, Honolulu, HI. 144 p.

NMFS. 2022c. Biological Opinion. Supplement to the Authorization of the American Samoa Longline Fishery; Effects to Oceanic Whitetip Sharks and Giant Manta Rays. NMFS, Pacific Island Regional Office, Honolulu, HI. 151 p.

NMFS 2023a. Biological Opinion on the Authorization of the American Samoa Longline Fishery. National Marine Fisheries Service, Pacific Island Regional Office, Honolulu HI. 336 p.

NMFS 2023b. Guidelines for distinguishing serious from non-serious injury of marine mammals pursuant to the Marine Mammal Protection Act. Protected Resources Policy Directive Process fo Distinguishing Serious from Non-Serious Injury 02-238. 55 p.

NMFS and USFWS. 1998a. Recovery Plan for U.S. Pacific Populations of the Green Turtle. Silver Spring, MD. p. 97.

NMFS and USFWS. 1998b. Recovery Plan for U.S. Pacific Populations of the Leatherback Turtle. Silver Spring, MD. p. 76.

NMFS and USFWS. 1998c. Recovery Plan for U.S. Pacific Populations of the Loggerhead Turtle (*Caretta caretta*). National Marine Fisheries Service, Office of Protected

Resources, Silver Spring MD and U.S. Fish and Wildlife Service, Southeast Region Jacksonville Ecological Services Office, Jacksonville, FL. p. 71.

NMFS and USFWS. 1998d. Recovery plan for U.S. Pacific populations of the olive ridley turtle (*Lepidochelys olivacea*). National Marine Fisheries Service, Silver Spring, Maryland. p. 52.

NMFS and USFWS. 2007a. Green sea turtle (*Chelonia mydas*) 5-year review: summary and evaluation. National Marine Fisheries Service, Office of Protected Resources, Silver Spring MD and U.S. Fish and Wildlife Service, Southeast Region Jacksonville Ecological Services Office, Jacksonville, FL. p. 105.

NMFS and USFWS. 2007b. Leatherback Sea Turtle (*Dermochelys coriacea*) 5-Year Review: Summary and Evaluation. National Marine Fisheries Service, Office of Protected Resources, Silver Spring MD and U.S. Fish and Wildlife Service, Southeast Region Jacksonville Ecological Services Office, Jacksonville, FL. p. 81.

NMFS and USFWS. 2007c. Loggerhead sea turtle (*Caretta caretta*) 5-Year Review: Summary and Evaluation. National Marine Fisheries Service, Office of Protected Resources, Silver Spring MD and U.S. Fish and Wildlife Service, Southeast Region Jacksonville Ecological Services Office, Jacksonville, FL. p. 67.

NMFS and USFWS. 2007d. Olive Ridley Sea Turtle (*Lepidochelys olivacea*) 5-Year Review: Summary and Evaluation. National Marine Fisheries Service, Office of Protected Resources, Silver Spring MD and U.S. Fish and Wildlife Service, Southeast Region Jacksonville Ecological Services Office, Jacksonville, FL. p. 67.

NMFS and USFWS. 2011. Endangered and Threatened Species; Determination of Nine Distinct Population Segments of Loggerhead Sea Turtles as Endangered or Threatened. 76 FR 58868. p. 85.

NMFS and USFWS. 2013. Leatherback Sea Turtle (*Dermochelys coriacea*) 5-Year Review: Summary and Evaluation. National Marine Fisheries Service, Office of Protected Resources, Silver Spring, MD, and U.S. Fish and Wildlife Service, Southeast Region, Jacksonville Ecological Services Office, Jacksonville, FL. p. 91.

NMFS and USFWS. 2014. Olive Ridley Sea Turtle (*Lepidochelys olivacea*) 5-Year Review Summary and Evaluation. National Marine Fisheries Service, Office of Protected Resources, Silver Spring, MD and U.S. Fish and Wildlife Service, Southeast Region Jacksonville Ecological Services Office, Jacksonville, FL. p. 87.

NMFS and USFWS. 2020a. Endangered Species Act status review of the leatherback turtle (*Dermochelys coriacea*). Report to the National Marine Fisheries Service, Office of Protected Resources, Silver Spring, MD, and U.S. Fish and Wildlife Service, Southeast Region, Jacksonville Ecological Services Office, Jacksonville, FL. 396 p.

NMFS and USFWS. 2020b. Loggerhead Sea Turtle (*Caretta caretta*) North Pacific Ocean DPS 5-Year Review: Summary and Evaluation. National Marine Fisheries Service, Office of Protected Resources, Silver Spring, MD and U.S. Fish and Wildlife Service, Southeast Region Jacksonville Ecological Services Office, Jacksonville, FL. p. 84.

NOAA (National Oceanic and Atmospheric Administration). 2015. Pacific Islands Marine Debris Encounter Reports 2007-2015. p. 2.

NOAA. 2016. Addendum to the Biennial Report to Congress Pursuant to Section 403(a) of the Magnuson-Stevens Fishery Conservation and Management Reauthorization Act of 2006. Positive Certification Determination for Mexico's 2013 Identification for Bycatch of North Pacific Loggerhead Sea Turtles. U.S. Department of Commerce. 1401 Constitution Avenue, N.W. Washington, D.C. 20230. p. 5.

NOAA Marine Debris Program. 2014 Report on the Entanglement of Marine Species in Marine Debris with an Emphasis on Species in the United States. Silver Spring, MD. 28 pp

Noriega, R., J. M. Werry,W. Sumpton, D. Mayer, and S. Y. Lee. 2011. Trends in annual CPUE and evidence of sex and size segregation of *Sphyrna lewini*: Management implications in coastal waters of northeastern Australia. Fisheries Research 110: 472-477.

Northrop, J., W. C. Cummings, and M. F. Morrison. 1971. Underwater 20-Hz Signals Recorded near Midway Island. The Journal of the Acoustical Society of America. 49(6B):1909-1910.

Notarbartolo-di-Sciara G, and E.V. Hillyer. 1989. Mobulid rays off eastern Venezuela (Chondrichthyes, Mobulidae). Copeia.607-614.

NRC (National Research Council). 1990a. Decline of the sea turtles: causes and prevention. Washington, D.C.: National Academies Press. p. 259.

NRC (National Research Council). 1990b. Sea Turtle Mortality Associated with Human Activities. Decline of the Sea Turtles: Causes and Prevention. National Academies Press. p. 74-117.

NRC (National Research Council). 2003. National Research Council (US) Committee on Potential Impacts of Ambient Noise in the Ocean on Marine Mammals. Ocean Noise and Marine Mammals. Washington (DC): National Academies Press (US).

Obura, D., and G. S. Ston. 2002. Phoenix Islands Summary of Marine and Terrestrial Assessments Conducted in the Republic of Kiribati, June 5 – July 10, 2002. New England Aquarium, Boston, MA, USA.

Odell, D. K., and K. M. McClune. 1999. False killer whale *Pseudorca crassidens* (Owen, 1846). In: Ridgway, S. H., R. Harrison, editors. Handbook of marine mammals. p. 213-243.

O'Grady, J. J., M. A. Burgman, D. A. Keith, L. L. Master, S. J. Andelman, B. W. Brook, G. A. Hammerson, T. Regan, and R. Frankham. 2004. Correlations among Extinction Risks Assessed by Different Systems of Threatened Species Categorization. Conservation Biology. 18(6):1624-1635.

Oleson, E. M., C. H. Boggs, K. A. Forney, M. B. Hanson, D. R. Kobayashi, B. L. Taylor, P. R. Wade, and G. M. Ylitalo. 2010. Status review of Hawaiian insular false killer whales (*Pseudorca crassidens*) under the Endangered Species Act.

Oleson, E. M., C. H. Boggs, K. A. Forney, M. B. Hanson, D. R. Kobayashi, B. L. Taylor, P. R. Wade, and G. M. Ylitalo. 2012. Reevaluation of the DPS Designation for Hawaiian (now

Main Hawaiian Islands) Insular False Killer Whales. PIFSC Internal Report IR-12-03. 39 p.

O'Malley, M. P., K. Lee-Brooks, and H. B. Medd. 2013. The global economic impact of manta ray watching tourism. PLoS One. 8(5):e65051.

Omar-Ali, A., C. Hohn, P. J. Allen, J. L. Rodriguez, and L. Petrie-Hanson. 2015. Tissue PAH, blood cell and tissue changes following exposure to water accommodated fractions of crude oil in alligator gar, *Atractosteus spatula*. Marine Environmental Research. 108:33-44.

Oros, J., A. Torrent, P. Calabuig, and S. Deniz. 2005. Diseases and causes of mortality among sea turtles stranded in the Canary Islands, Spain (1998–2001). Diseases of Aquatic Organisms. 63(1):13-24.

Orrego, C. M., and R. Arauz. 2004. Mortality of sea turtles along the Pacific Coast of Costa Rica. Proceedings of the Twenty-First Annual Symposium on Sea Turtle Biology and Conservation. NOAA Tech Memo NMFS-SEFSC-528. p. 265-266. https://repository.library.noaa.gov/view/noaa/3412

O'Shea, O. R., M. J. Kingsford, and J. Seymour. 2010. Tide-related periodicity of manta rays and sharks to cleaning stations on a coral reef. Marine and Freshwater Research. 61:65-73.

Pacoureau, N., Rigby, C.L., Kyne, P.M. et al. 2021. Half a century of global decline in oceanic sharks and rays. Nature 589:567–571.

Paig-Tran, E.W., T. Kleinteich, and A. P. Summers. 2013. The filter pads and filtration mechanisms of the devil rays: Variation at macro and microscopic scales. Journal of Morphology. 274(9):1026-1043.

Pakiding, F., Zohar, K., Allo, A.Y., Keroman, S., Lontoh, D., Dutton, P.H. and Tiwari, M. 2020. Community engagement: an integral component of a multifaceted conservation approach for the transboundary western Pacific leatherback. Frontiers in Marine Science, 7, p.549570.

Palacios, A. I. 2012a. Sea turtle stock and nesting assessment in the CNMI. Commonwealth of the Northern Mariana Islands, Department of Lands and Natural Resources, Saipan, CNMI.

Palacios, A. I. 2012b. CNMI Sea Turtle Program Annual Report January 1, 2011 - December 31, 2011. Commonwealth of the Northern Mariana Islands, Department of Lands & Natural Resources. p. 36.

Palacios, D. M., and B. R. Mate. 1996. Attack by false killer whales (*Pseudorca crassidens*) on sperm whales (*Physeter macrocephalus*) in the Galapagos Islands. Marine Mammal Science. 12(4):582-587.

Palau Bureau of Marine Resources. 2008. Palau Marine Turtle Conservation & Monitoring Program Final Report. p. 31.

Pandav, B., K. Bunugopan, D. Sutari, and B. C. Choudhury. 2000. Fidelity of male olive ridley sea turtles to a breeding ground. Marine Turtle Newsletter. 87:9-10.

Pandav, B., and B. C. Choudhury. 1999. An update on the mortality of the olive ridley sea turtles in Orissa, India. Marine Turtle Newsletter. 83:10-12.

Parker, D. M., and G. H. Balazs. 2010. Draft Map Guide to Marine Turtle Nesting and Basking in the Hawaiian Islands. NMFS. p. 29.

Parker, R. W., J. L. Blanchard, C. Gardner, B. S. Green, K. Hartmann, P. H. Tyedmers, and R. A. Watson. 2018. Fuel use and greenhouse gas emissions of world fisheries. Nature Climate Change. 8(4):333.

Parra, M., S. L. Deem, and E. Espinoza. 2011. Green Turtle (*Chelonia mydas*) Mortality in the Galápagos Islands, Ecuador During the 2009 – 2010 Nesting Season. Marine Turtle Newsletter. 130:10-15.

Pate, J. H., and A. D. Marshall. 2020. Urban manta rays: potential manta ray nursery habitat along a highly developed Florida coastline. Endangered Species Research. 43:51-64.

Patricio, A. R., A. Marques, C. Barbosa, A. C. Broderick, B. J. Godley, L. A. Hawkes, R. Rebelo, A. Regalla, and P. Catry. 2017. Balanced primary sex ratios and resilience to climate change in a major sea turtle population. Marine Ecology Progress Series. 577:189-203.

Payne, P., J. Nicholas, L. O'Brien, and K. Powers.1986. The distribution of the humpback whale, *Megaptera novaeangliae,* on Georges Bank and in the Gulf of Maine in relation to densities of the sand eel, Ammodytes americanus. Fisheries Bulletin. 84:271-277.

Payne, P. M., D. N. Wiley, S. B. Young, S. Pittman, P. J. Clapham, and J. W. Jossi. 1990. Recent Fluctuations in the Abundance of Baleen Whales in the Southern Gulf of Maine in Relation to Changes in Selected Prey. Fishery Bulletin. 88(4):687-696.

Peatman, T., V. Allain, S. Caillot, T. Park, P. Williams, I. Tuiloma, N. Smith, A. Panizza, and S. Fukofuka. 2018a. Summary of purse seine fishery bycatch at a regional scale, 2003-2017. Busan, Republic of Korea 8-16 August 2018. p. 13.

Peatman, T., L. Bell, V. Allain, P. Caillot, S. Williams, I. Tuiloma, A. Panizza, L. Tremblay-Boyer, S. Fukofuka, and N. Smith. 2018b. Summary of longline fishery bycatch at a regional scale, 2003-2017 Rev 2 (22 July 2018). Busan, Republic of Korea 8-16 August 2018. p. 61.

Peckham, H. P. 2010. Integrated initiative for the conservation of the North Pacific loggerhead sea turtle: threat assessment, threat mitigation, and management. ProPeninsula/Ocean Foundation. 4 p.

Peckham, S. H., and D. Maldonado-Diaz. 2012. Empowering small scale fishermen to be conservation heroes: A trinational fishermen's exchange to protect loggerhead turtles. In: Seminoff, J. A., B. P. Wallace, editors. Sea Turtles of the Eastern Pacific. University of Arizona, Tucson.

Peckham, S. H., D. Maldonado Diaz, A. Walli, G. Ruiz, L. B. Crowder, and W. J. Nichols. 2008. High mortality of loggerhead turtles due to bycatch, human consumption and strandings at Baja California Sur, Mexico, 2003 to 2007. Endangered Species Research. 5:171-183.

Peckham, S. H., D. Maldonado Diaz, V. Koch, A. Mancini, A. Gaos, M. T. Tinker, and W. J. Nichols. 2007. Small-scale fisheries bycatch jeopardizes endangered Pacific loggerhead turtles. PLoS One. 2(10):e1041.

Perez, C., A. Velando, I. Munilla, M. López-Alonso, and D. J. E. s. Oro. 2008. Monitoring polycyclic aromatic hydrocarbon pollution in the marine environment after the Prestige oil spill by means of seabird blood analysis. Environmental Science Technology. 42(3):707-713.

Petro, G., F. R. Hickey, and K. Mackay. 2007. Leatherback turtles in Vanuatu. Chelonian Conservation and Biology. 6(1):135-137.

Philip, M. 2002. Marine turtle conservation program Jamurba-Medi nesting beach, north coast of the Bird's Head Peninsula, Papua. Proceedings of the Western Pacific Sea Turtle Cooperative Research and Management Workshop. February 5–8, 2002.; Honolulu, Hawaii, USA. Western Pacific Regional Fishery Management Council. p. 143-146.

Piacenza, S. E., G. H. Balazs, S. K. Hargrove, P. M. Richards, and S. S. Heppell. 2016. Trends and variability in demographic indicators of a recovering population of green sea turtles *Chelonia mydas*. Endangered Species Research. 31:103-117.

Pichel, W. G., J. H. Churnside, T. S. Veenstra, D. G. Foley, K. S. Friedman, R. E. Brainard, J. B. Nicoll, Q. Zheng, and P. Clemente-Colon. 2007. Marine debris collects within the North Pacific Subtropical Convergence Zone. Marine Pollution Bulletin. 54(8):1207-1211.

Pike, D. A. 2014. Forecasting the viability of sea turtle eggs in a warming world. Global Change Biology. 20(1):7-15.

Pike, D. A., R. L. Antworth, and J. C. Stiner. 2006. Earlier nesting contributes to shorter nesting seasons for the Loggerhead Seaturtle, *Caretta caretta*. Journal of Herpetology. 40(1):91-94.

Pike, D. A., E. A. Roznik, and I. Bell. 2015. Nest inundation from sea-level rise threatens sea turtle population viability. Royal Society Open Science. 2(7):150127.

Pilcher, N. 2010. Community-based conservation of leatherback turtles along the Huon coast, Papua New Guinea. Final Project Report. Marine Research Foundation, Malaysia. p. 20.

Pilcher, N. 2011. Community-based conservation of leatherback turtles along the Huon coast, Papua New Guinea. Final Project Report. Marine Research Foundation, Malaysia. p. 13.

Piovano, S., A. Batibasaga, A. Ciriyawa, E. L. LaCasella, and P. H. Dutton. 2019. Mixed stock analysis of juvenile green turtles aggregating at two foraging grounds in Fiji reveals major contribution from the American Samoa Management Unit. Scientific Reports. 9(1):3150.

Pita, J., and D. Broderick. 2005. Hawksbill turtles in the Solomon Islands. In: Kinan I, editor. Proceedings of the Second Western Pacific Sea Turtle Cooperative Research and Management Workshop. Volume I: West Pacific Leatherback and Southwest Pacific Hawksbill Sea Turtles. Western Pacific Regional Fishery Management Council, Honolulu, Hawaii, USA. p. 101-102.

Pitman, R. L. 1991. Sea turtle associations with flotsam in the eastern tropical Pacific Ocean. In: Salmon M, Wyneken J, editors. Proceedings of the Eleventh Annual Workshop on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-302. https://repository.library.noaa.gov/view/noaa/6067

Pitman, R. L. 1993. Seabird associations with marine turtles in the eastern Pacific Ocean. Colonial Waterbirds. 16(2):194-201.

Plotkin, P., and J. Bernardo. 2003. Investigations into the basis of the reproductive behavioral polymorphism in Lepidochelys olivacea. Proceedings of the Twenty-second Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-503. 29 p. https://repository.library.noaa.gov/view/noaa/3348

Plotkin, P. T. 1994. Migratory and reproductive behavior of the olive ridley turtle, *Lepidochelys olivacea* (Eschscholtz, 1829), in the eastern Pacific Ocean. Ph.D. Dissertation. Texas A&M University, College Station, Texas.

Plotkin, P. T. 2010. Nomadic behaviour of the highly migratory olive ridley sea turtle *Lepidochelys olivacea* in the eastern tropical Pacific Ocean. Endangered Species Research. 13(1):33-40.

Plotkin PT, Byles RA, and Owens DW. 1994. Migratory and reproductive behavior of *Lepidochelys olivacea* in the eastern Pacific Ocean. Page 138 in B.A. Schroeder and B. E. Witherington (compilers), Proceedings of the Thirteenth Annual Symposium on Sea Turtle Biology and Conservation.; 1994; U.S. Dep. of Comm., NOAA Tech. Memo. NMFS-SEFSC-341. 281 p. https://repository.library.noaa.gov/view/noaa/6160

Plotkin, P. T., R. A. Byles, D. C. Rostal, and D. W. Owens. 1995. Independent versus socially facilitated oceanic migrations of the olive ridley, *Lepidochelys olivacea*. Marine Biology. 122:137-143.

Plotkin, P. T., D. W. Owens, R. A. Byles, and R. Patterson. 1996. Departure of Male Olive Ridley Turtles (*Lepidochelys olivacea*) from a Nearshore Breeding Ground. Herpetologica. 52(1):1-7.

Plotkin, P. T., (Ed). 1995. National Marine Fisheries Service and the U.S. Fish and Wildlife Service Status Reviews for Sea Turtles Listed under the Endangered Species Act of 1973. National Marine Fisheries Service, Silver Spring, Maryland.

Poeta, G., E. Staffieri, A. T. Acosta, and C. Battisti. 2017. Ecological effects of anthropogenic litter on marine mammals: A global review with a “black-list” of impacted taxa. Hystrix the Italian Journal of Mammalogy. 28(2):253–264.

Poloczanska, E. S., C. J. Limpus, and G. C. Hays. 2009. Vulnerability of marine turtles to climate change. In: Sims DW, editor. Advances in Marine Biology. 56:151-211.

Polovina, J. J., G. H. Balazs, E. A. Howell, D. M. Parker, M. P. Seki, and P. H. Dutton. 2004. Forage and migration habitat of loggerhead (*Caretta caretta*) and olive ridley (*Lepidochelys olivacea*) sea turtles in the central North Pacific Ocean. Fisheries Oceanography. 13(1):36-51.

Polovina, J. J., E. Howell, D. M. Parker, and G.H. Balazs. 2003. Dive-depth distribution of loggerhead (*Caretta caretta*) and olive ridley (*Lepidochelys olivacea*) sea turtles in the central North Pacific: might deep longline sets catch fewer turtles? Fishery Bulletin. 101(1):189-193.

Portnoy, D. S., J. R. McDowell, E. J. Heist, J. A. Musick, and J. E. Graves. 2010. World phylogeography and male-mediated gene flow in the sandbar shark, Carcharhinus plumbeus. Molecular Ecology. 19(10):1994-2010.

Pritchard, P. C. H. 1969. Studies of the systematics and reproductive cycles of the genus Lepidochelys. University of Florida.

Pritchard, P. C. H., and P. T. Plotkin. 1995. Olive ridley sea turtle, Lepidochelys olivacea. In: Plotkin PT, editor. National Marine Fisheries Service and US Fish and Wildlife Service status reviews for sea turtles listed under the Endangered Species Act of 1973. National Marine Fisheries Service, Silver Spring, Maryland.

Pultz, S., D. O. O'Daniel, S. C. Krueger, H. E. McSharry, and G. H. Balazs. 1999. Marine turtle survey on Tinian, Mariana Islands. Micronesia. 32(1):85-94.

Purves, M. G., D. J. Agnew, E. Balguerias, C. A. Moreno, and B. Watkins. 2004. Killer whale (*Orcinus orca*) and sperm whale (*Physeter macrocephalus*) interactions with longline vessels in the Patagonian toothfish fishery at South Georgia, South Atlantic. CCAMLR Science. 11(111-126).

Quinn, T. J., and H. J. Niebauer. 1995. Relation of eastern Bering Sea walleye pollock (*Theragra chalcogramma*) recruitment to environmental and oceanographic variables. Canadian Special Publication of Fisheries and Aquatic Sciences.497-507.

Rambahiniarison, J. M., M. J. Lamoste, C. A. Rohner, R. Murray, S. Snow, J. Labaja, G. Araujo, and A. Ponzo. 2018. Life History, Growth, and Reproductive Biology of Four Mobulid Species in the Bohol Sea, Philippines. Frontiers in Marine Science. 5.

Ramos-Cartelle, A., Garcia-Cortes, B., Ortíz de Urbina, J., Fernandez-Costa, J., Gonzalez-Gonzalez, I. and Mejuto, J. (2012) Standardized catch rates of the oceanic whitetip shark (*Carcharhinus longimanus*) from observations of the Spanish longline fishery targeting swordfish in the Indian Ocean during the 1998-2011 period. IOTC-2012-WPEB08-27. 15pp.

Raven, J., K. Caldeira, H. Elderfield, O. Hoegh-Guldberg, P. Liss, U. Riebesell, J. Shepherd, C. Turley, and A. Watson. 2005. Ocean acidification due to increasing atmospheric carbon dioxide. The Royal Society.

Reed, D. H. 2004. Extinction risk in fragmented habitats. Animal Conservation. 7(2):181-191.

Reeves, R. R., and H. Whitehead. 1997. Status of the sperm whale, *Physeter macrocephalus*, in Canada. Canadian field-naturalist Ottawa ON. 111(2):293-307.

Reeves, R. R., S. Leatherwood, and R. W. Baird. 2009. Evidence of a Possible Decline since 1989 in False Killer Whales (*Pseudorca crassidens*) around the Main Hawaiian Islands. Pacific Science. 63(2):253-261.

Reid, D. D., and M. Krogh. 1992. Assessment of catches from protective shark meshing off New South Wales beaches between 1950 and 1990. Australian Journal of Marine and Freshwater Research. 43:283-296.

Reina, R. D., J. R. Spotila, F. V. Paladino, and A. E. Dunham. 2008. Changed reproductive schedule of eastern Pacific leatherback turtles *Dermochelys coriacea* following the 1997–98 El Niño to La Niña transition. Endangered Species Research. 7:155-161.

Rice, J., F. Carvalho, M. Fitchett, S. Harley, and A. Ishizaki. 2021. Future Stock Projections of Oceanic Whitetip Sharks in the Western and Central Pacific Ocean. Western and Central Pacific Fisheries Commission Scientific Committee 17th Regular Session WCPFC-SC17-2021/SA-IP-21.

Rice, D. W. 1974. Whales and whale research in the eastern North Pacific. The whale problem: A status report Harvard Press, Cambridge, MA. p. 170-195.

Rice, D. W. 1989. Sperm whale *Physeter macrocephalus Linnaeus*, 1758. H. In: Ridgway, S., R. Harrison, editors. Handbook of marine mammals, vol. 4. p. 177-233. London: Academic Press.

Rice, J., F. Carvalho, M. Fitchett, S. Harley, and A. Ishizaki. 2020. Future Projections of Oceanic Whitetip Sharks in the Western and Central Pacific Ocean. Western Pacific Regional Fishery Management Council 137th Science and Statiscal Meeting. 9 September 2020. Honolulu. 23 p.

Rice, J., and S. Harley. 2012. Stock assessment of sillky sharks in the western and central Pacific Ocean. Paper presented at: 8th Regular Session of the Scientific Committee of the WCPFC. Busan, Republic of Korea.

Rice, J. S., Tremblay-Boyer, L., Scott, R., Hare, S., and A. Tidd. 2015. Analysis of stock status and related indicators for key shark species of the Western Central Pacific Fisheries Commission. Paper presented at: 11th Regular Session of the Scientific Committee of the WCPFC. Pohnpei, Federated States of Micronesia.

Rice, M.R., and G.H. Balazs. 2008. Diving behavior of the Hawaiian green turtle (*Chelonia mydas*) during oceanic migrations. Journal of Experimental Marine Biology and Ecology. 356(1-2):121-127.

Richards, P. M., S. P. Epperly, S. S. Heppell, R. T. King, C. R. Sasso, F. Moncada, G. Nodarse, D. J. Shaver, Y. Medina, and J. Zurita. 2011. Sea turtle population estimates incorporating uncertainty: a new approach applied to western North Atlantic loggerheads *Caretta caretta*. Endangered Species Research. 15(2):151-158.

Ridley, M. 2003. Evolution. 3rd edition. Blackwell Pub., Oxford. 800 pages. Available at: https://www.blackwellpublishing.com/ridley/

Robinson, R. A., H. Q. P. Crick, J. A. Learmonth, I. M. D. Maclean, C. D. Thomas, F. Bairlein, M. C. Forchhammer, C. M. Francis, J. A. Gill, B. J. Godley *et al.* 2008. Travelling through a warming world: climate change and migratory species. Endangered Species Research. 7:87-99.

Rochman, C. M., T. Kurobe, I. Flores, and S. J. Teh. 2014. Early warning signs of endocrine disruption in adult fish from the ingestion of polyethylene with and without sorbed chemical pollutants from the marine environment. Science of the Total Environment. 493:656-661.

Romo-Curiel, A. E., Z. Ramirez-Mendoza, A. Fajardo-Yamamoto, M. R. Ramirez-Leon, M. C. Garcia-Aguilar, S. Z. Herzka, P. Perez-Brunius, L. E. Saldana-Ruiz, J. Sheinbaum, K. Kotzakoulakis et al. 2022. Assessing the exposure risk of large pelagic fish to oil spills scenarios in the deep waters of the Gulf of Mexico. Marine Pollution Bulletin. 176:113434.

Ross, M. A. 1984. A Quantitative Study of the Stony Coral Fishery in Cebu, Philippines. Marine Ecology-Pubblicazioni Della Stazione Zoologica Di Napoli I. 5(1):75-91.

Roe, J.H., Morreale, S.J., Paladino, F.V., Shillinger, G.L., Benson, S.R., Eckert, S.A., Bailey, H., Tomillo, P.S., Bograd, S.J., Eguchi, T. *et al*. 2014. Predicting bycatch hotspots for endangered leatherback turtles on longlines in the Pacific Ocean. Proceedings of the Roal Society B: Biological Sciences. 281(1777):20132559.

Rohner, C. A., S. J. Pierce, A. D. Marshall, S. J. Weeks, M. B. Bennett, and A. J. Richardson. 2013. Trends in sightings and environmental influences on a coastal aggregation of manta rays and whale sharks. Marine Ecology Progress Series. 482:153-168.

Ruck, C. L. 2016. Global genetic connectivity and diversity in a shark of high conservation concern, the oceanic whitetip, *Carcharhinus longimanus* [Master of Science]. Nova Southeastern University. p. 64.

Ruiz, G.A. 1994. Sea turtle nesting population at Playa La Flor, Nicaragua: an olive ridley "arribada" beach. Pages 129-130 in Bjorndal, K.A., A.B. Bolten, D.A. Johnson, and P.J. Eliazar (compilers). Proceedings of the Fourteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-351. https://repository.library.noaa.gov/view/noaa/6185

Ruiz-Cooley, R.I., P.L. Koch, P.C. Fiedler, M.D. McCarthy. 2014. Carbon and nitrogen isotopes from top predatory amino acids reveal rapidly shifting ocean biochemistry in the outer California current. PLoS ONE 9(10): e110355. 8 p.

Ryder, C. E., T. A. Conant, and B. A. Schroeder. 2006. Report of the Workshop on Marine Turtle Longline Post-Interaction Mortality. U.S. Dep. Commerce, NOAA Technical Memorandum NMFS-F/OPR-29. 36 p. https://repository.library.noaa.gov/view/noaa/23893

Sadoyama, A., N. Kamezaki, and I. Miyawaki. 1996. Recapture in Vietnam of the loggerhead turtle, nested in the Miyakojima Island, Okinawa Archipelago. Umigame Newsletter. 29:9.

Salazar, C. P., J. V. Perez, E. A. Padilla, and R. M. Millan. Twenty five years nesting of olive ridley sea turtle *Lepidochelys olivacea* in Escobilla Beach, Oaxaca Mexico, pages 27-29 in Abreu-Grobois, F.A., R. Briseño-Dueñas, R. Márquez, and L. Sarti, compilers. 2000. Proceedings of the Eighteenth International Sea Turtle Symposium. U.S. Dep. Commer.

NOAA Tech. Memo. NMFS-SEFSC-436, 293 p. https://repository.library.noaa.gov/view/noaa/30887

Sales, G., B. B. Giffoni, F. N. Fiedler, V. G. Azevedo, J. E. Kotas, Y. Swimmer, and L. Bugoni. 2010. Circle hook effectiveness for the mitigation of sea turtle bycatch and capture of target species in a Brazilian pelagic longline fishery. Aquatic Conservation-Marine and Freshwater Ecosystems. 20(4):428-436.

Sanchez, S. 2016. Improved Surveillance to Protect Ecuador's Manta Populations. https://wildaid.org/improved-surveillance-to-protect-ecuadors-manta-populations/

Santana, F. M., P. J. Duarte-Neto, and R. P. Lessa. 2004. *Carcharhinus longimanus*. In: Lessa RP, Nobrega MF, Bezerra Jr. JL, editors. Dinamica de Populacoes e Avaliacao de Estoques dos Recursos Pesqueiros da Região Nordeste. Vol II. Universidade Federal Rural de Pernambuco Deoartanebti de Pesca. Laboratório de Dinâmica de Populacoes Marinhas - DIMAR.

Santidrian-Tomillo, P., F. V. Paladino, J. S. Suss, and J. R. Spotila. 2010. Predation of leatherback turtle hatchlings during the crawl to the water. Chelonian Conservation and Biology. 9(1):18-25.

Santidrian-Tomillo, P. S., D. Oro, F. V. Paladino, R. Piedra, A. E. Sieg, and J. R. Spotila. 2014. High beach temperatures increased female-biased primary sex ratios but reduced output of female hatchlings in the leatherback turtle. Biological Conservation. 176:71-79.

Santidrian-Tomillo, P., V. S. Saba, R. Piedra, F. V. Paladino, and J. R. Spotila. 2008. Effects of illegal harvest of eggs on the population decline of leatherback turtles in Las Baulas Marine National Park, Costa Rica. Conservation Biology. 22(5):1216-1224.

Santos, R. G., R. Andrades, M. A. Boldrini, and A. S. Martins. 2015. Debris ingestion by juvenile marine turtles: an underestimated problem. Marine Pollution Bulletin. 93(1-2):37-43.

Sargeant, B. L., and J. Mann. 2009. Developmental evidence for foraging traditions in wild bottlenose dolphins. Animal Behaviour. 78(3):715-721.

Sarti, L., S. A. Eckert, P. H. Dutton, A. Barragan, and N. Garcia. 1999. The current situation of the leatherback population of the Pacific Coast of Mexico and Central America, abundance and distribution of the nestings: an update. 19th Annual Sea Turtle Symposium, South Padre Island, Texas.

Scarff, J. 1986. Historic and present distribution of the right whale (*Eubalaena glacialis*) in the eastern North Pacific south of 50 N and east of 180 W. Reports International Whaling Commission.43-63.

Schuyler, Q. A. 2014. Ingestion of Marine Debris by Sea Turtles. The University of Queensland.

Schuyler, Q. A., C. Wilcox, K. A. Townsend, K. R. Wedemeyer-Strombel, G. Balazs, E. van Sebille, and B. D. Hardesty. 2015. Risk analysis reveals global hotspots for marine debris ingestion by sea turtles. Global Change Biology. 22(2):567-576.

Schwartz, F. J. 1984. A blacknose shark from North Carolina deformed by encircling monofilament line. Florida Scientist.62-64.

Scott, M., E. Cardona, K. Scidmore-Rossing, M. Royer, J. Stahl, and M. J. M. P. Hutchinson. 2022. What's the catch? Examining optimal longline fishing gear configurations to minimize negative impacts on non-target species. 143:105186.

Seki, M. P., J. J. Polovina, D. R. Kobayashi, R. R. Bidigare, and G. T. Mitchum. 2002. An oceanographic characterization of swordfish (*Xiphias gladius*) longline fishing grounds in the springtime subtropical North Pacific. Fisheries Oceanography. 11(5):251-266.

Seki, T., T. Taniuchi, H. Nakano, and M. Shimizu. 1998. Age, Growth and Reproduction of the Oceanic Whitetip Shark from the Pacific Ocean. Fisheries Science. 64(1):14-20.

Seminoff, J. A. 1994. Conservation of the marine turtles of Mexico: a survey of nesting beach conservation projects. University of Arizona, Tucson.

Seminoff, J. A., C. D. Allen, G. H. Balazs, P. H. Dutton, T. Eguchi, H. L. Haas, S. A. Hargrove, M. P. Jensen, D. L. Klemm, A. M. Lauritsen *et al.* 2015. Status Review of the Green Sea Turtle (*Chelonia mydas*) under the Endangered Species Act. NOAA Technical Memorandum, NOAA NMFS-SWFSC-539. 571 p. https://repository.library.noaa.gov/view/noaa/4922

Seminoff, J. A., S. R. Benson, K. E. Arthur, T. Eguchi, P. H. Dutton, R. F. Tapilatu, and B. N. Popp. 2012. Stable isotope tracking of endangered sea turtles: validation with satellite telemetry and δ15N analysis of amino acids. PLoS One. 7(5):e37403.

Seminoff, J. A., D. T. Crouse, N. J. Pilcher, K. L. Eckert, M. A. Marcovaldi, J. A. Mortimer, and W. J. Nichols. 2004. 2004 global status assessment - Green turtle (*Chelonia mydas*). IUCN.71.

Seminoff, J. A., T. Eguchi, J. Carretta, C. D. Allen, D. Prosperi, R. Rangel, J. W. Gilpatrick, K. Forney, and S. H. Peckham. 2014. Loggerhead sea turtle abundance at a foraging hotspot in the eastern Pacific Ocean: implications for at-sea conservation. Endangered Species Research. 24(3):207-220.

Seminoff, J. A., T. T. Jones, A. Resendiz, W. J. Nichols, and M. Y. Chaloupka. 2003. Monitoring green turtles (*Chelonia mydas*) at a coastal foraging area in Baja California, Mexico: multiple indices describe population status. Journal of the Marine Biological Association of the United Kingdom. 83(6):1355-1362.

Seminoff, J. A., A. Resendiz, and W. J. Nichols. 2002a. Diet of East Pacific Green Turtles (*Chelonia mydas*) in the Central Gulf of California, México. Journal of Herpetology. 36:447-453.

Seminoff, J. A., A. Resendiz, and W. J. Nichols. 2002b. Home range of green turtles *Chelonia mydas* at a coastal foraging area in the Gulf of California, Mexico. Marine Ecology Progress Series. 242:253-265.

Senko, J., A. J. Schneller, J. Solis, F. Ollervides, and W. J. Nichols. 2011. People helping turtles, turtles helping people: Understanding resident attitudes towards sea turtle conservation and opportunities for enhanced community participation in Bahia Magdalena, Mexico. Ocean & Coastal Management. 54(2):148-157.

Setala, O., V. Fleming-Lehtinen, and M. Lehtiniemi. 2014. Ingestion and transfer of microplastics in the planktonic food web. Environmental Pollution. 185:77-83.

Shallenberger, E. W. 1981. The status of Hawaiian cetaceans. Final Report to U.S . Marine Mammal Commission in Fulfillment of Contract MH7AC028. 27 p.

Shanker, K., and B. Mohanty. 1999. Guest Editorial: Operation Kachhapa: In search of a solution for the olive ridleys of Orissa. Marine Turtle Newsletter. 86:1-3.

Shanker, K., B. Pandav, and B. C. Choudhury. 2003. An assessment of the olive ridley turtle (*Lepidochelys olivacea*) nesting population in Orissa, India. Biological Conservation. 115(1):149-160.

Shanker, K., J. Ramadevi, B. C. Choudhury, L. Singh, and R. K. Aggarwal. 2004. Phylogeography of olive ridley turtles (*Lepidochelys olivacea*) on the east coast of India: implications for conservation theory. Molecular Ecology. 13(7):1899-1909.

Shelden, K. E., S. E. Moore, J. M. Waite, P. R. Wade, and D. J. Rugh. 2005. Historic and current habitat use by North Pacific right whales *Eubalaena japonica* in the Bering Sea and Gulf of Alaska. Mammal Review. 35(2):129-155.

Shaver, D. J., C. Gredzens, J. S. Walker, C. A. Godard-Codding, J. E. Yacabucci, A. Frey, P. H. Dutton, and C. J. Schmitt. 2021. Embryo deformities and nesting trends in Kemp's ridley sea turtles *Lepidochelys kempii* before and after the Deepwater Horizon oil spill. Endangered Species Research. 44:277–289.

Shillinger, G. L., D. M. Palacios, H. Bailey, S. J. Bograd, A. M. Swithenbank, P. Gaspar, B. P. Wallace, J. R. Spotila, F. V. Paladino, R. Piedra *et al.* 2008. Persistent leatherback turtle migrations present opportunities for conservation. PLoS Biol. 6(7):e171.

Shillinger, G. L., A. M. Swithenbank, S. J. Bograd, H. Bailey, M. R. Castelton, B. P. Wallace, J. R. Spotila, F. V. Paladino, R. Piedra, and B. A. Block. 2011. Vertical and horizontal habitat preferences of post-nesting leatherback turtles in the South Pacific Ocean. Marine Ecology Progress Series. 422:275-289.

Shillinger, G. L., A. M. Swithenbank, H. Bailey, S. J. Bograd, M. R. Castelton, B. P. Wallace, J. R. Spotila, F. V. Paladino, R. Piedra, and B. A. Block. 2010. Identification of high-use internesting habitats for eastern Pacific leatherback turtles: role of the environment and implications for conservation. Endangered Species Research. 10:215-232.

Siders, Z. A., Martin, S. L., Ahrens, R. N., Littnan, C., and T. T. Jones. 2023. Update to NOAA Technical Memorandum NMFS-PIFSC-101: Incorporating Uncertainty in Maturation and Latest Fishery Takes. PIFSC Internal Report IR-23-03. 29 p.

Silber, G. K., M. D. Lettrich, Thomas PO, Baker JD, Baumgartner M, Becker EA, Boveng P, Dick DM, Fiechter J, Forcada J *et al.* 2017. Projecting Marine Mammal Distribution in a Changing Climate. Frontiers in Marine Science. 4:413.

Silva, I. 2013. Short Note: Presence and Distribution of Hawaiian False Killer Whales (*Pseudorca crassidens*) in Maui County Waters: A Historical Perspective. Aquatic Mammals. 39(4):409-414.

Silverman, B. W. 1986. Density estimation for statistics and data analysis. Chapman and Hall, London.

Simmonds, M. P., and W. J. Eliott. 2009. Climate change and cetaceans: concerns and recent developments. Journal of the Marine Biological Association of the United Kingdom. 89(01).

Skomal, G.B. and J. W. Mandelman. 2012. The physiological response to anthropogenic stressors in marine elasmobranch fishes: a review with a focus on the secondary response. Comparative Biochemistry and Physiology Part A: Molecular & Integrative Physiology 162(2), pp.146-155.

Smallegange, I. M., I. B. van der Ouderaa, and Y. Tibirica. 2016. Effects of yearling, juvenile and adult survival on reef manta ray (*Manta alfredi*) demography. PeerJ. 4:e2370.

Smith, S. E., D. W. Au, and C. Show. 1998. Intrinsic rebound potentials of 26 species of Pacific sharks. Marine and Freshwater Research. 49(7):663-678.

Smith, S. E., D. W. Au, and C. Show C. 2008. Intrinsic rates of increase in pelagic elasmobranchs. Sharks of the open ocean: biology, fisheries conservation. p. 288-297.

Smith, S. C., and H. Whitehead. 1993. Variations in the feeding success and behaviour of Galapagos sperm whales (*Physeter macrocephalus*) as they relate to oceanographic conditions. Canadian Journal of Zoology 71:1991-1996.

Spotila, J. R. 2004. Sea Turtles. John Hopkins University Press, Baltimore, MD.

Sreelekshmi, S., Sukumaran, S., Kishor, T.G., Sebastian W., and A. Gopalakrishnan. 2020. Population genetic structure of the oceanic whitetip shark, *Carcharhinus longimanus*, along the Indian coast. Marine Biodiversity 50:78

Stacy, B. 2012. Summary of findings for sea turtles documented by directed captures, stranding response, and incidental captures under response operations during the BP DWH MC252 oil spill. DWH Sea Turtles NRDA Technical Working Group Report. National Oceanic Atmospheric Administration Assessment Restoration Division, Seattle, WA, USA.

Stacy, N. I., C. L. Field, L. Staggs, R. A. MacLean, B. A. Stacy, J. Keene, D. Cacela, C. Pelton, C. Cray, M. Kelley *et al.* 2017. Clinicopathological findings in sea turtles assessed during the Deepwater Horizon oil spill response. Endangered Species Research. 33:25-37.

Stacey, P., S. Leatherwood, and R. W. Baird. 1994. *Pseudorca crassidens*. Mammalian Species. 456:1-6.

State of Hawaii Department of Business, E. D. a. T. R. E. A. D. 2019. Hawaii Economic Issues Data Report 2019 – State of Hawaii Energy Data and Trends. April 2019. Available at: http://files.hawaii.gov/dbedt/economic/data_reports/reports-studies/energy-datatrend-2019.pdf.

Stevens, G., D. Fernando, and G. N. Di Sciara. 2018. Guide to the Manta and Devil Rays of the World. Princeton University Press.

Stevens, J. D., R. Bonfil, N. K. Dulvy, and P. A. Walker. 2000. The effects of fishing on sharks, rays, and chimaeras (chondrichthyans), and the implications for marine ecosystems. ICES Journal of Marine Science. 57(3):476-494.

Stevens, J. D., and J. M. Lyle. 1989. Biology of Three Hammerhead Sharks (*Eusphyra blochii*, *Sphyrna mokarran* and *S. lewini*) from Northern Australia. Australian Journal of Marine and Freshwater Research. 40:129-146.

Stewart, J. D., C. S. Beale, D. Fernando, A. B. Sianipar, R. S. Burton, B. X. Semmens, and O. Aburto-Oropeza. 2016a. Spatial ecology and conservation of *Manta birostris* in the Indo-Pacific. Biological Conservation. 200:178-183.

Stewart, J. D., E. M. Hoyos-Padilla, K. R. Kumli, and R. D. Rubin. 2016b. Deep-water feeding and behavioral plasticity in *Manta birostris* revealed by archival tags and submersible observations. Zoology. 119(5):406-413.

Stewart, J. D., M. Nuttall, E. L. Hickerson, and M. A. Johnston. 2018. Important juvenile manta ray habitat at Flower Garden Banks National Marine Sanctuary in the northwestern Gulf of Mexico. Marine Biology. 165(7).

Strasburg, D. W. 1958. Distribution, abundance, and habits of pelagic sharks in the central Pacific Ocean. Fisheries. 1:2S.

Stokes, L., and Bergmann, C. (editors). 2019. Careful release protocols for sea turtle release with minimal injury. NOAA Technical Memorandum NMFS-SEFSC-735, 74 pp. https://doi.org/10.25923/mr6j-e5

Suarez, M. M., and C. H. Starbird. 1995. A traditional fishery of leatherback turtles in Maluku, Indonesia. Marine Turtle Newsletter. 68:16-18.

Suarez, A, and C. H. Starbird. 1996. Subsistence hunting of leatherback turtle, *Dermochelys coriacea*, in the Kai Islands, Indonesia. Chelonian Conservation and Biology. 2(2):190.

Suganuma, H. 1985. Green turtle research program in Ogasawara. Marine Turtle Newsletter. 33:2-3.

Suganuma, H, S. Tanaka, and E. Inoguchi. 2012. Conservation of sea turtles and management of the monitoring system by local people for leatherback turtle, *Dermochelys coriacea*, in Manokwari region, West Papua, Indonesia. Everlasting Nature of Asia. p. 6.

Swimmer, Y., and E. Gilman. 2012. Report of the Sea Turtle Longline Fishery Post-release Mortality Workshop, November 15–16, 2011. U.S. Dep. Commer., NOAA Tech. Memo., NOAA-TM-NMFS-PIFSC-34, 31 p. https://repository.library.noaa.gov/view/noaa/4216

Swimmer, Y., A. Gutierrez, K. Bigelow, C. Barceló, B. Schroeder, K. Keene, K. Shattenkirk, and D. G. Foster. 2017. Sea Turtle Bycatch Mitigation in U.S. Longline Fisheries. Frontiers in Marine Science. 4.

Tapilatu, R. F. 2014. The conservation of the western Pacific leatherback sea turtle (*Dermochelys coriacea*) at Bird's Head peninsula, Papua Barat, Indonesia [Doctor of Philosophy]. University of Alabama. p. 1-227.

Tapilatu, R. F., P. H. Dutton, M. Tiwari, T. Wibbels, H. V. Ferdinandus, W. G. Iwanggin, and B. H. Nugroho. 2013. Long-term decline of the western Pacific leatherback, *Dermochelys coriacea*: a globally important sea turtle population. Ecosphere. 4(2):1-15.

Tapilatu, R. F., H. Wona, and P. P. Batubara. 2017. Status of sea turtle populations and its conservation at Bird's Head Seascape, Western Papua, Indonesia. Biodiversitas, Journal of Biological Diversity. 18(1):129-136.

Tapilatu, R. F., and M. Tiwari. 2007. Leatherback turtle, *Dermochelys coriacea*, hatching success at Jamursba-Medi and Wermon beaches in Papua, Indonesia. Chelonian Conservation and Biology. 6(1):154-158.

Tesoro Oil Spill Natural Resource Trustees. 2000. Final Restoration Plan and Environmental Assessment for the August 24, 1998 Tesoro Hawaii Oil Spill (Oahu and Kauai, Hawaii). Prepared by: National Oceanic and Atmospheric Administration, U.S. Department of the Interior, and State of Hawaii. 90 pp.

The Hawaii Association for Marine Education and Research Inc. 2005. Manta rays. http://www.hamerinhawaii.org

Thompson, P. O., and W. A. Friedl. 1982. A long term study of low frequency sounds from several species of whales off Oahu, Hawaii. Cetology. 45:1-19.

Tiwari, M., G. H. Balazs, and S. Hargrove S. 2010. Estimating carrying capacity at the green turtle nesting beach of East Island, French Frigate Shoals. Marine Ecology Progress Series. 419:289-294.

Tiwari, M., B. P. Wallace, and M. Girondot. 2013. *Dermochelys coriacea* (West Pacific Ocean subpopulation), Leatherback. The IUCN Red List of Threatened Species 2013: e.T46967817A46967821. http://dx.doi.org/10.2305/IUCN.UK.2013-2.RLTS.T46967817A46967821.en

Tixier, P., N. Gasco, G. Duhamel, M. Viviant, M. Authier, and C. Guinet. 2010. Interactions of Patagonian toothfish fisheries with killer and sperm whales in the Crozet Islands exclusive economic zone: an assessment of depredation levels and insights on possible mitigation strategies. CCAMLR Science. 17:179-195.

Tolotti, M., R. Bauer, F. Forget, P. Bach, L. Dagorn, and P. Travassos. 2017. Fine-scale vertical movements of oceanic whitetip sharks (*Carcharhinus longimanus*). Fishery Bulletin. 115(3):380-395.

Tolotti, M. T., P. Bach, F. Hazin, P. Travassos, and L. Dagorn. 2015. Vulnerability of the Oceanic Whitetip Shark to Pelagic Longline Fisheries. PLoS One. 10(10):e0141396.

Tolotti, M. T., P. Travassos, F. L. Fredou, C. Wor, H. A. Andrade, and F. Hazin. 2013. Size, distribution and catch rates of the oceanic whitetip shark caught by the Brazilian tuna longline fleet. Fisheries Research. 143:136-142.

Towns, D. R., and C. H. Daugherty. 1994. Patterns of range contractions and extinctions in the New Zealand herpetofauna following human colonisation. New Zealand Journal of Zoology. 21(4):325-339.

Tremblay-Boyer, L., and S. Brouwer. 2016. Western and Central Pacific Fisheries Commission Scientific Committee, editor. Review of available information on non-key shark species including mobulids and fisheries interactions. Twelfth Regular Session. Bali, Indonesia, August 3-11; 2016.

Tremblay-Boyer, L., F. Carvalho, P. Neubauer, and G. Pilling. 2019. Stock assessment for oceanic whitetip shark in the Western and Central Pacific Ocean. Scientific Committee Fifteenth Regular Session. Pohnpei, Federated States of Micronesia. WCPFC-SC15-2019/SA-WP-06. 99 p.

Tripathy, B., and B. Pandav. 2007. Beach Fidelity and Internesting Movements of Olive Ridley Turtles (*Lepidochelys Olivacea*) at Rushikulya, India. Herpetological Conservation and Biology. 3(1):40-45.

Tuato'o-Bartley, N., T. E. Morrell, and P. Craig. 1993. Status of sea turtles in American Samoa in 1991. Pacific Science. 47(1):215-221.

Tucker, A.D., and N.B. Frazer. 1991. Reproductive variation in leatherback turtles, *Dermochelys coriacea*, at Culebra National Wildlife Refuge, Puerto Rico. Herpetologica 47: 155-124.

Turner-Tomaszewicz, C. N., J. A. Seminoff, L. Avens, L. R. Goshe, S. H. Peckham, J. M. Rguez-Baron, K. Bickerman, and C. M. Kurle. 2015. Age and residency duration of loggerhead turtles at a North Pacific bycatch hotspot using skeletochronology. Biological Conservation. 186:134-142.

Udvardy, M. D. F. 1969. The concept of faunal dynamism and the analysis of an example. Bonner Zoologische Beitrage. 20:1-10.

University of Hawaii News. 2023. Dead sperm whale on Kaua‘i had nets, plastic bags, float in stomach. Accessed on: 5/10/2023. Available at: https://www.hawaii.edu/news/2023/02/02/dead-sperm-whale-kauai/

Ulrich, G. F., C. M. Jones, W. B. Driggers, J. M. Drymon, D. Oakley, and C. Riley. 2007. Habitat utilization, relative abundance, and seasonality of sharks in the estuarine and nearshore waters of South Carolina. American Fisheries Society Symposium. 50:125-139.

U.S. Navy. 2017. Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase III). SSC Pacific Technical Report. p. 194.

U.S. Department of Health and Human Services. 1995. Toxicological profile for polycyclic aromatic hydrocarbons (PAHs). US Agency for Toxic Substances Disease Registry, Atlanta, GA. 487 p.

U.S. Energy Information Administration (EIA). 2019. U.S. EIA, Petroleum & Other Liquids, Imports by Area of Entry. U.S.EIA, Editor: Washington, DC. Available from: https://www.eia.gov/petroleum/data.php.

U.S. Environmental Protection Agency (EPA). 2022. EPA Biological Evaluation for the Hawaii Dispersant Preauthorization Plan (July 2019). 256 p.

EPA. 2010. Climate Change Indicators in the United States. 80 p.

USFWS and NMFS. 1998. Endangered Species Consultation Handbook. Procedures for Conducting Consultation and Conference Activities under Section 7 of the Endangered Species Act. p. 315.

Vanderlaan, A. S. M., J. J. Corbett, S. L. Green, J. A. Callahan, C. Wang, R. D. Kenney, C. T. Taggart, and J. Firestone. 2009. Probability and mitigation of vessel encounters with North Atlantic right whales. Endangered Species Research. 6:273-285.

Valverde, R. A., C. M. Orrego, M. T. Tordoir, F. M. Gomez, D. S. Solis, R. A. Hernandez, G. B. Gomez, L. S. Brenes, J. P. Baltodano, L. G. Fonseca *et al.* 2012. Olive Ridley Mass Nesting Ecology and Egg Harvest at Ostional Beach, Costa Rica. Chelonian Conservation and Biology. 11(1):1-11.

Van Buskirk, J., and L. B. Crowder. 1994. Life-history variation in marine turtles. Copeia. 66-81.

Van Houtan, K. S. 2011. Assessing the impact of fishery actions to marine turtle populations in the North Pacific using classical and climate-based models. NMFS. 25 p.

Van Houtan, K. S., and J. M. Halley. 2011. Long-term climate forcing in loggerhead sea turtle nesting. PLoS One. 6(4):e19043.

Van Houtan, K. S., J. N. Kittinger, A. L. Lawrence, C. Yoshinaga, V. R. Born, and A. Fox. 2012. Hawksbill Sea Turtles in the Northwestern Hawaiian Islands. Chelonian Conservation and Biology. 11(1):117-121.

Von Essen, E., H. P. Hansen, H. Nordström Källström, M. N. Peterson, and T. R. Peterson. 2014. Deconstructing the poaching phenomenon: A review of typologies for understanding illegal hunting. British Journal of Criminology 54(4):632-651.

Vose, F. E., and B. V. Shank. 2003. Predation on loggerhead and leatherback post-hatchlings by gray snapper. Marine Turtle Newsletter. 99:11-14.

Wade, P. R., and T. Gerrodette. 1993. Estimates of Cetacean Abundance and Distribution in the Eastern Tropical Pacific. Report of the International Whaling Commission. 477-493 p.

Wade, P., M. P. Heide-Jorgensen, K. Shelden, J. Barlow, J. Carretta, J. Durban, R. LeDuc, L. Munger, S. Rankin, and A. Sauter. 2006. Acoustic detection and satellite-tracking leads to discovery of rare concentration of endangered North Pacific right whales. Biology Letters. 2(3):417-419.

Wade, P., A. De Robertis, K. Hough, R. Booth, A. Kennedy, R. LeDuc, L. Munger, J. Napp, K. Shelden, and S. Rankin. 2011. Rare detections of North Pacific right whales in the Gulf of Alaska, with observations of their potential prey. Endangered Species Research. 13(2):99-109.

Wallace, B. P., A. D. DiMatteo, B. J. Hurley, E. M. Finkbeiner, A. B. Bolten, M. Y. Chaloupka, B. J. Hutchinson, F. A. Abreu-Grobois, D. Amorocho, K. A. Bjorndal *et al.* 2010a.

Regional management units for marine turtles: a novel framework for prioritizing conservation and research across multiple scales. PLoS One. 5(12):e15465.

Wallace, B. P., C. Y. Kot, A. D. DiMatteo, T. Lee, L. B. Crowder, and R. L. Lewison. 2013a. Impacts of fisheries bycatch on marine turtle populations worldwide: toward conservation and research priorities. Ecosphere. 4(3).

Wallace, B. P., R. L. Lewison, S. L. McDonald, R. K. McDonald, C. Y. Kot, S. Kelez, R. K. Bjorkland, E. M. Finkbeiner, S. Helmbrecht, and L. B. Crowder. 2010b. Global patterns of marine turtle bycatch. Conservation Letters. 3(3):131-142.

Wallace, B. P., and V. S. Saba. 2009. Environmental and anthropogenic impacts on intra-specific variation in leatherback turtles: opportunities for targeted research and conservation. Endangered Species Research. 7(1):11-21.

Wallace, B. P., M. Tiwari, and M. Girondot. 2013b. *Dermochelys coriacea*, Leatherback. The IUCN Red List of Threatened Species 2013: e.T6494A43526147. https://www.iucnredlist.org/species/6494/43526147

Wallace, B. P., B. A. Stacy, E. Cuevas, C. Holyoake, P. H. Lara, A. C. J. Marcondes, J. D. Miller, H. Nijkamp, N. J. Pilcher, I. Robinson et al. 2020. Oil spills and sea turtles: documented effects and considerations for response and assessment efforts. Endangered Species Research. 41:17-37.

Waples, R. S. 2006. A bias correction for estimates of effective population size based on linkage disequilibrium at unlinked gene loci. Conservation Genetics. 7(2):167-184.

Waples, R. S., and C. Do. 2008. LDNE: a program for estimating effective population size from data on linkage disequilibrium. Molecular ecology resources. 8(4):753-756.

Warden, M. L., and K. T. Murray. 2011. Reframing protected species interactions with commercial fishing gear: Moving toward estimating the unobservable. Fisheries Research. 110(3):387-390.

Warden, M. L., H. L. Haas, K. A. Rose, and P. M. Richards. 2015. A spatially explicit population model of simulated fisheries impact on loggerhead sea turtles (*Caretta caretta*) in the Northwest Atlantic Ocean. Ecological modelling. 299:23-39.

Ward-Paige, C.A., B. Davis, and B. Worm. 2013. Global population trends and human use patterns of Manta and Mobula rays. PlosOne 8: e74835.

Watkins, W. A., M. A. Daher, N. A. Dimarzio, A. Samuels, D. Wartzok, K. M. Fristrup, P. W. Howey, and R. R. Maiefski. 2002. Sperm whale dives tracked by radio tag telemetry. Marine Mammal Science. 18(1):55-68.

Watkins, W. A., M. A. Daher, K. M. Fristrup, T. J. Howald, and G. N. J. M. m. s. Di Sciara. 1993. Sperm whales tagged with transponders and tracked underwater by sonar. 9(1):55-67.

Watwood, S. L., P. J. Miller, M. Johnson, P. T. Madsen, and P. L. Tyack. 2006. Deep-diving foraging behaviour of sperm whales (*Physeter macrocephalus*). J Anim Ecol. 75(3):814-825.

WCPFC (Western and Central Pacific Fisheries Commission). 2013. Annual Report to the Commission. Part 1: Information on fisheries, research and statistics. Republic of the Marshall Islands. WCPFC-SC9-AR/CCM-12. 15 p.

WCPFC. 2014. Annual Report to the Commission. Part 1: Information on fisheries, research and statistics. Republic of the Marshall Islands. WCPFC-SC10-AR/CCM-13. 19 p.

WCPFC. 2015. Annual Report to the Commission. Part 1: Information on fisheries, research and statistics. Republic of the Marshall Islands. WCPFC-SC11-AR/CCM-13. 20 p.

WCPFC. 2016. Annual Report to the Commission. Part 1: Information on fisheries, research and statistics for 2015. Fiji. WCPFC-SC12-AR/CCM-07. 15 p.

WCPFC. 2017. Annual Report to the Commission. Part 1: Information on fisheries, research and statistics. Republic of the Marshall Islands. WCPFC-SC13-AR/CCM-13 Rev. 1. 2017 p.

WCPFC. 2018a. Conservation and Management Measure of Sea Turtles. Conservation and Management Measure 2018-04. Fifteenth Regular Session. Honolulu, Hawaii, USA.

WCPFC. 2018b. Annual Report to the Commission. Part 1: Information on fisheries, research and statistics for 2017. Fiji. WCPFC-SC14-AR/CCM-07. 17 p.

WCPFC. 2021a. Science and Scientific Data Functions. Available at: https://www.wcpfc.int/node/29966 Accessed last on: February 6, 2023.

WCPFC. 2021b. Longline hammerhead shark summary. 1 p. Available at: https://www.wcpfc.int/doc/ll-hammerhead-shark-summary Accessed last on February 6, 2023.

WCPFC. 2021c. An update on available data on cetacean interactions in the WCPFC longline and purse seine fisheries. Scientific Committee Seventeenth Regular Session, 11 –19 August 2021. WCPFC-SC17-2021/ST IP-10. 32 p.

Weaver, M. L. 1996a. Broadening environmental management in Fiji. Pacific Conservation Biology. 2:315-320.

Weaver, S. 1996b. A strategy to guide the next three years of research and management of sea turtles in Fiji. New Zealand. p. 54.

Wegner, N. C., and D. P. Cartamil. 2012. Effects of prolonged entanglement in discarded fishing gear with substantive biofouling on the health and behavior of an adult shortfin mako shark, *Isurus oxyrinchus*. Marine Pollution Bulletin. 64(2):391-394.

Wells, J. V., and M. E. Richmond. 1995. Populations, metapopulations, and species populations: what are they. Wildlife Society Bulletin. 23(3):458-462.

White, E. R., M. C. Myers, J. M. Flemming, and J. K. Baum. 2015. Shifting elasmobranch community assemblage at Cocos Island—an isolated marine protected area. Conservation Biology. 29(4):1186-1197.

White, F. N. 1994. Swallowing dynamics of sea turtles. P. 89-95. In: Balazs GH, Pooley SG, editors. Research plan to assess marine turtle hooking mortality: results of an expert workshop held in Honolulu, Hawaii, November 16-18, 1993. NOAA-TM-NMFS-SWFSC-201. 172 p. https://repository.library.noaa.gov/view/noaa/6181

White, W. T., C. Bartron, and I. C. Potter. 2008. Catch composition and reproductive biology of *Sphyrna lewini* (Griffith & Smith) (Carcharhiniformes, Sphyrnidae) in Indonesian waters. Journal of fish biology. 72(7):1675-1689.

Whitehead, H. 1995. Status of Pacific sperm whale stocks before modern whaling. Reports International Whaling Commission. 45:407–412.

Whitehead, H. 1997. Sea surface temperature and the abundance of sperm whale calves off the Galapagos Islands: implications for the effects of global warming. Rep. Int. Whal. Commn. 47:941–944.

Whitehead, H. 2002. Estimates of current global population size and historical trajectory for sperm whales. Marine Ecology Progress Series 242: 295-304.

Whitehead, H., R. Antunes, S. Gero, S. N. P. Wong, D. Engelhaupt, and L. Rendell. 2012. Multilevel Societies of Female Sperm Whales (*Physeter macrocephalus*) in the Atlantic and Pacific: Why Are They So Different? International Journal of Primatology. 33(5):1142-1164.

Whiting, S. D., J. L. Long, and M. Coyne. 2007a. Migration routes and foraging behaviour of olive ridley turtles *Lepidochelys olivacea* in northern Australia. Endangered Species Research. 3(1):1-9.

Whiting, S. D., J. L. Long, K. M. Hadden, A. D. Lauder, and A. U. Koch. 2007b. Insights into size, seasonality and biology of a nesting population of the Olive Ridley turtle in northern Australia. Wildlife Research. 34(3):200-210.

Whittow, G. C., and G. H. Balazs. 1982. Basking behavior of the Hawaiian green turtle (*Chelonia mydas*). Pacific Science. 36(2):129-140.

Wibbels, T. 2003. Critical approaches to sex determination in sea turtles. In: Lutz PL, Musick JA, Wyneken J, editors. Biology of Sea Turtles, Volume II. CRC Press, Boca Raton, Florida. p. 103-134.

Wibbels, T. 2007. Sex determination and sex ratios in ridley turtles. In: Plotkin PT, editor. Biology and conservation of ridley sea turtles. Johns Hopkins University Press, Baltimore, MD. p. 167-189.

Wilkinson, C., and D. Souter. 2008. Status of Caribbean coral reefs after bleaching and hurricanes in 2005. Townsville: Global Coral Reef Monitoring Network Reef and Rainforest Research Centre. p. 152.

Williams, S. E., L. P. Shoo, J. L. Isaac, A. A. Hoffmann, and G. J. P. b. Langham. 2008. Towards an integrated framework for assessing the vulnerability of species to climate change. 6(12):e325.

Williamson, J. 2011. Proposed determination: The scalloped hammered – *Sphyrna lewini* as an Endangered Species. Fisheries Scientific Committee, Ref. No. PD50, File No. FSC 10/02. 7 p.

Wilson, S. M., G. D. Raby, N. J. Burnett, S. G. Hinch, and S. J. Cooke. 2014. Looking beyond the mortality of bycatch: sublethal effects of incidental capture on marine animals. Biological Conservation. 171:61-72.

Work, T. M. 2000. Manual de Necropsia de Tortugas Marinas para Biologos en Refugios o Areas Remotas. Hawaii, U.S. Geological Survey National Wildlife Health Center, Hawaii Field Station. p. 25.

Worm, B., H. K. Lotze, I. Jubinville, C. Wilcox, and J. Jambeck. 2017. Plastic as a Persistent Marine Pollutant. Annual Review of Environment and Resources. 42(1):1-26.

WPRFMC. 2009a. Fishery Ecosystem Plan for Pacific Pelagic Fisheries of the Western Pacific Region. Honolulu, HI. p. 251.

WPRFMC. 2009b. Management Modifications for the Hawaii-based Shallow-set Longline Swordfish Fishery that Would Remove Effort Limit, Eliminate the Set Certificate Program, and Implement New Sea Turtle Interaction Caps. Amendment 18 to the Fishery Management Plan for Pelagic Fisheries of the Western Pacific Region, Including a Final Supplemental Environmental Impact Statement, Regulatory Impact Review, and Initial Regulatory Flexibility Analysis. p. 333.

WPRFMC. 2016. Report of the Rare Events Bycatch Workshop Series. Honolulu, HI. p. 45.

WPRFMC. 2018. 2017 Annual Stock Assessment and Fishery Evaluation Report Pacific Island Pelagic Fishery Ecosystem Plan. In: Kingma E, Ishizaki A, Remington T, Spalding S, editors. Western Pacific Regional Fishery Management Council. Honolulu, Hawaii 96813

WPRFMC. 2019. Annual Stock Assessment and Fishery Evaluation Report Pacific Island Pelagic Fishery Ecosystem Plan 2018. Remington, T., Fitchett, M., Ishizaki, A., (Eds.) Western Pacific Regional Fishery Management Council. Honolulu, Hawaii 96813 USA. 375 pp. + Appendices. USA.

WPRFMC. 2020. Annual Stock Assessment and Fishery Evaluation Report Pacific Island Pelagic Fishery Ecosystem Plan 2019. Remington, T., Fitchett, M., Ishizaki, A., DeMello, J. (Eds.) Western Pacific Regional Fishery Management Council. Honolulu, Hawaii 96813 USA. 494 pp. + Appendices.

WPRFMC. 2021. Annual Stock Assessment and Fishery Evaluation Report Pacific Island Pelagic Fishery Ecosystem Plan 2020. Remington, T., Fitchett, M., Ishizaki, A., DeMello, J. (Eds.) Western Pacific Regional Fishery Management Council. Honolulu, Hawaii 96813 USA. 410 pp. + Appendices.

WPRFMC. 2022. Annual Stock Assessment and Fishery Evaluation Report Pacific Island Pelagic Fishery Ecosystem Plan 2021. Remington, T., Fitchett, M., Ishizaki, A., DeMello, J (Eds.) Western Pacific Regional Fishery Management Council, Honolulu, Hawaii. 391 pp. Appendices.

Wuebbles, D. J., D. W. Fahey, and K. Hibbard. 2017. Climate Science Special Report (CSSR). Washington, DC: US Global Change Research Program.

WWF (World Wildlife Fund). 2018. Interim Report of sea turtle conservation activities in Buru and Kei Island Maluku Province. 14 p.

WWF. 2022. Leatherback Sea Turtle Nesting Dynamics in the Maluku Region. 42 p.Xavier, R., A. Barata, L. P. Cortez, N. Queiroz, and E. Cuevas. 2006. Hawksbill turtle (*Eretmochelys imbricata* Linnaeus 1766) and green turtle (*Chelonia mydas Linnaeus* 1754) nesting activity (2002-2004) at El Cuyo beach, Mexico. Amphibia-Reptilia. 27(4):539-547.

Ylitalo, G. M., R. W. Baird, G. K. Yanagida, D. L. Webster, S. J. Chivers, J. L. Bolton, G. S. Schorr, and D. J. McSweeney. 2009. High levels of persistent organic pollutants measured in blubber of island-associated false killer whales (*Pseudorca crassidens*) around the main Hawaiian Islands. Cascadia Research Collective. Olympia, WA. 6 p.

Yokawa, K. and Y. Semba. 2012. Update of the standardized CPUE of oceanic whitetip shark (*Carcharhinus longimanus*) caught by Japanese longline fishery in the Indian Ocean. IOTC–2012–WPEB08–26.

Young, C. N., and J. K. Carlson. 2020. The biology and conservation status of the oceanic whitetip shark (*Carcharhinus longimanus*) and future directions for recovery. Reviews in Fish Biology and Fisheries. 30(2):293-312.

Young, C. N., J. Carlson, M. Hutchinson, C. Hutt, D. Kobayashi, C. T. McCandless, and J. Wraith. 2017. Status review report: oceanic whitetip shark (*Carcharhinius longimanus*). Final Report to the National Marine Fisheries Service, Office of Protected Resources. December 2017. 170 p. https://repository.library.noaa.gov/view/noaa/17097

Zarate, P., K. A. Bjorndal, M. Parra, P. H. Dutton, J. A. Seminoff, and A. B. Bolten. 2013. Hatching and emergence success in green turtle *Chelonia mydas* nests in the Galápagos Islands. Aquatic Biology. 19(3):217-229.

Zarate, P., A. Fernie, and P. H. Dutton. 2003. First results of the East Pacific green turtle, *Chelonia mydas*, nesting population assessment in the Galapagos Islands. In: Seminoff JA, editor. Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-503, Miami, FL. p. 70-73. https://repository.library.noaa.gov/view/noaa/3348

Zarate, P. M., S. S. Cahoon, M. C. D. Contato, P. H. Dutton, and J. A. Seminoff. Frick. 2006. Nesting beach monitoring of green turtles in the Galapagos Islands: a 4-year evaluation. In: Frick, M., A. Panagopoulou, A. Rees, K. Williams, editors. Book of Abstracts: Twenty Sixth Annual Symposium on Sea Turtle Biology and Conservation. Athens, Greece. p. 337: International Sea Turtle Society.

Zug, G. R., M. Chaloupka, and G. H. Balazs. 2006. Age and growth in olive ridley sea turtles (*Lepidochelys olivacea*) from the North-central Pacific: a skeletochronological analysis. Marine Ecology. 27(3):263-270.Zychowski, G. V., and C. A. Godard-Codding. 2017. Reptilian exposure to polycyclic aromatic hydrocarbons and associated effects. Environmental Toxicology and Chemistry. 36(1):25-35.

9 APPENDIX A: EXPOSURE PROFILES

To assess potential exposure of threatened and endangered species in the Action Area, we began by parsing species by the general location of their exposure (coastal or pelagic), whether there were unique temporal characteristics to their potential exposure, for instance would exposure likely occur only when a vessel was transiting to and from harbor (e.g., Hawaiian monk seal). Next we reviewed whether we had data (observations) on the species exposure, or reasoned information that exposure could occur (potential) to one or more of the action's stressors: fishery interactions (e.g., vessel noise, vessel collision, or entanglement in gear); vessel waste, discharge, and emissions. We ended up with four general exposure scenarios to which a species was assigned based on its primary geographic area or occupancy: (1) observed pelagic exposure; (2) observed coastal exposure; (3) potential pelagic exposure; 4) potential coastal exposure.

Each exposure profile that results for each species may represent different combinations of stressors of a different magnitude or exposure to those stressors. Even so, some patterns emerged, specifically when we focused on the primary stressors within each profile. Given the nature of vessel waste, discharge and emissions, these stressors have the potential to affect all exposure profiles. The species listed in the potential coastal exposure would not be exposed to entanglement/entrapment from fishing gear; however, they would likely be exposed to vessel transit and vessel noise.

There are documented observations of interactions with pelagic species and fishing gear resulting in entrapment or entanglement. These data shaped the observed pelagic exposure profile. Although species within this exposure profile may collide with a vessel, the primary stressor from the proposed action is capture in gear (entrapment or entanglement). Lastly, the potential pelagic exposure is comprised of species that could potentially interact with fishing gear or collide with a vessel. However, there are no documented observations of any fishery interactions between these species and the DSLL fishery.

This subsection also summarizes the results of our evaluations of the probability of a response given exposure, which considers susceptibility. Here we briefly describe the species and stressors that are not likely to adversely affect listed species, and our reasoning for this conclusion.

In addition, vessel noise, wastes and discharges, and vessel emissions would have discountable effects on leatherback sea turtles, loggerhead sea turtles, all green sea turtles under consideration, olive ridley sea turtles, sperm whales, MHI IFKW, giant manta rays, oceanic whitetip sharks, and Indo-West Pacific scalloped hammerhead sharks. However, because interactions with longline gear are observed for all of these species (that is, these species fall into observed pelagic exposures category) we discuss these species in greater detail in this biological opinion (see section 6, *Effects of the Action*).

Table 55. General location (coastal or pelagic waters) of likely exposure of species to proposed action's stressors. *These species fell within two exposure profiles. ** Species are prorated M/SI based on the area of overlap with the fishery, however have no observed interactions.

Observed Pelagic Exposure	**Potential Pelagic Exposure**
Sperm whale	Eastern Pacific scalloped hammerhead shark
North Pacific loggerhead sea turtle	Guadalupe fur seal*
Leatherback sea turtle*	Mexico Humpback whale
Green sea turtles*[11]	Central America Humpback whale
Olive Ridley sea turtle*	Blue whale
Oceanic whitetip shark	Fin whale
Giant manta ray*	Sei whale
Indo-West Pacific scalloped hammerhead shark	North pacific right whale
Hawaiian monk sea critical habitat	West Pacific humpback whale
	Southern Resident Killer Whale
	Loggerhead sea turtle (both North and South Pacific)*
	Hawksbill sea turtle*
	Chambered Nautilus
	MHI Insular false killer whale**
Observed Coastal Exposure	**Potential Coastal Exposure**
Hawaiian monk seal critical habitat	Green sea turtle*
	Hawksbill sea turtle*
	Leatherback sea turtle*
	Leatherback sea turtle critical habitat
	Black abalone

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[11] Central North Pacific, Central South Pacific, Central West Pacific, East Pacific, East Indian – West Pacific, and Southwest Pacific DPS of green sea turtles.

Observed Pelagic Exposure	**Potential Pelagic Exposure**
	Black abalone critical habitat
	White abalone
	Central California coast coho salmon
	Central Valley spring-run Chinook salmon
	Central California coast coho salmon critical habitat
	Sacramento River winter-run Chinook salmon
	Sacramento River winter-run Chinook salmon critical habitat
	California coast steelhead
	California coast steelhead critical habitat
	California Central Valley steelhead
	Southern California coast steelhead
	Southern North American green sturgeon
	Southern North American green sturgeon critical habitat
	Giant manta ray*
	Southern Resident killer whale
	Hawaiian monk seal
	MHI Insular false killer whale critical habitat
	Stellar sea lion critical habitat
	Guadalupe fur seal
	Acropora globiceps
	Acropora jacquelineae
	Acropora retusa
	Acropora speciosa
	Euphyllia paradivisa
	Isopora crateriformis

9.1 Potential Coastal Exposure

By far the largest category, 29 species fall into the category of potential coastal exposures. Many of the species listed in Table 2 have distributions that are either limited to the Pacific coast of the continental United States or have pelagic distributions outside of the Action Area for this consultation. Black abalone and their critical habitat, white abalone and their critical habitat, Southern Resident killer whale, Guadalupe fur seal, stellar sea lion critical habitat, leatherback sea turtle critical habitat, Central California coast coho salmon, Central Valley spring-run chinook salmon, Sacramento River winter-run Chinook salmon, California coast steelhead, California Central Valley steelhead, southern California coast steelhead, southern North American green sturgeon, occur in freshwater ecosystems, estuaries, or coastal waters where DSLL fishing is prohibited. As a result, these species would be exposed only to potential stressors associated with the transit of fishing vessels to and from port. Those being; possible ship strikes, vessel noise, and vessel discharges.

Fresh, estuarine, and coastal waters of the Pacific Coast receive discharges from numerous wastewater systems, are exposed to extensive at-sea disposals of dredged materials and sewage effluent, oil spills, dredging, ocean dumping and disposal, and coastal development. These activities are sources of an extensive suite of point-source and non-point source pollutants. Against this background of physical and chemical stressors, it would be nearly impossible to detect the contribution of the relatively small number of fishing vessels that depart from Pacific Coast ports to participate in the DSLL fishery. These vessels cumulatively carry a few thousand gallons of diesel and lubricants which present such a minor pollution risk as to be discountable. The occasional loss of some (or all) material from a single vessel or spill will quickly evaporate or become dispersed in the water column when considering the vast volume of the ocean. However, any contribution of these vessels to background levels of the stressors listed above to the degradation of coastal habitat along the Pacific Coast of the United States would be immeasurably small.

Additional species expected to occur in the Action Area include *Acropora globiceps, A. jacquelineae, A. retusa, A. speciosa, Euphyllia paradivisa, Isopora crateriformis*, MHI IFKW critical habitat, all the green sea turtle DPSs, hawksbill sea turtle, and the giant manta ray. As mentioned above, fishing is restricted from coastal areas. As such, these species are exposed to all or any of the stressors emanating from vessel transits (e.g., vessel collision, vessel noise, vessel waste, discharge and emissions). These stressors may result in immediate exposures, or exposures later in time (e.g., interaction with derelict gear or vessel waste). However, we are reasonably certain that for all the species in the category of potential coastal exposures, effects from all or any stressors related to vessel transiting to be extremely unlikely and therefore discountable, and that the effects from vessel noise will not rise to the level of harm or harassment. Therefore, we conclude that these species and critical habitat are not likely to be adversely affected by the DSLL fishery. For detailed information, see the applicable stressor discussions below.

9.2 Observed Coastal Exposure

Longline fishing does not occur in coastal areas, as a result stressors emanating from vessel transits (e.g., vessel collision, vessel grounding, vessel noise, vessel waste, discharge and emissions) are the only stressors expected in coastal waters. We know of only one recorded observation of an interaction between the proposed action and ESA-listed resources in coastal areas which will be discussed in further detail below (see Section 9.5.14).

9.3 Potential Pelagic Exposure

Thirteen species fall into the potential pelagic exposure category for the vessel transit routes to ports only (i.e., they are not anticipated to occur in the fishery portion of the Action Area): Mexico humpback whale, West Pacific humpback whale, Southern Resident killer whale, Central America humpback whale, Guadalupe fur seal, blue whale, fin whale, sei whale, North Pacific right whale, East Pacific scalloped hammerhead shark, South Pacific loggerhead, hawksbill sea turtle, and chambered nautilus. These species would only be exposed to stressors emanating from vessel transits (e.g., vessel collision, vessel noise, vessel waste, discharge and emissions). These stressors may result in immediate exposures, or exposures later in time (e.g., interaction with derelict gear or vessel waste).

None of the species listed above have been observed to be hooked or entangled in DSLL fishery. Therefore, we are reasonably certain that for all the species in the potential pelagic exposure profile, effects from all or any stressors related to fishery interactions to be extremely unlikely and therefore discountable. For more information, see the applicable stressor discussion below.

9.4 Observed Pelagic Exposure

Fourteen species fall into the observed pelagic exposure category: North Pacific loggerhead sea turtles, Central North Pacific green sea turtles, Central South Pacific green sea turtles, Central West Pacific green sea turtles, East Pacific green sea turtles, East Indian-West Pacific green sea turtles, Southwest Pacific green sea turtles, olive ridley sea turtles, leatherback sea turtles, oceanic whitetip sharks, Indo-West Pacific scalloped hammerhead shark, giant manta rays, sperm whales, and MHI IFKWs. These species could be exposed to all or any of the stressors emanating from the proposed action: fishery interactions (e.g., vessel noise, vessel collision, entanglement in gear); and vessel waste, discharge, or emissions. These stressors may result in immediate exposures, or exposures later in time (e.g., interaction with derelict gear or vessel waste). As previously described, there are documented observations of interactions with fishing gear resulting in entrapment or entanglement with the species listed in the observed pelagic exposure profile. These are described in greater detail of this biological opinion in the *Effects of the Action* section.

9.5 Stressors Not Likely to Adversely Affect Listed Resources

9.5.1 Vessel Noise

The proposed action would expose all ESA-listed marine species under NMFS' jurisdiction found in both the coastal and pelagic exposure categories (both potential and observed) to noise from the vessels. Vessel sizes range up to nearly the maximum 100-ft limit, but the average size is 65 to 70 ft. DSLL vessels have displacement hulls and travel at speeds less than 10 kts. Man-made sounds can affect animals exposed to them in several ways such as: non-auditory damage to gas-filled organs, hearing loss expressed in permanent threshold shift (PTS) or temporary threshold shift (TTS) hearing loss, and behavioral responses. They may also experience reduced hearing by masking (i.e., the presence of one sound affecting the perception of another sound). Masking and behavioral avoidance are the most likely responses of animals in the vicinity of the DSLL fishing vessels.

Given the size of the DSLL fishery (the small number of vessels in the fishery and the wide area they cover), the fact that the sound field produced by the vessels in the fishery is relatively small and would move with the vessel, the animals would be moving as well, vessel transit vectors would be predictable, sudden or loud noises would be unlikely or infrequent, and, we would expect that any exposure to noises generated by this fishery would be short-term and transient. Numerous studies demonstrate that marine animals are unlikely to change their behavior when confronted with stimuli with these attributes. Although hydraulics may have the potential to create loud noises; due to the expected above water operations, frequency and duration of time these species spend at the surface, dissipation of sound from the source, and the poor transference of airborne generated sounds from the vessel to ocean water through the hull, it is highly unlikely noises generated from vessel operations would rise to the level of harming ESA-listed species considered in this consultation. Thus, NMFS expects this stressor would have insignificant effects on the ESA-listed resources in Table 2.

9.5.2 Collision with vessels

The proposed action would expose all ESA-listed marine species under NMFS' jurisdiction found in both the coastal and pelagic exposure categories (both potential and observed) to the risk of collision with vessels. Vessel sizes range up to nearly the maximum 100-ft limit, but the average size is 65 to 70 ft. DSLL vessels have displacement hulls and travel at speeds less than 10 kts. Vessel speed is an important component of the risk for a collision between a vessel and an individual from a listed species.

Turtles and monk seals

Kelly (2020) documented vessel collisions with sea turtles resulting in lethal and sub-lethal injuries. Sea turtles could potentially be struck by the transiting vessel during the proposed activities. NMFS (2008) estimated 37.5 vessel strikes of sea turtles per year from an estimated 577,872 trips per year from vessels of all sizes in Hawaii. More recently, we estimated as many as 200 green sea turtle strikes annually in Hawaii (Kelly 2020). If these turtle strikes are evenly distributed around the islands, the probability of a green sea turtle strike from any one vessel trip is extremely low (on average 0.035%, calculated by dividing the most recent strike estimate of 200 per year by the best estimate of all vessel transits of 577,872 per year). However, green sea turtle strikes are not evenly distributed throughout the islands. They are concentrated in areas where small vessel activity is highest (e.g., near small boat harbors and boat launches), such as Kaneohe Bay and Pearl Harbor on Oahu (Kelly 2020). Green sea turtles are most vulnerable to small vessels (< 15 m), travelling at fast rates (>10 kts) (Kelly 2020). Increased vessel speed decreases the ability of sea turtles to recognize a moving vessel in time to dive and escape being hit, as well as the vessel operator's ability to recognize the turtle in time to avoid it. The vessels used in the proposed action are not small and will be under a speed restriction in areas of known turtle activity. The Action Area is not in a location identified by Kelly (2020) as a hot spot for green sea turtle strikes, nor is it an area with significant overlap of high-density boating activity and sea turtle habitat. Therefore, the probability of a green sea turtle strike is likely less than the overall rate calculated above. Thus, we are reasonably certain the likelihood of exposure of any green sea turtle to vessel strikes from this action is extremely unlikely, and therefore discountable.

Tuna from this fishery is also landed in Pago Pago, American Samoa; which has a considerably smaller density of sea turtles in their surrounding waters compared to the density of green sea turtles around the Hawaiian Islands. We expect that the chances of a DSLL fishery vessel strike is even less due to the lower density of turtles around the islands where tuna is landed compared to the density around Hawaii.

The other sea turtle species have a lower rate of striking than green sea turtles. This is likely mostly due to their low abundance numbers and preference for deeper offshore waters (Kelly 2020). There were only four documented vessel strikes of hawksbill sea turtles between 1984 and 2020 and two olive ridley sea turtles in Hawaii (Kelly 2020). We have no documentation of vessel strikes on leatherback or loggerhead sea turtles in Hawaii. Because the probability of a vessel striking any other sea turtles is even lower than that of a green sea turtle, and because of the transit speeds in to port are slow, we are reasonably certain the likelihood of exposure of any individual is extremely unlikely, and therefore discountable.

According to PIFSC’s database there have been only four verified vessel strikes of Hawaiian monk seals between 1981 and 2016 (PIFSC 2017). Other wounds and blunt force trauma have been documented but wounds, especially those that have healed, are difficult to distinguish between vessel strikes and other blunt force trauma such as intentional killing.

Considering that vessels involved in this fishery do not move at speeds that typically pose collision risks when transiting, the rarity of document vessel strikes, that vessels would only be expected to transit through areas where monk seals may occur, and the low abundance and widely scattered nature of monk seals in the Action Area; we are reasonably certain the likelihood of exposure of any monk seal to vessel strikes from this proposed action is extremely unlikely, and therefore discountable.

Whales

Whales surface to breathe, with calves surfacing more regularly than adults. While at the surface, a whale is at risk of being struck by a vessel. Vanderlaan and Taggart (2007) found that the severity of injury to large whales is directly related to speed, the probability of lethal injury from large ships increased from 21% for vessels traveling at 8.6 kts, to over 79% for vessels moving at 15 kts or more. In a study by Lammers et al. (2013), 22 whale/vessel incidents were recorded from 1975 – 2003, with 14 of those occurring during the years from 1994 – 2003. Using the ten-year period of highest vessel strikes, and the same number vessel transits mentioned above, that calculates to a probably of a collision between a whale and a transiting vessel to be 0.0000024%. According to the study by Lammers et al. (2013), the vast majority (17) of the vessel strikes were from vessels traveling at speeds in excess of 15 kts, and nearly all of them occurred in close proximity to the coastline of the main four Hawaiian Islands.

Based on the expected transit speeds for vessels in this fishery, the collision risks from the references cited above, and the low abundance and widely scattered nature of the whale species in the Action Area; we are reasonably certain the likelihood of an individual from the whale being struck is extremely unlikely, and therefore discountable.

Sharks and Rays

Studies on scalloped hammerhead sharks have shown that they have well-developed electrosensory systems and vision (Kajiura 2001) that presumably enables them to detect activity in the water at a distance and to quickly move away from slow-moving vessels. While specific studies have not been conducted for oceanic whitetip sharks or giant manta rays for vessel avoidance, they are also elasmobranchs and highly mobile species. The lateral line in manta rays is poorly understood, however they also have a suite of other biological functions, which are considered highly sophisticated sensory systems (Bleckmann and Hoffmann 1999; Deakos 2010). This suggests that they possess similar capabilities of detection as other elasmobranchs and could avoid slow moving vessels as well. In addition, all three species remain below the surface of the water the vast majority of the time.

Because ESA-listed sharks and rays spend minimal time at the surface of the water, are highly mobile and likely able to detect and avoid a transiting vessel, and are widely scattered throughout the waters of the Action Area, we are reasonably certain the likelihood of exposure of any individual to vessel strikes from this proposed action is extremely unlikely, and therefore discountable.

Invertebrates

Chambered nautiluses are closely associated with steeply-sloped forereefs and muddy bottoms and are found in depths typically between 200 and 500 m and are not known to swim in the open water column nor found in shallow water depths except for rare occasions when the water is cold enough (Miller 2018). Open ocean environments and specific temperature gradients are considered geographic barriers to movement as the species does not swim through the mid-water (Miller 2018). Therefore, it is extremely unlikely a chambered nautilus would be exposed to vessels at the surface within this fishery and would only pertain to vessel trips that transit to American Samoa.

While it has properly been assumed for listed vertebrate species that physical contact of equipment or humans with an individual constitutes an adverse effect due to high potential for harm or harassment, the same assumption does not hold for ESA-listed corals due to two key biological characteristics:

1. All corals are simple, sessile invertebrate animals that rely on their stinging nematocysts for defense, rather than predator avoidance via flight response. So whereas it is logical to assume that physical contact with a vertebrate individual results in stress that constitutes harm and/or harassment, the same does not apply to corals because they have no flight response.
2. Most reef-building corals, including all the listed species, are colonial organisms, such that a single larva settles and develops into the primary polyp, which then multiplies into a colony of hundreds to thousands of genetically-identical polyps that are seamlessly connected through tissue and skeleton. Colony growth is achieved mainly through the addition of more polyps, and colony growth is indeterminate. The

> colony can continue to exist even if numerous polyps die, or if the colony is broken apart or otherwise damaged. The individual of these listed species is defined as the colony, not the polyp, in the final coral listing rule (79 FR 53852). Thus, affecting some polyps of a colony does not necessarily constitute harm to the individual.

Corals are sessile invertebrates which do not move locations except for extenuating circumstances such as when progeny are broadcasted into ocean currents or breakage and recolonization of substrate from severe weather events. Vessels are expected to use established transportation channels or be deep enough water to avoid contact with corals and would only pertain to transits in American Samoa.

For this same reason (vessels using established transportation channels), and the small number of transits that are expected to occur within Californian waters, and species' distribution and abundance, we would not expect either white or black abalone to be exposed to this particular stressor.

In conclusion, given the small number of vessels participating in the fishery, the small number of anticipated vessel trips, the slow vessel speeds during vessel transiting, the expectation that ESA-listed marine species would be widely scattered throughout the proposed Action Area, the potential for an incidental vessel strike is extremely unlikely to occur. Thus, we are reasonably certain the probability of exposure to this stressor by the ESA-listed resources in Table 2 and is discountable.

9.5.3 Introduction of Vessel Wastes and Discharges, Gear Loss, and Vessel Emissions

The diffuse stressors associated with the longline fisheries: vessel waste discharge, gear loss, and carbon emissions and greenhouse gasses, can affect both pelagic and coastal areas. ESA-listed resources could be exposed to discharges, and run-off from vessels that contain chemicals such as fuel oils, gasoline, lubricants, hydraulic fluids and other toxicants. Although leakage, wastes, and gear loss would occur as a result of the United States DSLL fishery, given the small number of vessels participating in the fishery, the small number of anticipated vessel trips, the expectation that ESA-listed marine species would be widely scattered throughout the proposed Action Area, the small chance that ESA-listed resources would be exposed, NMFS is reasonably certain the probability of exposure to measurable or detectable amounts of leakage, wastes, or gear from this fishery is extremely unlikely, and therefore discountable on the ESA-listed resources in Table 2 and Table 55.

United States DSLL fishery vessels burn fuel and emit carbon into the atmosphere during fishing operations and transiting. Parker et al. (2018), estimates that in 2011, the world's fishing fleets burned 40 billion liters of fuel and emitted 179 million tons of carbon dioxide greenhouse gasses into the atmosphere. Between 1990 and 2011, emissions grew by 28% primarily due to increased harvests of crustaceans, a fuel intensive fishery (Parker et al. 2018). While we don't have an accurate estimate of the carbon footprint of the United States DSLL fishery, we are reasonably certain the contribution to global greenhouse gases would not rise to the level of harm or harassment of any ESA-listed individuals based on the low number of participants in the fishery.

9.5.4 Species Not Likely to be Adversely Affected

Based on the preceding discussion the following species are not likely to be adversely affected by the proposed action because they occur in coastal areas only, where they are not subject to fishing but are only subject to potential stressors related to vessel transiting. Therefore, their exposure is considered highly unlikely and therefore discountable. These are: black abalone, white abalone, Southern Resident killer whale, Central California coast coho salmon, Central Valley spring-run chinook salmon, Sacramento River winter-run Chinook salmon, Hawaiian monk seal, hawksbill sea turtle, Eastern Pacific scalloped hammerhead shark, California coast steelhead, California Central Valley steelhead, southern California coast steelhead, southern North American green sturgeon, blue whale, fin whale, Mexico humpback whale, North Pacific right whale, and sei whale. Additionally, based on our evaluation of the proposed action, the following designated critical habitat for the following species is not likely to be adversely affect by the proposed action: black abalone critical habitat, white abalone critical habitat, leatherback sea turtle critical habitat, Hawaiian monk seal critical habitat, and MHI IFKW critical habitat.

Our analysis demonstrates that several species may be exposed to fishing, yet none of the species listed in the potential pelagic exposure category which overlap with the operational range of the fishery have been caught in DSLL fishery. We have concluded, based on the information that follows, that the effect of the action on the following species is discountable. As such, the following species are also not likely to be adversely affected by the proposed action: hawksbill sea turtle, Hawaiian monk seal, eastern Pacific scalloped hammerhead shark, blue whale, fin whale, Mexico humpback whale, North Pacific right whale, sei whale, and designated critical habitats for the Hawaiian monk seal and MHI IFKW.

9.5.5 Hawksbill sea turtle

Hawksbill life history is characterized by early development in the pelagic zone followed by later development in nearshore habitats. Hawksbills recruit to coastal habitats of the neritic zone at greater than 40 cm carapace length. Sub-adults and adults forage on coral reefs, primarily on sponges.

Unlike other species of sea turtles, the density of hawksbills is extremely low in the Action Area. In Hawaii, tracking of post- nesting hawksbills from the Big Island of Hawaii and Maui suggest that primary adult foraging habitat occurs along the Hamakua coast of the Big Island (Parker et al. 2009), and adults rarely enter deeper pelagic waters where the Hawaii-based fishery occurs (approximately 75 nm from the MHI). As evidence of this, there have been no records of hawksbill sea turtles being captured in United States longline fisheries in the Pacific (NMFS unpublished observer data). In addition to rarely occurring outside of coastal waters, densities of hawksbill turtles in the central North Pacific are also very low. In the entire central North Pacific, only a few dozen females are thought to nest annually in the region.

The combination of a coastal occurrence, low abundance in pelagic waters, and the absence of interactions with longline fisheries suggest that hawksbill turtles have probabilities of being exposed to the fishery that are extremely unlikely. Therefore, we are reasonably certain that exposure of hawksbill sea turtles is discountable.

9.5.6 Hawaiian monk seal

Evidence from the early 1990s suggests that Hawaiian monk seals may have been exposed to the longline fisheries (NMFS 2007). However, in 1991, NMFS established a permanent 50-mile protected species zone around the NWHI that is closed to longline fishing (56 FR 52214). Since 1993, no interactions with Hawaiian monk seals in the Hawaii longline fisheries have been reported (WPRFMC 2009b) and observers have not reported any interactions since coverage of the longline fisheries began in 1994. Although monk seals commonly travel more than 43 nm offshore, they are unlikely to encounter longline gear because these travels are more likely to represent transits from one foraging area (island, reef, or seamount) to another rather than transits to pelagic waters (NMFS 2007).

Because the occurrence and migratory movement of Hawaiian monk seals beyond the protected areas around the MHI and NWHI is exceptionally rare and the fishery occurs outside of these protected areas, exposure of Hawaiian monk seals is extremely unlikely and, therefore discountable.

9.5.7 Eastern Pacific scalloped hammerhead shark

Although scalloped hammerhead sharks have been captured in the DSLL fishery, those interactions are not likely to have involved Eastern Pacific scalloped hammerhead sharks for several reasons (for comparison, see narratives on Indo-West Pacific scalloped hammerhead sharks later in this section of the biological opinion, in the Environmental Baseline, and the Effects of the Action sections of this opinion). First, although the Action Area and the range of eastern Pacific scalloped hammerhead sharks overlap, no interactions have been reported in the area of overlap. This is likely because Eastern Pacific scalloped hammerhead sharks tend to have a coastal distribution and rarely engage in long distance migrations in pelagic waters.

The combination of a coastal occurrence, low abundance in pelagic waters, and the absence of interactions with longline fisheries suggest that exposure of Eastern Pacific scalloped hammerhead sharks is extremely unlikely, and therefore discountable.

9.5.8 Blue whale

A “stock” of endangered blue whales occurs in waters surrounding the Hawaiian archipelago (from the MHI West to at least Midway Island), although blue whales are rarely reported from Hawaiian waters. Acoustic monitoring has recorded blue whales off Oahu and the Midway Islands much more recently (Northrop et al. 1971; Thompson and Friedl 1982; Barlow 1994; McDonald and Fox 1999). Recordings made off Oahu showed bimodal peaks throughout the year, suggesting that the animals migrate into the area during summer and winter (Thompson and Friedl 1982; McDonald and Fox 1999). There are no reports of blue whale strandings in Hawaiian waters.

In biological opinions NMFS prepared on United States Navy training activities off the Hawaiian Islands, blue whales densities off the Hawaiian archipelago were estimated to range between 0.0002 animals per square kilometer (NMFS 2008). More recently, Bradford et al. (2017) report a uniform density value for blue whales of 0.00005 animals per square kilometer that is applicable to the United States Navy’s Hawaii Range Complex study area (see also United States

Navy 2017). That study area overlaps with the Action Area so these densities should be representative of blue whale densities in the Action Area. At these densities, blue whales have probabilities of being exposed to vessels involved in the DSLL fishery that are extremely unlikely, and therefore discountable.

9.5.9 Fin whale

In the North Pacific Ocean, fin whales occur in summer foraging areas in the Chukchi Sea, the Sea of Okhotsk, around the Aleutian Islands, and the Gulf of Alaska; in the eastern Pacific, they occur south to California; in the western Pacific, they occur south to Japan. Fin whales in the eastern Pacific winter from California south; in the western Pacific, they winter from the Sea of Japan, the East China and Yellow Seas, and the Philippine Sea (Gambell 1985).

In the North Pacific Ocean, the IWC recognizes two "stocks": (1) East China Sea and (2) rest of the North Pacific (Donovan 1991). However, as discussed in NMFS (2013) "Mizroch et al. (1984) concluded that there were five possible "stocks" of fin whales within the North Pacific: (1) East and West Pacific that intermingle around the Aleutian Islands; (2) East China Sea; (3) British Columbia; (4) Southern-Central California to Gulf of Alaska; and (5) Gulf of California." Additionally, "Mizroch et al. (1984) also identified five fin whale "feeding aggregations" in the Pacific Ocean: (1) eastern and (2) western groups that move along the Aleutians (Berzin and Rovnin 1966; Nasu 1974); (3) an East China Sea group; (4) a group that moves north and south along the west coast of North America between California and the Gulf of Alaska (Rice 1974); and (5) a group centered in the Sea of Cortez (Gulf of California"; NMFS 2013).

These stock groupings suggest that fin whales occur in waters off the Hawaiian archipelago infrequently, which is supported by density estimates of fin whales in Hawaii. In biological opinions NMFS prepared on United States Navy training activities off the Hawaiian Islands, fin whale densities off the Hawaiian archipelago were estimated to range between 0.0001 and 0.0007 animals per square kilometer (see NMFS 2008). More recently, Bradford et al. (2017) report a uniform density value for fin whales of 0.00006 animals per square kilometer that is applicable to the United States Navy's Hawaii Range Complex study area, which overlaps with and should be representative of their densities in the Action Area (see also United States Navy 2017). At these densities, fin whales have probabilities of being exposed to vessels involved in the DSLL fishery that are extremely unlikely, and therefore discountable.

9.5.10 Mexico humpback whale

Since 2001, the DSLL fishery has had four observed interactions with humpback whales (recognizing that the fishery only has 20% observer coverage). All four interactions occurred in an area in the migratory path for Hawaiian humpback whales, which is not listed as threatened or endangered, and does not overlap with migratory areas of Mexico humpback whales (NMFS unpublished data). Because there is almost no overlap between the distribution of DSLL fishing vessels and Mexico humpback whales, the probability of these humpback whales being exposed to the DSLL fishery is extremely unlikely, and therefore discountable.

9.5.11 North Pacific right whale

Historically, endangered North Pacific right whales occurred in waters north of the Hawaiian archipelago (Scarff 1986; Clapham et al. 2004). However, contemporary sightings of North Pacific right whale have been much farther north in the Okhotsk Sea, southeastern Bering Sea, adjacent areas of the Aleutian Islands, and in the western Gulf of Alaska (Brownell et al. 2001; Wade et al. 2006; Wade et al. 2011). Only a small handful of recent records exist in Hawaiian waters and are likely vagrant individuals (Herman et al. 1980; Sheldon et al. 2005; NMFS 2017c). Additionally, we believe their abundance is lower than 500 individuals, Pacific-wide (see https://www.fisheries.noaa.gov/species/north-pacific-right-whale).

NMFS' critical habitat designation for this species places the southern limit of its distribution at 40°N latitude, which is five degrees of latitude (more than 500 km) north of the northernmost sets associated with the DSLL fishery. More importantly, all of the observations of North Pacific right whales reported in the North Pacific since 1973 have been north of 50°N latitude or more than 1,600 km north of the northernmost sets associated with the DSLL fishery. Because the known distribution of North Pacific right whales does not appear to overlap with the Action Area, low abundance, and lack of many records in Hawaii waters, exposure of North Pacific right whales is extremely unlikely, and therefore discountable.

9.5.12 Sei whale

In the North Pacific Ocean, sei whales occur from the Bering Sea south to California (on the east) and the coasts of Japan and Korea (on the west). During the winter, sei whales are found from 20° to 23°N (Mizroch 1984; Gambell 1985). Horwood (1987) reported that 75 - 85% of the North Pacific population of sei whales resides east of 180° longitude. Sei whales are seen infrequently in waters off Hawaii and are considered more abundant in the area during the cool seasons (Barlow, 2006).

In biological opinions NMFS prepared on United States Navy training activities off the Hawaiian Islands, sei whale densities off the Hawaiian archipelago were estimated to range between 0.0001 and 0.0007 animals per square kilometer (see NMFS 2008). More recently, Bradford et al. (2017) report a uniform density value for sei whales of 0.00016 animals per square kilometer that is applicable to the United States Navy's Hawaii Range Complex study area, which overlaps with and should be representative of their densities in the Action Area (see also United States Navy 2017). At these densities, exposure of sei whales is extremely unlikely, and therefore discountable.

9.5.13 Designated critical habitats outside of the fishing area

Designated critical habitat for black abalone, Sacramento River winter-run Chinook salmon, Central California Coast coho salmon, Steller sea lion, leatherback sea turtles, California Coast steelhead, and Southern North American green sturgeon all occur outside the operational fishing range of the fishery. All of the designated area is contained within the EEZ off the United States Pacific coast. Because vessels participating in the DSLL fishery are prohibited from fishing in the EEZ off the United States Pacific coast (50 CFR 660), fishing operations should have no effect on this designated critical habitat, its physical or biological features (including the quantity or quality of prey that might be available to these species in the critical habitat), or its value to these species.

Conversely, these critical habitats would only be exposed to stressors emanating from vessel transits (e.g., vessel collision, vessel noise, vessel waste, discharge and emissions). Any contribution of the relatively small number of fishing vessels (e.g. *n*= 7 in 2018; *n*= 11 in 2019; *n*= 2 in 2020; NMFS unpublished logbook and observer data) that depart from Pacific Coast ports to participate in the DSLL fishery to the degradation of critical habitat that has been designated for these species would be immeasurably small. Therefore, we are reasonably certain the effects from the DSLL fishery will not measurably reduce the conservation value of the physical or biological features of critical habitat for any of these species, and are therefore insignificant.

9.5.14 Designated critical habitat for Hawaiian monk seals

In 1988, NMFS expanded the pre-existing critical habitat designation for the Hawaiian monk seal to include nearly all beach areas and nearshore waters of the Northwestern Hawaiian Islands. In August 2015, NMFS revised the critical habitat for monk seals again to generally include terrestrial and marine habitat in the MHI from 5 m inland of the shoreline seaward to the 200 meter depth contour and to extend the earlier designation in the Northwest Hawaiian Islands out to the 200 m depth contour (80 FR 50925, August 21, 2015). Nevertheless, the entirety of the critical habitat that has been designated for Hawaiian monk seals is encompassed in the MHI Longline Fishing Prohibited Area (see Figure 2.3). Therefore, fishing operations should have no effect on the designated critical habitat, its physical or biological features (including the quantity or quality of prey that might be available to Hawaiian monk seals with the designated area), or its value to Hawaiian monk seals.

However, we note that in October 2017, a 90 foot longliner (the F/V Pacific Paradise) grounded on a reef 450 yds. offshore of Waikiki within designated Hawaiian monk seal critical habitat. An emergency consultation (I-PI-18-1601-AG, PIR-2018-10305) was completed in 2018 to examine the effect of the removal and disposal of the vessel on designated monk seal critical habitat. NMFS determined that the removal and disposal of the grounded F/V Pacific Paradise was NLAA to designated monk seal critical habitat. The vessels participating in the DSLL are small, while the designated monk seal critical habitat is vast in size including terrestrial and marine habitat from 5 m inland of the shoreline seaward to the 200m depth contour (80 FR 50925, August 21, 2015). Therefore the potential effects of vessel grounding during transit are not expected to adversely affect the designated habitat when considering the areas vessels are likely to ground (nearshore transportation channels), speeds at which vessels operate in these areas, and the quality of these nearshore habitats when considering the essential features. Additionally, while there is some vertical habitat complexity in some nearshore reef habitats where vessels are likely to ground, Hawaiian monk seals generally tend to forage further offshore as the majority of their prey are bottom-associated species.

Considering the one vessel grounding between 2004 and 2017, results in a 3.72×10^{-3} chance a vessel may potentially ground when considering all trips that occurred in the fishery during this period (26,810 trips; Table 11). We also recognize that vessels ground due to user error, captains falling asleep at the wheel or failing to maintain proper watch standards as required by the

United States Coast Guard, navigation errors, or mechanical failures. Because the vessels participating in the DSLL fishery are small in size when compared to designated critical habitat (meaning they'd likely not affect large portions of the habitat at any one time), the monk seal critical habitat is within the Longline Fishing Prohibited Area (see Figure 2), and monk seals typically forage offshore compared to locations where vessels may ground; the grounding of a vessel would not be expected to adversely affect designated critical habitat, its physical or biological features (including the quantity or quality of pray that might be available to Hawaiian monk seals with the designated area), or its value to Hawaiian monk seals. Therefore, we are reasonably certain the effects from the DSLL fishery will not measurably reduce the conservation value of the physical or biological features of critical habitat for Hawaiian monk seals, and are therefore insignificant.

9.5.15 Designated critical habitat for Insular Main Hawaiian Islands false killer whales

NOAA Fisheries published a final rule on July 24, 2018, to designate critical habitat for the MHI IFKW by designating waters from the 45 m depth contour to the 3,200 m depth contour around the MHI from Niihau East to Hawaii, pursuant to section 4 of the ESA (83 FR 35062; NMFS 2018e). The entirety of the MHI IFKW critical habitat is encompassed in the MHI Longline Fishing Prohibited Area. We note, between 2004 and 2020, 26 sets were made in what is now designated critical habitat. However, all sets were in the geographical area before critical habitat was designated in 2018. Only one unauthorized set occurred in the longline exclusion zone after 2011 in an area now designated as critical habitat. However, at the time the set was made, critical habitat was not designated. Currently 5.4% of the DPS range overlaps with the DSLL fishery, accounting for approximately 4,223 square miles where interactions with the DPS may still occur (until such a time the area is closed), however this range does not overlap designated critical habitat. To date, no known adverse effects have been documented from fishing activities to MHI IFKW critical habitat.

MHI IFKW critical habitat would also be exposed to stressors emanating from vessel transits (e.g., vessel collision, vessel noise, vessel waste, discharge and emissions). Any contribution of the relatively small number of fishing vessels that transit this zone to its physical or biological features (including the quantity or quality of prey that might be available to MHI IFKW within the designated area), or its value to DPS is highly unlikely as no fishing is expected to occur within designated critical habitat. Therefore, we are reasonably certain the effects from the DSLL fishery will not measurably reduce the conservation value of the physical or biological features of critical habitat for Hawaiian monk seals, and are therefore insignificant.

10 APPENDIX B: FULL RESULTS FOR THE GIANT MANTA RAY ANALYSIS

Table 56. Results of scenarios 1 and 2 for 10 year population projections and r = 0.019. Numbers represent the percent difference between initial abundance and abundance in year 10; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00.

r=0.019	**Initial Abundance**									
Population	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Maximum 5-year Running Average Annual Interactions; Scenario 1; 10 years										
Palmyra	1.00	0.77	0.28	0.04	0.05	0.09	0.11	0.14	0.16	0.17
Johnston	0.70	0.34	0.06	0.07	0.12	0.14	0.15	0.17	0.18	0.19
Hawaii	0.70	0.34	0.06	0.07	0.12	0.14	0.15	0.17	0.18	0.19
95th Percentile Future Annual Interactions; Scenario 2; 10 years										
Palmyra	1.00	1.00	0.61	0.20	0.06	0.00	0.05	0.10	0.12	0.14
Johnston	1.00	0.66	0.23	0.01	0.06	0.10	0.12	0.15	0.16	0.17
Hawaii	1.00	0.77	0.28	0.04	0.05	0.09	0.11	0.14	0.16	0.17

Table 57. Results of scenarios 1 and 2 for 20 year population projections and r = 0.019. Numbers represent the percent difference between initial abundance and abundance in year 20; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00.

r=0.019	**Initial Abundance**									
Population	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Maximum 5-year Running Average Interactions; Scenario 1; 20 years										
Palmyra	1.00	1.00	0.62	0.08	0.10	0.19	0.25	0.32	0.35	0.37
Johnston	1.00	0.74	0.14	0.16	0.26	0.31	0.34	0.38	0.40	0.41
Hawaii	1.00	0.74	0.14	0.16	0.26	0.31	0.34	0.38	0.40	0.41
95th Percentile Future Annual Interactions; Scenario 2; 20 years										
Palmyra	1.00	1.00	1.00	0.44	0.14	0.01	0.10	0.22	0.27	0.32

r=0.019	Initial Abundance									
Johnston	1.00	1.00	0.50	0.02	0.14	0.22	0.27	0.33	0.36	0.38
Hawaii	1.00	1.00	0.62	0.08	0.10	0.19	0.25	0.32	0.35	0.37

Table 58. Results of scenarios 1 and 2 for 40 year population projections and r = 0.019. Numbers represent the percent difference between initial abundance and abundance in year 40; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00.

r=0.019	**Initial Abundance**									
Population	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Maximum 5-year Running Average Annual Interactions; Scenario 1; 40 years										
Palmyra	1.00	1.00	1.00	0.20	0.25	0.47	0.60	0.78	0.85	0.92
Johnston	1.00	1.00	0.35	0.40	0.64	0.77	0.84	0.94	0.98	1.02
Hawaii	1.00	1.00	0.35	0.40	0.64	0.77	0.84	0.94	0.98	1.02
95th Percentile Future Annual Interactions; Scenario 2; 40 years										
Palmyra	1.00	1.00	1.00	1.00	0.35	0.03	0.25	0.54	0.66	0.78
Johnston	1.00	1.00	1.00	0.05	0.35	0.54	0.66	0.82	0.89	0.95
Hawaii	1.00	1.00	1.00	0.20	0.25	0.47	0.60	0.78	0.85	0.92

Table 59. Results of scenarios 1 and 2 for 10 year population projections and r = 0.042. Numbers represent the percent difference between initial abundance and abundance in year 10; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00.

r=0.042	**Initial Abundance**									
Population	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Maximum 5-year Running Average Annual Interactions; Scenario 1; 10 years										
Palmyra	1.00	0.57	0.03	0.25	0.34	0.39	0.41	0.45	0.46	0.48
Johnston	0.49	0.09	0.22	0.37	0.42	0.45	0.46	0.48	0.49	0.50
Hawaii	0.49	0.09	0.22	0.37	0.42	0.45	0.46	0.48	0.49	0.50

r=0.042	**Initial Abundance**									
95th Percentile Future Annual Interactions; Scenario 2; 10 years										
Palmyra	1.00	1.00	0.39	0.07	0.22	0.29	0.34	0.40	0.42	0.45
Johnston	1.00	0.45	0.04	0.28	0.36	0.40	0.42	0.46	0.47	0.48
Hawaii	1.00	0.57	0.03	0.25	0.34	0.39	0.41	0.45	0.46	0.48

Table 60. Results of scenarios 1 and 2 for 20 year population projections and r = 0.042. Numbers represent the percent difference between initial abundance and abundance in year 20; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00.

r=0.042	**Initial Abundance**									
Population	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Maximum 5-year Running Average Annual Interactions; Scenario 1; 20 years										
Palmyra	1.00	1.00	0.06	0.63	0.86	0.97	1.04	1.13	1.17	1.20
Johnston	1.00	0.22	0.55	0.93	1.06	1.12	1.16	1.21	1.23	1.25
Hawaii	1.00	0.22	0.55	0.93	1.06	1.12	1.16	1.21	1.23	1.25
95th Percentile Future Annual Interactions; Scenario 2; 20 years										
Palmyra	1.00	1.00	0.99	0.17	0.55	0.74	0.86	1.01	1.07	1.13
Johnston	1.00	1.00	0.09	0.70	0.91	1.01	1.07	1.15	1.19	1.22
Hawaii	1.00	1.00	0.06	0.63	0.86	0.97	1.04	1.13	1.17	1.20

Table 61. Results of scenarios 1 and 2 for 40 year population projections and r = 0.042. Numbers represent the percent difference between initial abundance and abundance in year 40; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00.

r=0.042	**Initial Abundance**									
Population	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Maximum 5-year Running Average Annual Interactions; Scenario 1; 40 years										
Palmyra	1.00	1.00	0.21	2.08	2.84	3.22	3.45	3.75	3.88	3.99

r=0.042	**Initial Abundance**									
Johnston	1.00	0.72	1.82	3.09	3.52	3.73	3.86	4.03	4.09	4.16
Hawaii	1.00	0.72	1.82	3.09	3.52	3.73	3.86	4.03	4.09	4.16
95th Percentile Future Annual Interactions; Scenario 2; 40 years										
Palmyra	1.00	1.00	1.00	0.55	1.82	2.46	2.84	3.35	3.55	3.75
Johnston	1.00	1.00	0.29	2.33	3.01	3.35	3.55	3.82	3.93	4.04
Hawaii	1.00	1.00	0.21	2.08	2.84	3.22	3.45	3.75	3.88	3.99

11 APPENDIX C: POWER ANALYSIS FOR DETECTING DIFFERENCES IN POPULATION GROWTH RATES WITH AND WITHOUT DSLL FISHERY MORTALITIES

As part of our risk analysis for species/DPSs that are in decline, we assumed that the current estimated population trends were inclusive of mortalities from the DSLL. We projected populations for 40 years (20 years for West Pacific leatherbacks, see Section 1.3.1, Jeopardy Analysis), each year adding back in the projected mortalities from the fishery. We then calculated the new population growth rate after removing fishery mortalities for 40 years (20 years for West Pacific leatherbacks). Here, we detail our power analysis approach to determining if the difference in population growth rates with and without fishery mortalities is statistically significant.

There are four key quantities in determining statistically significant differences between two mean population growth rates (Cohen 1988):

- Sample size (number of years needed to detect the difference).
- Effect size (difference between the means divided by the pooled standard deviation; Cohen 1988).
- Significance level or probability of making a Type I error (probability of determining the fishery is causing a statistically significant difference in a species rate of decline when it is not).
    - We use 0.05 for determining significance, in this case we are saying we will accept a 5% chance of making a Type I error, or we are 95% sure the fishery is not causing a statistically significant difference in a species rate of decline.
- Power (P) which equates to a (1-P) probability of making a Type II error (probability of determining that the fishery is not causing a statistically significant difference in a species rate of decline when it is).
    - We used P=80% which implies that we will accept a 20% (100%-80%) chance of determining that the fishery is not causing a statistically significant difference in a species rate of decline when it is. Similar to 0.05 being the common convention for determining statistical significance, 0.80 is the common convention for determining power (Button et al. 2013).

In a power analysis, you can determine any one of these quantities providing you know the other three.

Assumptions of the analysis:

- If we only had a 95% confidence or credible interval for a mean population growth rate, we assumed a normal distribution and that the standard deviation was the width of the confidence interval divided by 3.92. In a normal distribution, the 95% CI is 1.96 standard deviations away from the mean, therefore there are 1.96*2 = 3.92 standard deviations across the width of the 95% CI.

- We assume the same variance for the population growth rates with and without the fishery.

In our analysis, we asked the question "how many years of data would be needed to detect a significant difference between population growth rates with and without 40 years of mortalities from the DSLL fishery (20 years for West Pacific leatherback sea turtles) with a significance level of 5% and with a power of 80%". We used the R package pwrrs (Bulus 2023) to calculate the minimum number of years (i.e. sample size) required to detect the difference between population growth rates with and without 40 years of fishery mortalities given the level of significance and power describe above. Given that we know removing fishery mortalities will increase population growth rates (or reduce a population's rate of decline), we use a one-tailed test to determine the sample size needed to detect that the population's rate of decline is significantly reduced by the removal of fishery mortalities.

Hypotheses

- Null (H0): population growth rates are not different with and without DSLL fishery mortalities
- Alternate (HA): population growth rates are lower (declining faster) with the DSLL fishery mortalities compared to population growth rates with the fishery mortalities removed.

We used the following function of the 'pwrrs' package in R for this test is (with MHI IFKW values used as an example, see Table 1):

<pwrss.t.2means(mu1 = -0.0553, mu2 = -0.05331, sd1 = 0.0212, sd2 = 0.0212, kappa = 1, power = 0.80, alpha = 0.05, alternative = "less")>

- mu1 is the population growth rate with the fishery mortalities
- mu2 is the population growth rates with the fishery mortalities removed
- sd1 and sd2 are the standard deviations which we assume to be the same
- kappa is the ratio of the sample size needed for each group (i.e. mu1 and mu2), in this case the sample size is the same for both groups resulting in kappa = 1
- power is as we described above, the probability of making a Type II error and we use 0.80
- alpha is the significance level as describe above, the probability of making a Type I error and we use 0.05.
- alternative is the direction of the hypothesis test, in this case we are using a one-tailed test to detect that the population growth rate with the fishery mortalities is less than the population growth rate without the mortalities. For a two-tailed test

where we didn't know the direction of the difference we would use alternative = "not equal".

Table 62 contains the input and output values for the species/DPSs for which we made the calculations. We found that at least 1,405 years of data would be needed to detect a significant difference between the population growth rates with and without DSLL mortalities for the species/DPSs considered. These numbers of years are well beyond the 40 years (20 years for West Pacific leatherbacks) over which we removed fishery mortalities. Furthermore, they are well beyond any timeframe we can reasonably predict the action's effects or the populations' abundances We therefore would not be able to reject the null hypothesis that population growth rates are the same with and without the fishery mortalities with 40 years of data.

Overall, for all species/DPSs considered, we do not have the predictive power to detect a difference in population growth rates with and without mortalities from the DSLL fishery.

11.1 Literature Cited

Bulus, M. 2023. pwrss: Statistical Power and Sample Size Calculation Tools. R package version 0.3.1. https://CRAN.R-project.org/package=pwrss

Button, K.S., J.P. Ioannidis, C. Mokrysz, B.A. Nosek, J. Flint, E.S. Robinson and M.R. Munafò. 2013. Power failure: why small sample size undermines the reliability of neuroscience. Nature reviews neuroscience, 14(5), pp.365-376.

Cohen, J. 1988. Power analysis for the behavioral sciences. 2nd Edition. Lawrence Erlbaum Associates, Hilldale, NJ. 579 p.

Table 62. Input and output values used in R to determine the number of years of data required to detect a difference in population growth rates with and without DSLL fishery mortalities.

Species/DPS	Inputs used in the pwrss command in R			Number of years needed to detect a significant difference
	mu1 (population growth rate with fishery mortalities)	mu2 (population growth rate after removing fishery mortalities for 40 years [20 for WP leatherbacks])	Sd (standard deviation of population growth rate)	
Central South Pacific Green Sea Turtle	-0.058	-0.05786	0.018	204,404
Central West Pacific Green Sea Turtle	-0.058	-0.057942	0.018	1,190,934
East Indian-West Pacific Green Sea Turtle	-0.0111	-0.0110989	0.028	8,011,776,646
Leatherback Sea Turtle (East Pacific Population)	-0.0811	-0.0802	0.0979	146,313
Leatherback Sea Turtle (West Pacific Population)	-0.06	-0.0575	0.0896	15,884
Main Hawaiian Islands Insular False Killer Whale	-0.0553	-0.05331	0.0212	1,405