# Endangered Species Act Section 7(a)(2) Consultation

# Oceanic Whitetip Shark and Giant Manta Ray Supplemental Biological Opinion

| | |
|---|---|
| Action Agency: | National Marine Fisheries Service, Pacific Islands Region, Sustainable Fisheries Division |
| Federal Action: | Supplement to the Authorization of the Hawaii Deep-Set Longline Fishery; Effects to Oceanic Whitetip Sharks and Giant Manta Rays. |
| Consultation Conducted by: | National Marine Fisheries Service, Pacific Islands Region, Protected Resources Division |
| NMFS File No. (ECO): | PIRO-2022-02105 |
| PIRO Reference No.: | I-PI-22-2058-DG |
| Approved By: | MALLOY.SARAH.JOAN.1262526743 Digitally signed by MALLOY.SARAH.JOAN.1262526743 Date: 2022.09.28 09:34:38 -10'00'<br>Sarah Malloy<br>Acting Regional Administrator, Pacific Islands Region |
| Date Issued: | September 28, 2022 |

## Contents

1 Introduction 7
  1.1 Consultation History 8
  1.2 Description of the Proposed Action 12
    1.2.1 Requirements of the False Killer Whale Take Reduction Plan 18
    1.2.2 Jeopardy analyses 21
  1.3 Application of this Approach in this Consultation 24
    1.3.1 Action Area 24
  1.4 Approach to Evaluating Effects 25
    1.4.1 Tipping Points 28
    1.4.2 Climate Change 28
    1.4.3 Statistical Intervals and Anticipated Future Incidental Captures Used in this Assessment 30
    1.4.4 Evidence Available for this Consultation 32
2 Status of Listed Resources 35
  2.1 Status of Listed Species That Are Likely to be Affected 35
    2.1.1 Giant Manta Ray 36
    2.1.2 Oceanic Whitetip Shark 47
3 Environmental Baseline 55
  3.1 Threats Posed by A Changing Global Climate 55
  3.2 Fisheries Interactions 57
    3.2.1 Other Domestic Fisheries in the Action Area 61
    3.2.2 Hawaii Deep Set Longline Historical Effort 62
  3.3 Pollution and Marine Debris 68
  3.4 Synthesis of the Environmental Baseline 69
4 Effects of the Action 69
  4.1 Potential Stressors 70
  4.2 General Responses of ESA-listed Species to the Fishery 70
    4.2.1 Entanglement in Longline Gear 71
    4.2.2 Hooking 72
    4.2.3 Trailing Gear (Line) 73
  4.3 Species-Specific Analyses 73
    4.3.1 Giant Manta Ray 75
    4.3.2 Oceanic Whitetip Shark 93
  4.4 Cumulative Effects 109
5 Integration And Synthesis 110
  5.1 Giant Manta Ray 111
  5.2 Oceanic Whitetip Shark 113
6 Conclusion 115
7 Incidental Take Statement 115
  7.1 Amount or Extent of Take 115
  7.2 Reasonable and Prudent Measures 117
  7.3 Terms and Conditions 117
  7.4 Conservation Recommendations 118
  7.5 Reinitiation Notice 118

8 Literature Cited ........................................................................................119
9 Appendix A: Exposure Profiles ........................................................................138
9.1 Stressors Not Likely to Adversely Affect Listed Resources ...............................138
9.1.1 Vessel Noise ....................................................................................................138
9.1.2 Collision with vessels .....................................................................................138
9.1.3 Introduction of Vessel Wastes and Discharges, Gear Loss, and Vessel Emissions ........................................................................................................139
9.1.4 Vessel Groundings .........................................................................................139
10 Appendix B: Full Results For The Giant Manta Ray Analysis ...........................140

# List of Figures

Figure 1. Generalized depiction of swordfish (shallow) and tuna (deep) longline gear configurations (NMFS 2018a). ....................13

Figure 2. Fine scale map of the MHI fishing exclusion zone and critical habitats. ....................15

Figure 3. Boundaries of marine national monuments in the Pacific Islands where longline fishing is prohibited. ....................16

Figure 4. Map of the MHI longline fishing prohibited area, the FKWTRP southern exclusion zone, and the Papahānaumokuākea Monument. ....................19

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. ....................21

Figure 6. Location of Hawaii DSLL fishery as illustrated by effort (sets) from 2004 to 2020 (*n*= 65,245 sets). Colors from blue to red illustrate areas of lower to higher numbers of sets. ....................25

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. ....................27

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). ....................40

Figure 9. Geographical distribution of the oceanic whitetip shark (Last and Stevens 2009). ....................49

Figure 10. Projected ratios of 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). ....................52

Figure 11. The boundaries of the Western Central Pacific Fisheries Commission (WCPFC) (west of the line) and the Inter-American tropical tuna Commission (IATTC) (east of the line). ....................59

Figure 12. 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 2021; left panel) and for the DSLL (right panel) from 2016 to 2019. ....................60

Figure 13. Summary of effort for the Hawaii DSLL fishery since 2004. See Table 5 for the data (data from NMFS 2018a and pers. comm. J. Makaiau to A. Garrett, May 19, 2022). ....................63

Figure 14. Hooks per year deployed by the Hawaii DSLL fishery (gray 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. ....................64

Figure 15. Effort metrics for the Hawaii DSLL fishery. Top left shows the increase in hooks per vessel from 2004 to 2021, 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 2020. Bottom shows hooks per set from 2004 to 2021, dotted line represents a linear regression that is described by the equation. ....................................................................65

Figure 16. Trend in annual hooks per year for the DSLL from 2004 to 2021. 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. ....................................................66

Figure 17. Geographical representation of observed aggregate giant manta ray and unidentified *Manta/Mobula* and ray interactions with the DSLL fishery between 2004 and 2018 (*n* = 312)...77

Figure 18. Heatmap comparison of observed giant manta ray and unidentified *Manta/Mobula* and ray interactions (left; *n* = 312) versus aggregated fishing effort during this same time frame (right; *n*= 65,245 from 2004 to 2018). .........................................................................78

Figure 19. Numbers of observed giant manta ray incidentally captured per month in the DSLL fishery from 2004 to 2020. ..........................................................................................78

Figure 20. Giant manta ray interactions per year in the DSLL fishery between 2004 and 2020 for observed interactions (blue bars) and estimated interactions (orange bars; from McCracken 2019a and McCracken and Cooper 2020a, 2020b). .................................................................79

Figure 21. 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). .........................................................................86

Figure 22. Map of fishing area for the DSLL fishery. Small green circles are the center points of the subpopulations in Table 10. Yellow circles represent an area around the subpopulation center points with a radius of 500 km. ..........................................................................89

Figure 23. 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). ....................................................91

Figure 24. 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). .........................................................................92

Figure 25. Number of observed oceanic whitetip sharks incidentally captured by month in the Hawaii DSLL fishery from 2004 to 2020 (*n* = 5,149). .............................................................95

Figure 26. Oceanic whitetip shark interactions per year in the DSLL fishery between 2004 and 2020 for observed interactions (blue bars) and estimated interactions adjusted for observer coverage (orange bars; McCracken 2019a; McCracken and Cooper 2020a, 2020b). ..................96

Figure 27. Geographical representation of observed aggregate oceanic whitetip shark interactions with the DSLL fishery between 2004 and 2020 (*n*= 5,149). ......................................................97

Figure 28. Heatmap comparison of observed oceanic whitetip shark interactions (left; *n* = 5,149) versus aggregated fishing effort during this same time frame (right; *n*= 65,245 from 2004 to 2020). ..........................................................................................................98

## List of Tables

Table 1. Projections for certain climate parameters under Representative Concentration Pathway 8.5 (values from IPCC 2014)....29

Table 2. Listed resources within the action area that may be affected by the proposed action. ....35

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....42

Table 4. The estimated number and amount of exempted take of oceanic whitetip shark and giant manta rays, noting annual interactions and associated mortality with various species in the Hawaii SSLL fishery (NMFS 2019). ....61

Table 5. The number of active vessels and fishing effort in the Hawaii DSLL fishery, 2004-2020 (includes effort in both WCPO and EPO). Blue shading denotes the lowest values in the table, yellow shading denotes the highest values (NMFS 2018a). ....62

Table 6. The proportion of estimated incidental captures identified as giant manta ray and *Mobula* (devil rays) from 2004 to 2020 and the 95% confidence interval for the proportion. .....75

Table 7. The proportion of estimated incidental captures identified as giant manta ray, *Mobula* (devil rays), and pelagic stingrays from 2017 to 2019 and the 95% confidence interval for the proportion. ....76

Table 8. Anticipated annual and 95th percentile (from McCracken 2019a and McCracken and Cooper 2020a, 2020b; 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. Values in parentheses are the numbers anticipated inclusive of unidentified *Manta/Mobula*......79

Table 9. At-vessel, post-interaction, and total mortality rate estimates for giant manta ray interacting with the DSLL fishery from 2004 to 2020. 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. ....82

Table 10. 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. ....85

Table 11. Parameters and values used for Scenarios 1 and 2 subpopulation projections. ....89

Table 12. 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; NMFS 2021b) and the maximum 5-year running average and cumulative sum. Numbers in parentheses represent the *observable* portion of the interactions with the fishery switching to monofilament leaders.........94

Table 13. Estimates of number of oceanic whitetip sharks that will be incidentally captured and killed within the Hawaii DSLL fishery. ....100

Table 14. 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. ....................102

Table 15. 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). ....................103

Table 16. Estimated effect of Hawaii DSLL fishery on the Pacific population of oceanic whitetip sharks incidentally captured and killed by the fishery 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). ....................107

Table 17. Estimated effect of Hawaii DSLL fishery on the Pacific population of oceanic whitetip sharks incidentally captured and killed by the fishery 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). ....................108

Table 18. The anticipated mean annual captures and mortalities from the Hawaii DSLL. These are the estimated number of captures, expanded from observed number of captures. ....................116

Table 19. Reinitiation triggers for the Hawaii DSLL. These are the estimated number of captures, expanded from observed number of captures. Exceeding the maximum 5-year running average/sum over any five consecutive years is a reinitiation trigger. ....................116

# 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

---

[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 threatened giant manta ray and oceanic whitetip shark.

On July 5, 2022, 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. As a result, the 2019 regulations are once again in effect, and we are applying the 2019 regulations here. For purposes of this consultation, 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.

This supplemental biological opinion to the 2014 Hawaii Deep-set 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' status review for on the giant manta ray and oceanic whitetip shark (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), and other sources of information as cited herein.

## 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 2004). 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 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 (DPS), 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. Since a DPS is a "species" under the ESA, we use the term biological species to describe the larger taxonomic 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 2017). 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 increased the amount of take exempted for these species and split the incidental take for olive ridley turtles to account for both endangered and threatened olive ridley turtles, and the 11 green sea turtle DPS. Specifically, NMFS 1) increased the amount of take by interaction that was exempted for North Pacific loggerhead turtles from 9 to 18 across 3 years; 2) increased olive ridley turtles from a total of 99 (inclusive of both the endangered and threatened populations) to 141 for the endangered Mexico and threatened eastern Pacific populations, and 42 for the threatened western

Pacific population (combined total 183); and 3) increased the globally listed green sea turtle from a total of 9 interactions across three years to a total of 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).

On January 22, 2018, NMFS listed the giant manta ray and oceanic whitetip shark as threatened (83 FR 2916 and 83 FR 4153, respectively).

On October 4, 2018, SFD requested PRD to reinitiate formal consultation on the DSLL fishery to include giant manta ray and oceanic whitetip shark. NMFS reinitiated consultation pursuant to 50 CFR 402.16(a)(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 Western Pacific Fishery Management Council (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 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 Hawaii Longline Association started discussions to minimize the effects of the action.

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 Hawaii Longline Association for their proposal to replace wire leaders with monofilament, and other associated improvements to the fishery (letter to Hawaii Longline Association 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 results show that prohibition of wire leaders in the DSLL is 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 Fishery Ecosystem Plan 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 deep-set longline fishery 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 August 10, 2022, to ensure Section 7(a)(2) compliance, NMFS determined to complete a supplemental opinion to the 2014 Hawaii Deep-set Longline Biological Opinion (as supplemented in 2017) on recently listed species, including oceanic whitetip sharks and giant manta rays. This supplemental opinion addresses only impacts to 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, remains valid and effective until replaced.

## 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 (WPFMC 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 Exclusive Economic Zone (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 discrete unbroken section of 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 2019).

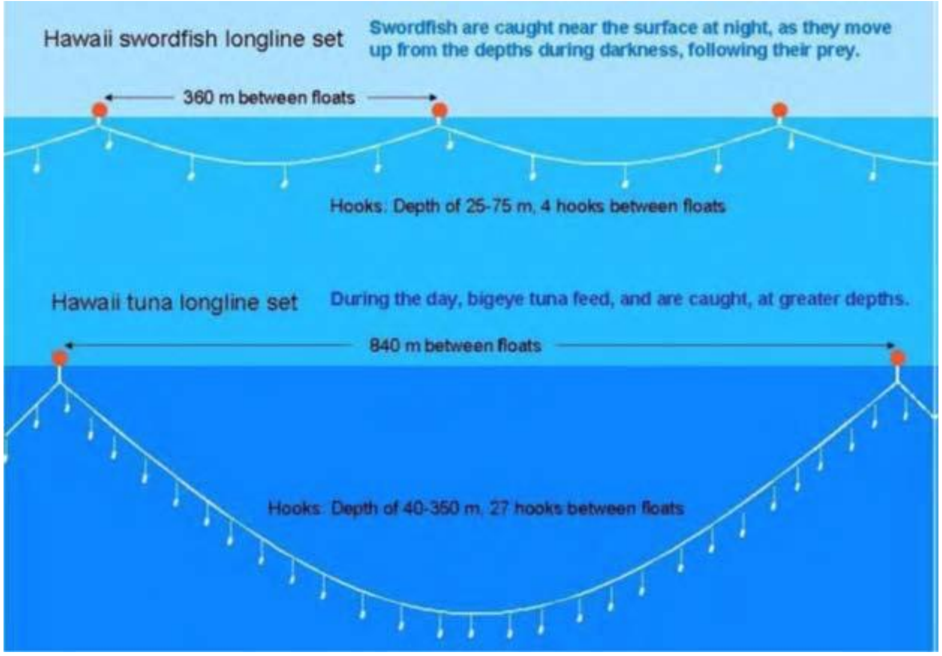


Figure 1. Generalized depiction of swordfish (shallow) and tuna (deep) longline gear configurations (NMFS 2018a).

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 Convention (WCPFC) Area Endorsement.
- Protected Species Workshop Certificate.
- Western Pacific Receiving Vessel Permit, if applicable.
- State of Hawaii Commercial Marine License.

Reporting, Monitoring, and Gear Identification;

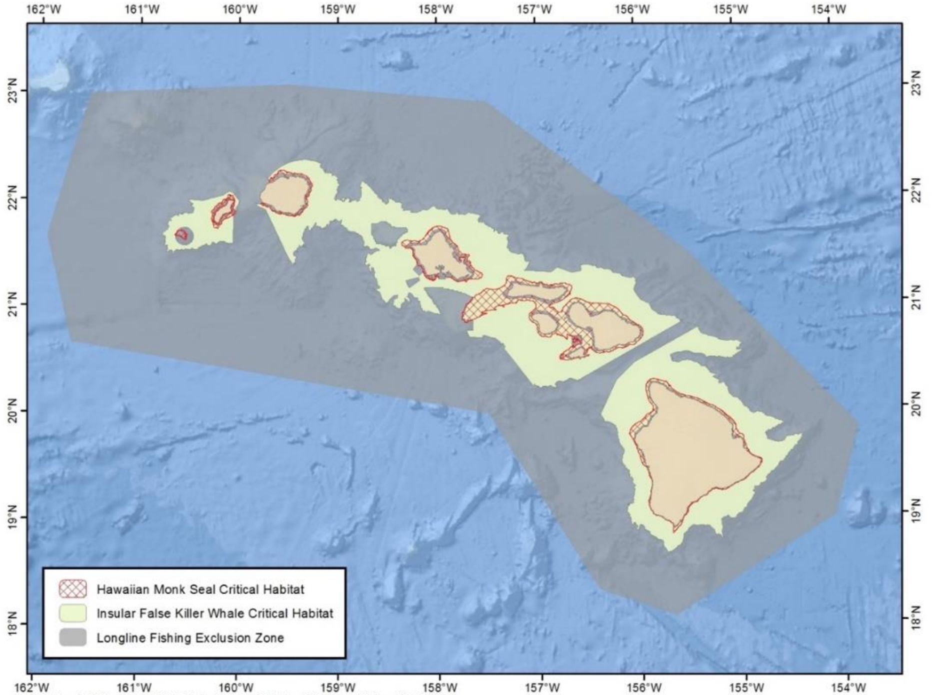
- NMFS Western Pacific Daily Longline Fishing Log for recording effort, catch, and other data.
- NMFS Pacific Transshipment Declaration Form, if applicable.
- Marine Mammal Authorization Program Mortality/Injury Reporting Form.
- Vessel monitoring system.
- Vessel and fishing gear identification.

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. NMFS places observers on every SSLL trip and on about 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 NWHI. 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).
- Papahānaumokuākea Marine National Monument: Prohibited commercial fishing in the Monument. Fishing vessels entering and leaving the boundaries of the Papahānaumokuākea Monument are required to notify the Papahānaumokuākea 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.

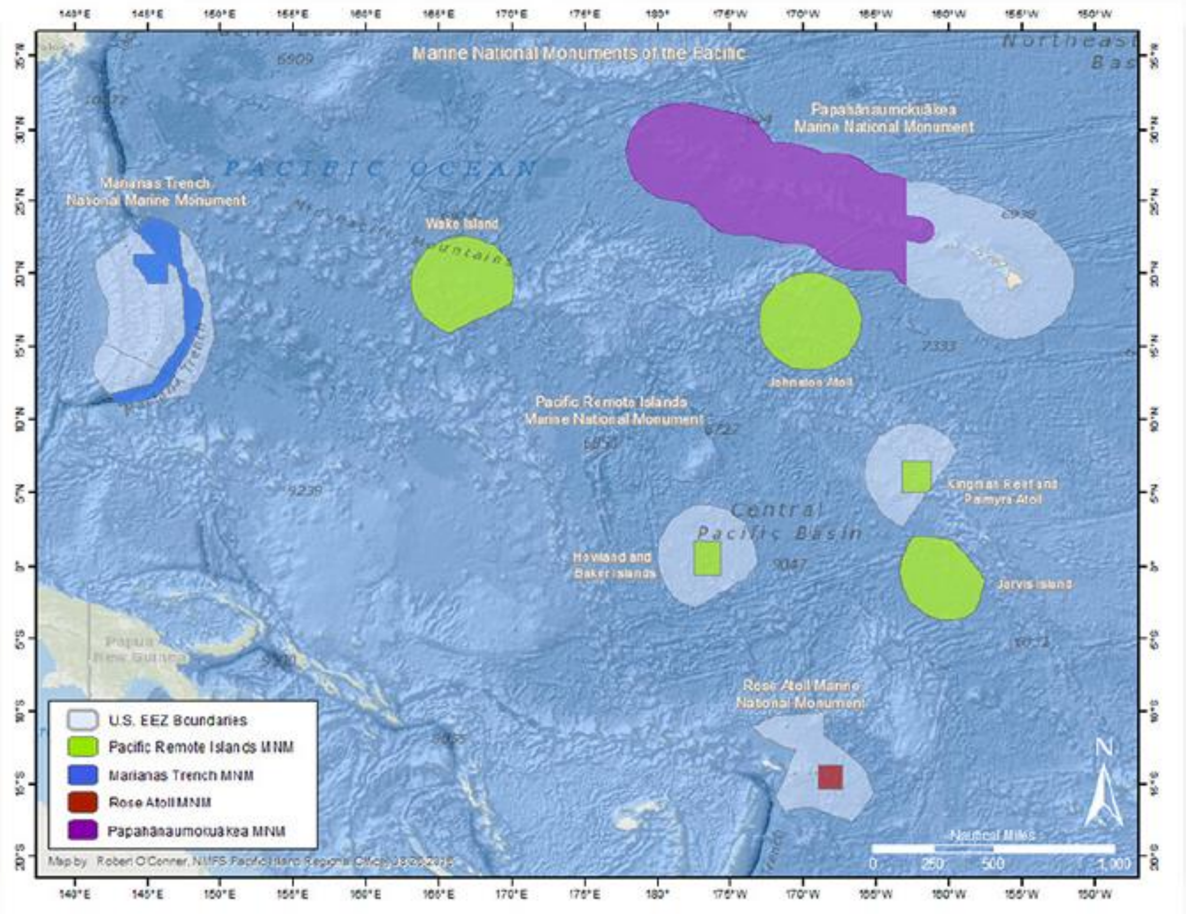


Figure 3. Boundaries of marine national monuments in the Pacific Islands where longline fishing is prohibited.

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.

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' proposed rule (86 FR 55790) 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 all164 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 (see Table 5 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 FFKWTRP 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).

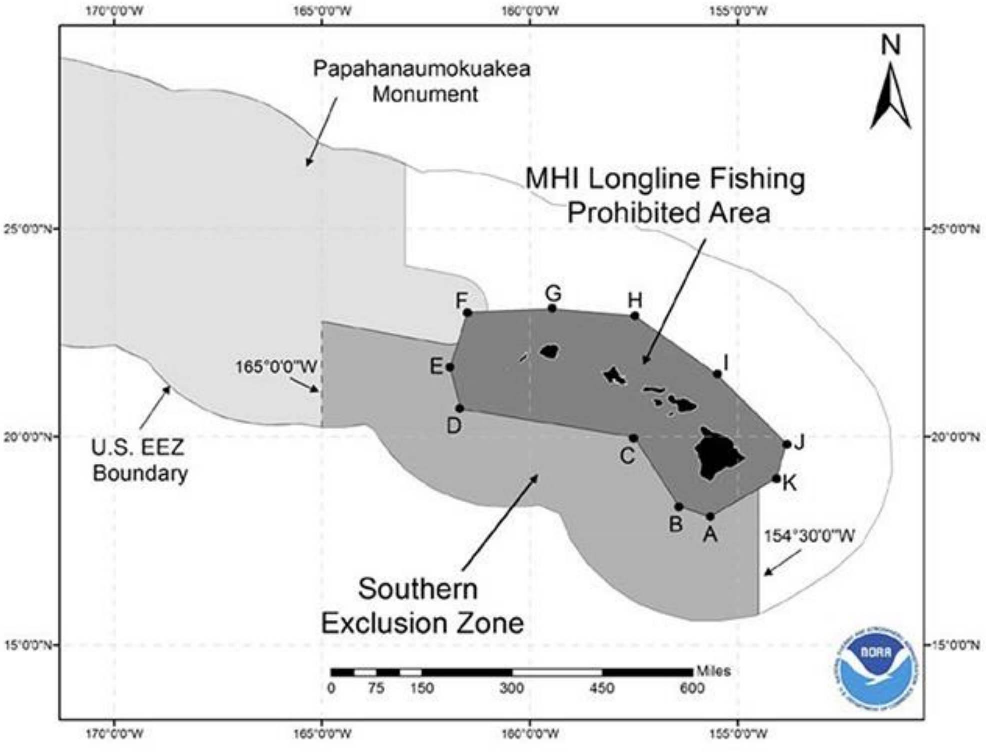


Figure 4. Map of the MHI longline fishing prohibited area, the FKWTRP southern exclusion zone, and the Papahānaumokuākea Monument.

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 Plan 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.

### *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. Every section of a biological opinion from its opening page and its conclusion and all of the information, evidence, reasoning, and analyses presented in between is designed to help answer these two questions. 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 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.

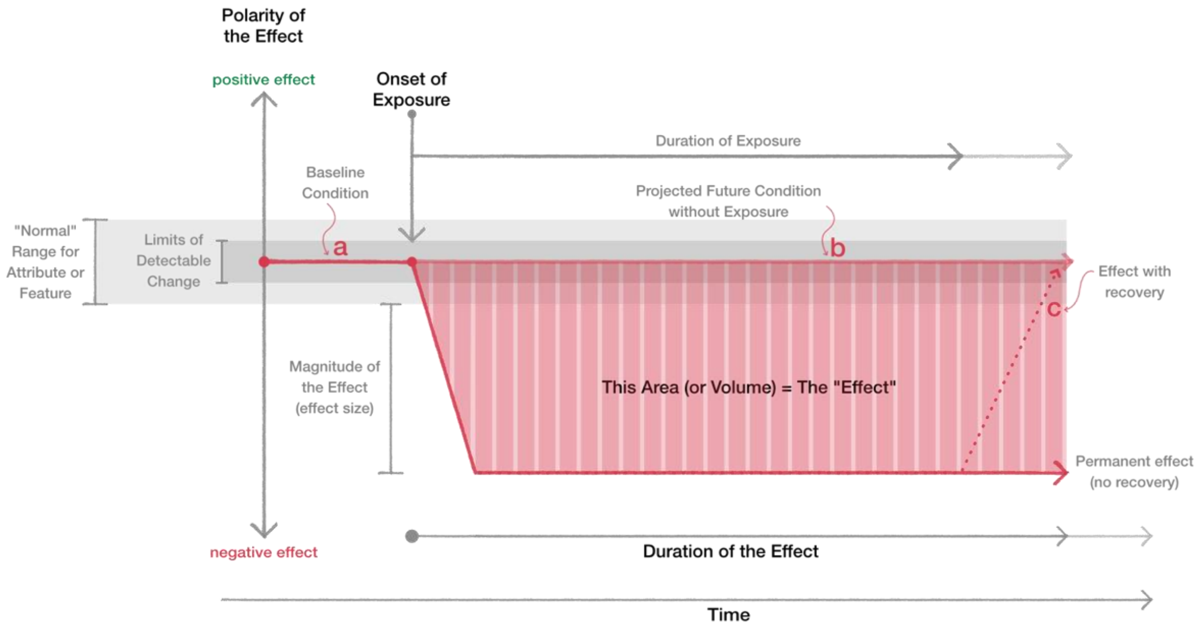


Figure 5. A schematic of the various elements encompassed by the word "effect." The vertical bars in the figure depict a series of annual "effects" (negative changes from a pre-existing or "baseline" condition) that are summed over time to estimate the action's full effect. See text for a more complete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

### 1.2.2 Jeopardy analyses

The 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.

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 the Species* 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). 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.

## 1.3 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 oceanic whitetip sharks and giant manta rays. 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.

### 1.3.1 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 10° N 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.

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 Papahānaumokuākea 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).

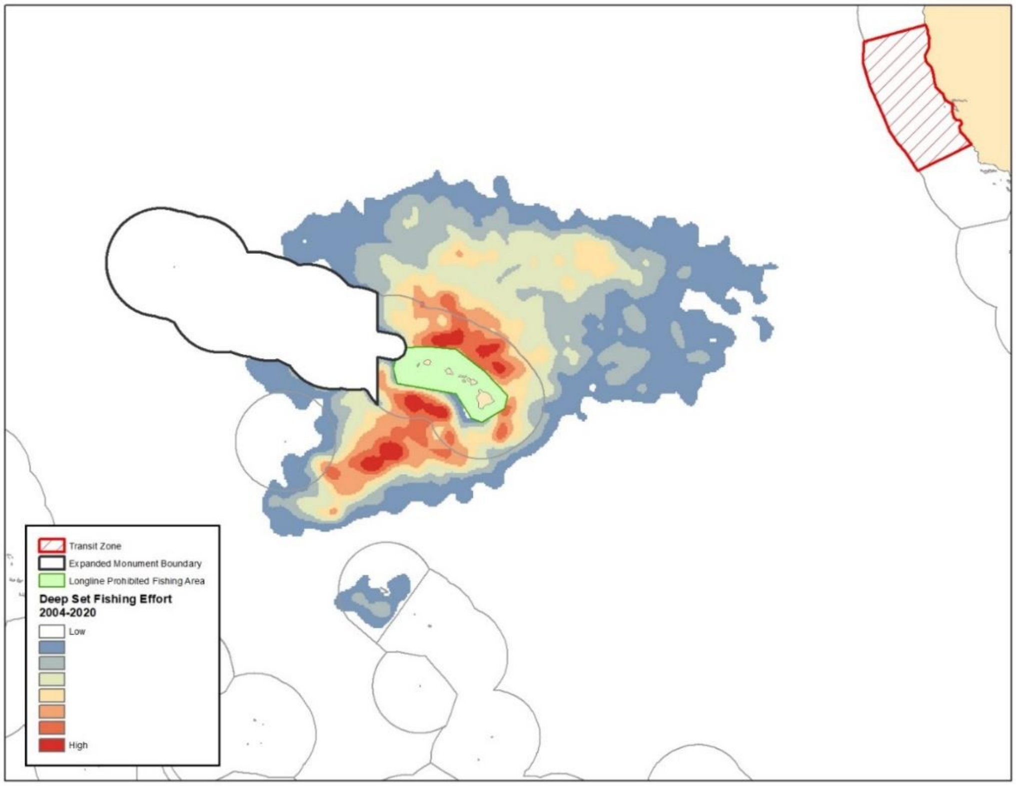


Figure 6. Location of Hawaii DSLL fishery as illustrated by effort (sets) from 2004 to 2020 (*n*= 65,245 sets). Colors from blue to red illustrate areas of lower to higher numbers of sets.

## 1.4 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 in 2021, the first and second quarters had 14.3% and 17.7% observer coverage respectively. In contrast, no data are available to characterize exposure to vessel strikes or discharges of waste in the DSLL.

We began by 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). We then evaluated the likelihood that 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 giant manta rays and oceanic whitetip sharks.

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 the giant manta ray and oceanic whitetip shark 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 to the listed species.

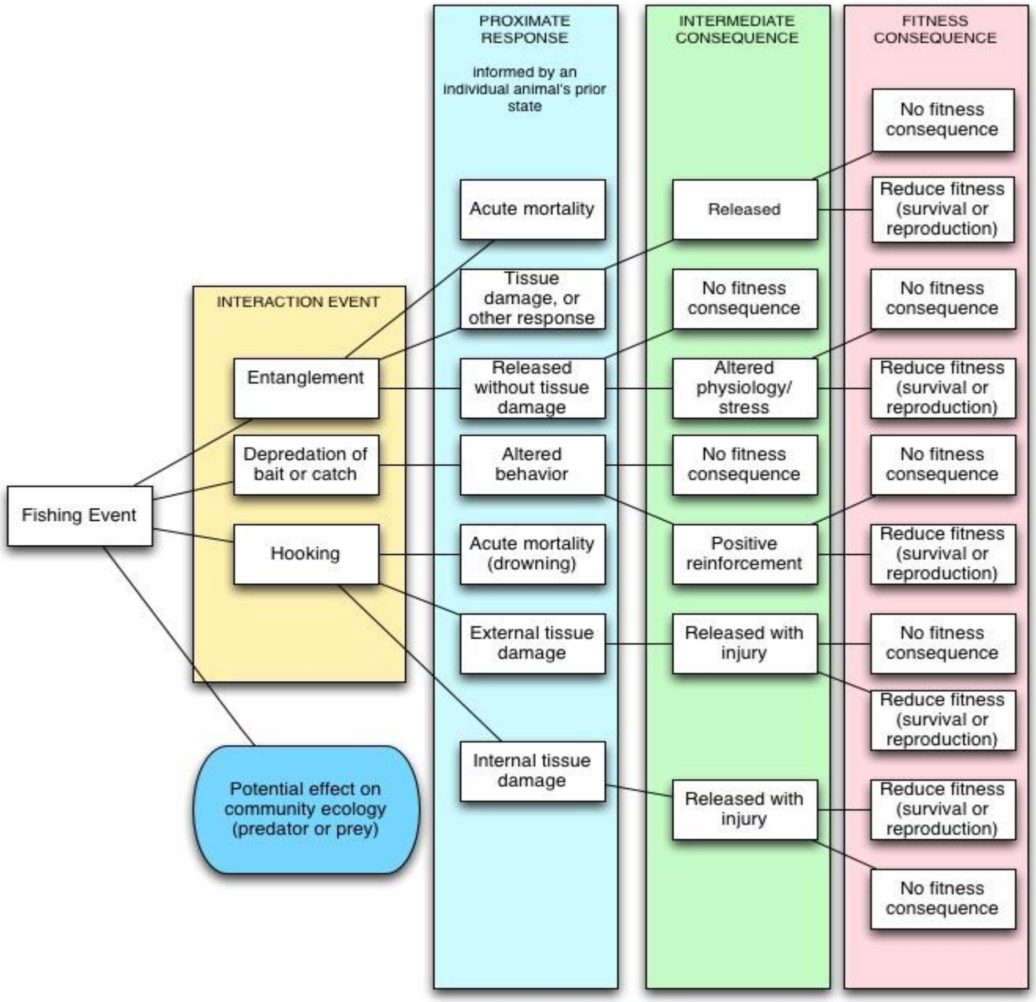


Figure 7. Conceptual model of the potential responses of an animal's exposure in the DSLL fishery and the pathways between exposure (the interaction event the individual animal has with the fishing gear), responses, and potential fitness consequences to the individual.

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 BE (NMFS 2018b) used data from 2004 to 2017 to analyze effects of the action. We now have observed incidental captures in the DSLL through 2020, 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 2020, as these are the years for which we have data and for which incidental captures in the DSLL were separated from incidental 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 incidental captures for 2018-2019. Similarly, for sea turtles, McCracken and Cooper (2021) applied the Bayesian inference model to estimate incidental captures in 2020 based on observed incidental captures but similar data are not available for sharks and rays as of yet. For sharks and rays, we used the expansion factor to estimate incidental captures using the percent observer coverage and observed incidental captures for 2020. From this we have a dataset of estimated incidental captures from 2002 to 2020, however we note again that incidental captures in 2002 and 2003 are representative of both the DSLL and SSLL. In conducting our calculations for the maximum 5-year running average and running sum (see Section 1.6.4), we used the estimated incidental capture numbers from 2004 to 2020. However, we also report the annual mean and 95th percentile values from McCracken (2019a) but caveat that these are based on data from 2002 to 2017.

### 1.4.1 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.

### 1.4.2 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. A few projected global values under RCP 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 2018). 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 2018).

Table 1. Projections for certain climate parameters under Representative Concentration Pathway 8.5 (values from IPCC 2014).

| **Projections** | **Scenarios (Mean and likely range)** | |
|---|---|---|
| | **Years 2046-2065** | **Years 2081-2100** |
| **Global mean surface temperature change (°C)** | 2.0 (1.4-2.6) | 3.7 (2.6-4.8) |
| **Global mean sea level increase (m)** | 0.30 (0.22-0.38) | 0.63 (0.45-0.82) |

Given the limited data available on 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, 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.

### 1.4.3 Statistical Intervals and Anticipated Future Incidental 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 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 in an average year, 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 some years higher or lower numbers of interactions may occur. 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. 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, 2021). 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 2020. 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.

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, 25 years) we rely on largely on the mean estimates because long time series are most likely to include values above and below the mean.

This approach is consistent with feedback that NMFS received during a 2016 public meeting sponsored by the Western Pacific Fishery Management Council specifically to examine NMFS' use of statistical tools in its fishery consultations (WPRFMC 2016). Calculating predictions of interactions and using those predictions in the incidental take statement was part of the discussion at the workshop. Motivation for the workshop was in large part because there were several instances when the fisheries exceeded the exempted take of ESA-listed species described in the incidental take statement. At this meeting it was acknowledged that an incidental take statement that was set too low may have relied on an analytical method that underestimated the impacts of the action on the listed species by under forecasting the interactions between the fishery and the listed species. In turn, this may also increase the frequency at which the fishery exceeded exempted take levels (WPRFMC 2016). To overcome these shortcomings, participants discussed that an upper prediction interval (e.g., 95%) better incorporates uncertainty in the prediction and may be the best way to account for stochasticity in the short-term predictions, whereas the mean might be a better predictor of the long-term effect of the action on the species. We used both values, the mean (including the 5-year running average) and the upper CI, throughout our analysis.

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 2020. As we cannot assume that the anticipated increase in hooks per year will not increase the likelihood of a species' capture, we compare 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 incidental captures of either giant manta rays or oceanic whitetip sharks. 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. 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. 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 beyond this 40-year period, accounting for the future effects of animals killed during the 40 years. 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.

### 1.4.4 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 giant manta rays and oceanic whitetip sharks collected as bycatch between 2004 and 2020 and (2) the Pacific Islands Fisheries Science Center's analyses of these data (McCracken 2019a; McCracken and Cooper 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), the Western Pacific Regional Fishery Management Council's *Stock Assessment and Fishery Evaluation Reports* (WPRFMC 2019, 2020, 2021). 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, 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 giant manta rays and oceanic whitetip sharks.
2. What effects have been reported for giant manta rays and oceanic whitetip sharks 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 giant manta rays and oceanic whitetip sharks 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 giant manta rays and oceanic whitetip sharks 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 the species.

For our literature searches, we used paired combinations of the keywords: *"Manta,"* "*Manta birostris*," "*Mobula*," "*Mobula birostris*," "*Carcharhinus longimanus*," "oceanic whitetip," "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," "habitat loss," "prey availability," "age to maturity," "tag retention." For giant manta rays 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*," 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 giant manta rays and oceanic whitetip sharks recorded as bycatch between 2004 and 2020. 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, data becomes information that is useful for decision-making. The processing of data is typically a multistep operation and involves such steps as organization (e.g., sorting, integration, splitting, restructuring) and analyzing (e.g., validating, duplicating, revising). When raw best available scientific and commercial data is processed it becomes the best available scientific and commercial *information*. 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 information 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 giant manta ray and oceanic whitetip shark (Table 2).

Table 2. Listed resources within the action area that may be affected by the proposed action.

| Species | Scientific Name | ESA Status | Listing Date | Federal Register Reference |
|---|---|---|---|---|
| Oceanic Whitetip Shark | *Carcharhinus longimanus* | Threatened | 1/30/2018 | 83 FR 4153 |
| Giant Manta Ray | *Manta birostris* | Threatened | 02/21/2018 | 83 FR 2916 |

The giant manta ray and oceanic whitetip shark do not have any designated critical habitat at this time; as such, this opinion does not analyze effects to any critical habitat.

## 2.1 Status of Listed Species That Are Likely to be Affected

The rest of this section of our biological opinion consists of narratives for the giant manta ray and oceanic whitetip shark that occurs in the *Action Area* and that may be adversely affected by the Hawaii-based DSLL fishery. 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.

### 2.1.1 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 populations, 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 populations. 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.

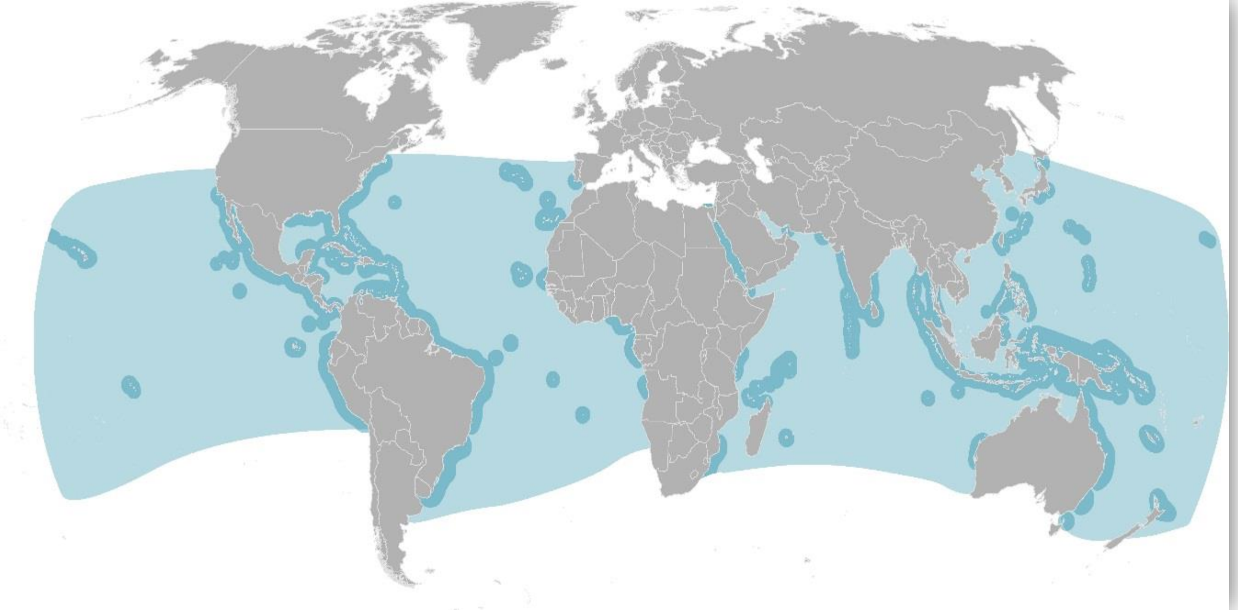


Figure 8. Distribution map for the giant manta ray. Extent of occurrence is depicted by light blue and the area of occupancy is noted in darker blue. (Figure 3 from Lawson et al. 2017).

### 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 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);<br>MantaMatcher (2016) |
| Egypt | 60 | - | Marine Megafauna (2011) as cited in CITES (2013) |
| 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);<br>Sanchez (2016) |
| Isla de la Plata, Ecuador | 2,464 | 25,250 | MantaMatcher (2016);<br>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).

### 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 vessels 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. has issued a proposed rule to put the handling practices in CMM 2019-05 into regulation for U.S. fisheries (86 FR 55790).

### 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.

## 2.1.2 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 9); 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 8 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 9 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 isn't 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.

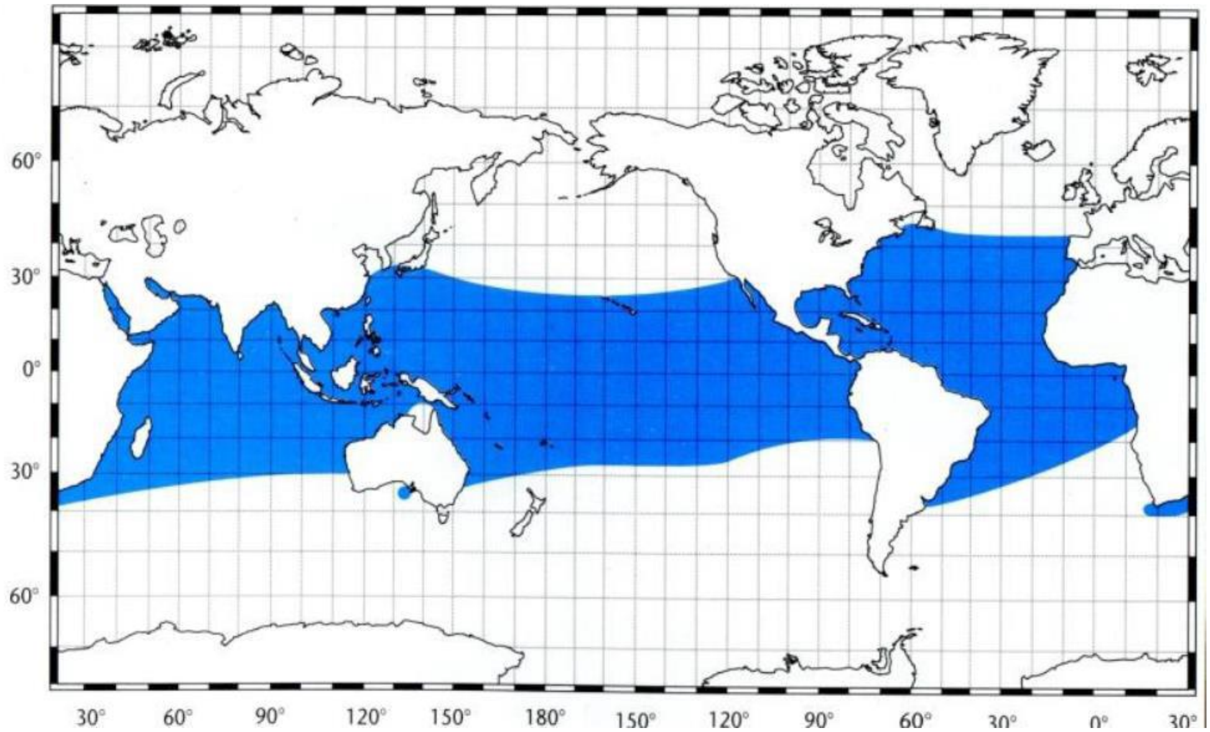


Figure 9. Geographical distribution of the oceanic whitetip shark (Last and Stevens 2009).

### 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 2019) 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 at 775,214) (see NMFS 2020). 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. 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, we focus our analysis on the minimum population size estimate of 775,214, but acknowledge that the total Pacific population size may exceed one million individuals.

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 10; 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). It is unclear if these values will apply to all WCPO longline fisheries, however the implementation of CMM-2019-04 is anticipated to improve the survival of released sharks throughout the WCPO.

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 10). 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.

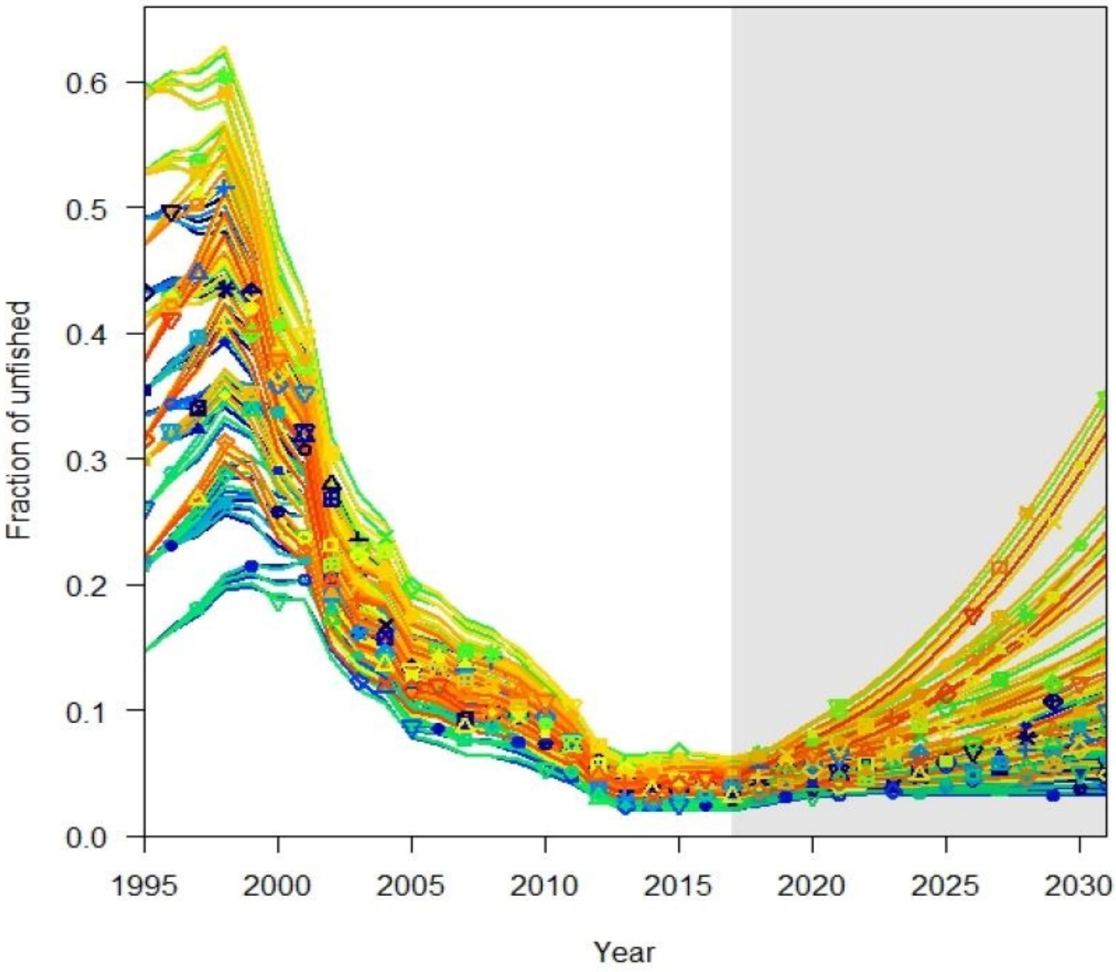


Figure 10. Projected ratios of of spawning biomass (projected to 2031) to the equilibrium unfished spawning biomass for WCPO oceanic whitetip sharks with updated at-vessel and post-release mortality rates and the prohibition of wire branchlines and shark line (Figure 7 in Bigelow et al. 2022).

### 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 (Young and Carlson 2020). 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. 2006). 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 the American Samoa longline fisheries, as well as the U.S. purse seine fishery. The SSLL is estimated to interact with up to 102 oceanic whitetip sharks a year (95th percentile; NMFS 2019). 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 American Samoa longline fishery 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 for all sharks, which retains the retention prohibition for oceanic whitetip sharks, and includes additional measures on minimizing bycatch (including some gear restrictions) and implementing safe release practices.

### *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. 2011; 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.

# 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 giant manta ray and oceanic whitetip shark 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; and pollution and marine debris. These categories of activities and their impact on giant manta rays and oceanic whitetip sharks 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). 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). Marine mammal species often exhibit strong dependence on or fidelity to particular habitat types, oceanographic features, and migration routes (Sequeira et al. 2018). 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 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.

Significant impacts to elasmobranch species 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). Increased ocean acidification may also have serious impacts on fish development and behavior (Raven et al. 2005), including sensory functions (Bignami et al. 2013) and fish larvae behavior that could impact fish populations (Munday et al. 2009) and piscivorous ESA-listed species that rely on those populations for food.

Other climatic aspects, such as extreme weather events, precipitation, ocean acidification and sea level rise also have potential to affect elasmobranch species. Changes in global climatic patterns will likely have profound effects on the coastlines of every continent, thus directly impacting marine species that use these habitats (Wilkinson and Souter 2008).

Because 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 hammerheads 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 giant manta rays and oceanic whitetip shark 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), but fisheries managers have only recently collected 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 the giant manta ray and oceanic whitetip shark.

The *Action Area* overlaps with the management areas of two Regional Fishery Management Organizations (Figure 11): 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 giant manta rays and oceanic whitetip sharks that occur in the *Action Area* have already been summarized in the narrative 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 these 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.

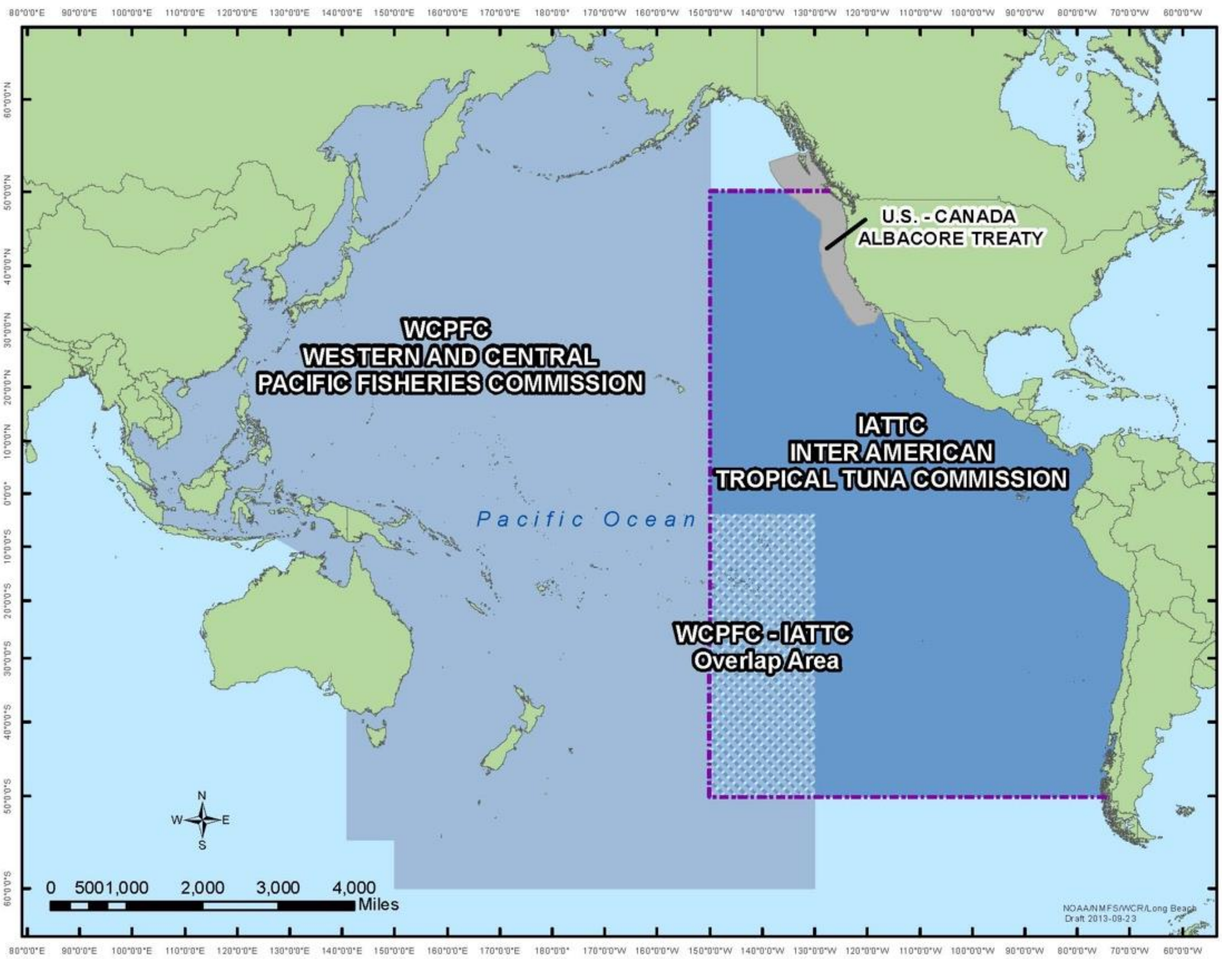


Figure 11. The boundaries of the Western Central Pacific Fisheries Commission (WCPFC) (west of the line) and the Inter-American tropical tuna Commission (IATTC) (east of the line).

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 2021) to assess these trends and compare it to trends in the Hawaii DSLL fishery (Figure 12). 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.

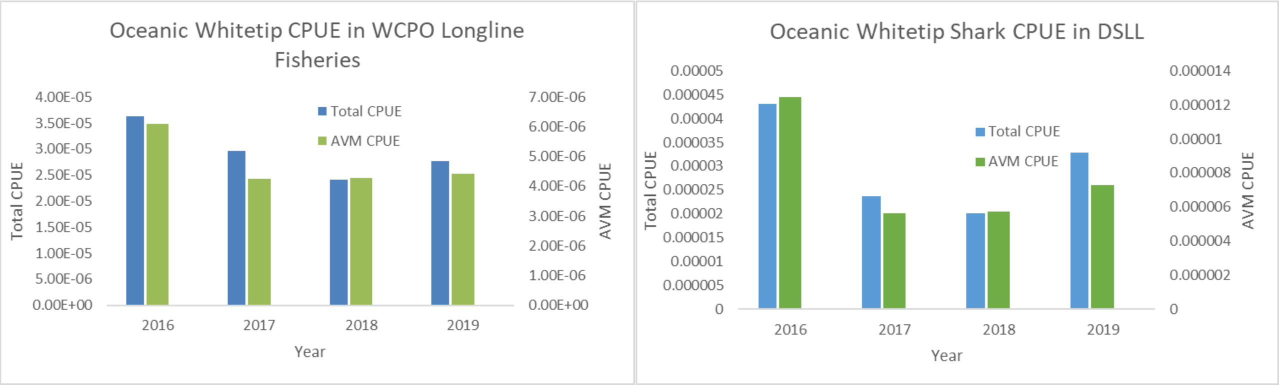


Figure 12. Total observed oceanic whitetip shark catch per observed hook (Total CPUE) and observed at-vessel mortality per observed hook (AVM CPUE) for all WCPO longline fisheries (inclusive of United States; exclusive of fisheries occurring in Vietnam and archipelagic waters of Papua New Guinea, Indonesia and the Philippines; WCPFC 2021; 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 and giant manta ray. However, capturing this number of oceanic whitetip sharks and giant manta ray 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 and giant manta ray.

### 3.2.1 Other Domestic Fisheries in the Action Area

In addition to the Hawaii DSLL fishery, two domestic federal fisheries occur in the *Action Area* – the Hawaii SSLL fishery and the West Coast DSLL fishery. 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. Between 2004 and 2018, 875 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 2019). Projected annual interaction rates are provided in Table 4 for the various species which may also be caught in this (the DSLL) fishery.

Table 4. The estimated number and amount of exempted take of oceanic whitetip shark and giant manta rays, noting annual interactions and associated mortality with various species in the Hawaii SSLL fishery (NMFS 2019).

| **Species** | **Annual** | |
|---|---|---|
| | **Number Captured** | **Total Number Killed** |
| Oceanic whitetip shark | 102 | 32 |
| Giant manta ray | 13 | 4 |

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. 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).

In addition to the federal fisheries, Hawaii troll, handline, and shortline fisheries overlap with the portion of the action area where DSLL vessels transit to and from Honolulu Harbor. They do not overlap where DSLL fishing occurs. These fisheries are known to capture oceanic whitetip sharks (e.g. Hawaii State Department of Land and Natural Resources. 2019)

### 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, 2017). As Table 5 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. The same is true of the 14 to 15 years.

Table 5. The number of active vessels and fishing effort in the Hawaii DSLL fishery, 2004-2020 (includes effort in both WCPO and EPO). Blue shading denotes the lowest values in the table, yellow shading denotes the highest values (NMFS 2018a).

| 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 |
| 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 | n.d. | 22,192 | 62,700,000 |

As shown in Figure 13, active participation in the deep-set fishery since 2004 has remained below 164, but has been increasing since 2010.

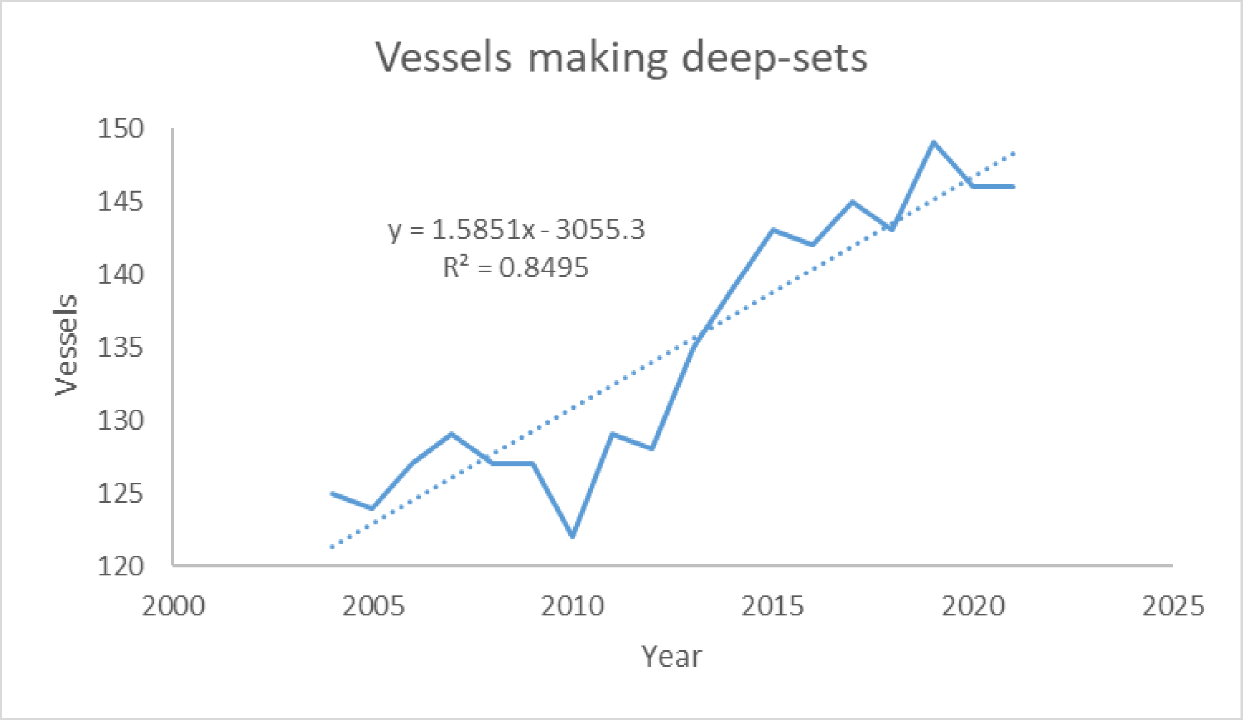


Figure 13. Summary of effort for the Hawaii DSLL fishery since 2004. See Table 5 for the data (data from NMFS 2018a and pers. comm. J. Makaiau to A. Garrett, May 19, 2022).

NMFS (2018a) 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 5). 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 2017), 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 14), and while this increase is expected to continue into the near future (pers comm J. Makaiau 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.

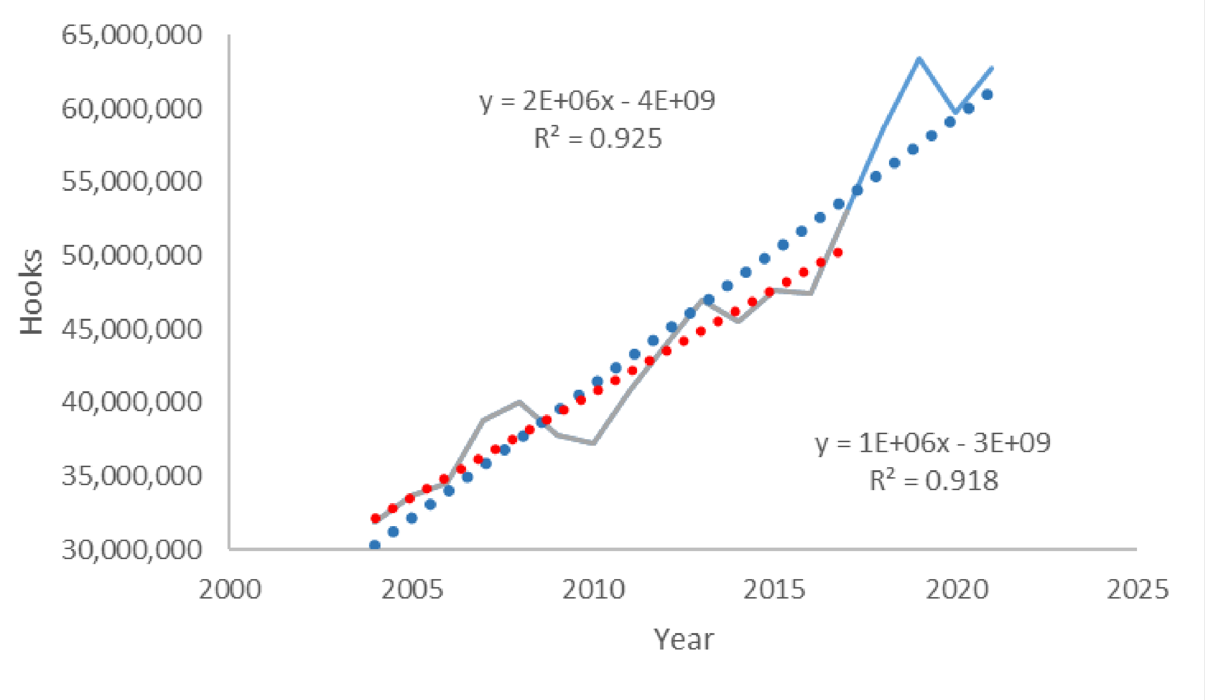


Figure 14. Hooks per year deployed by the Hawaii DSLL fishery (gray line). Orange dotted line and lower regression equation represents the relationship anticipated by NMFS (2018a); blue dotted line and upper regression equation represents the current rate of increase in hooks per year.

Part of the driver for the increase in hooks per year is obviously from the increase in vessels participating in the Hawaii DSLL fishery in recent years, however, the number of hooks per vessel has also been increasing over time as well (Figure 15). Drilling down further, trips per vessel per year has not changed substantially over time, although there has been a marked increase since 2016 (Figure 15). Similarly, sets per vessel per year mirrors and amplifies the increase in trips per vessel noted since 2016 (Figure 15). Finally, hooks per set has increased over time. 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); 2) increasing trips and sets per vessel since 2016; and 3) increased numbers of hooks deployed per set. There are no regulatory limits (although there are likely logistical limits) to the second two factors impacting hooks per year and therefore, for the purpose of our analysis, we assume that the linear increase in number of hooks per year will continue into the future (Figure 16). This relationship suggests an annual increase of ~1,815,000 hooks per year (95% CI: 1,540,671 to 2,088,479). 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.

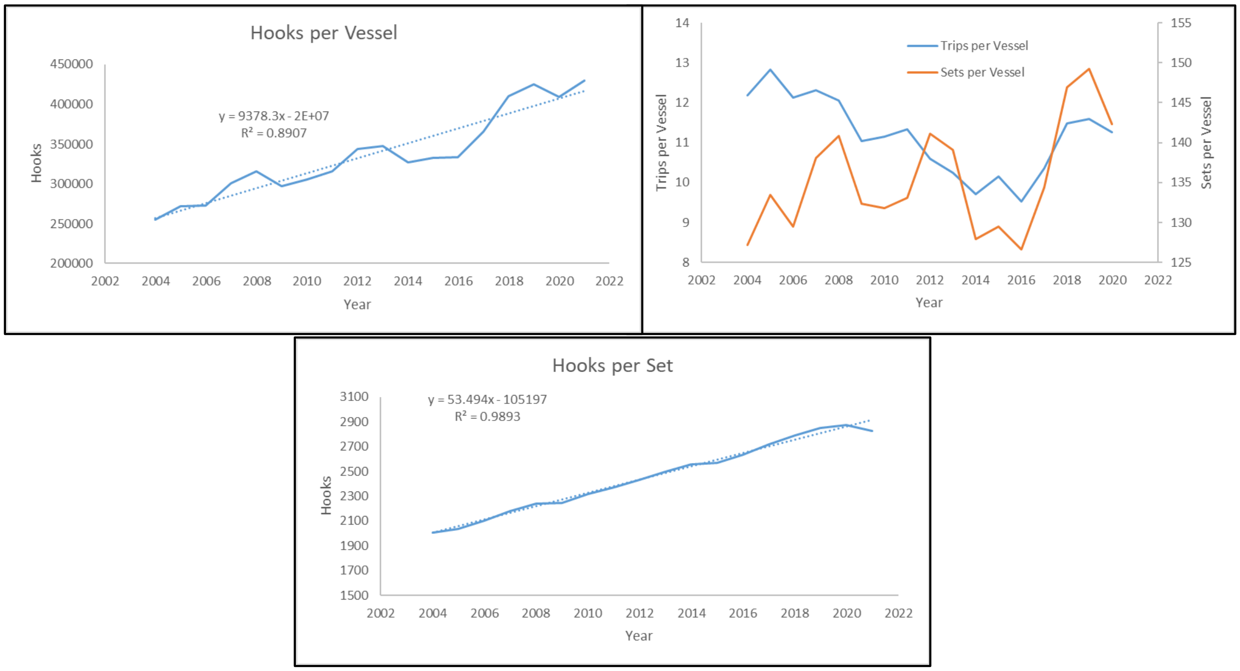


Figure 15. Effort metrics for the Hawaii DSLL fishery. Top left shows the increase in hooks per vessel from 2004 to 2021, dotted line represents a linear regression that is described by the equation. Top right shows the trends in trips per vessel (blue line) and sets per vessel (orange line) from 2004 to 2020. Bottom shows hooks per set from 2004 to 2021, dotted line represents a linear regression that is described by the equation.

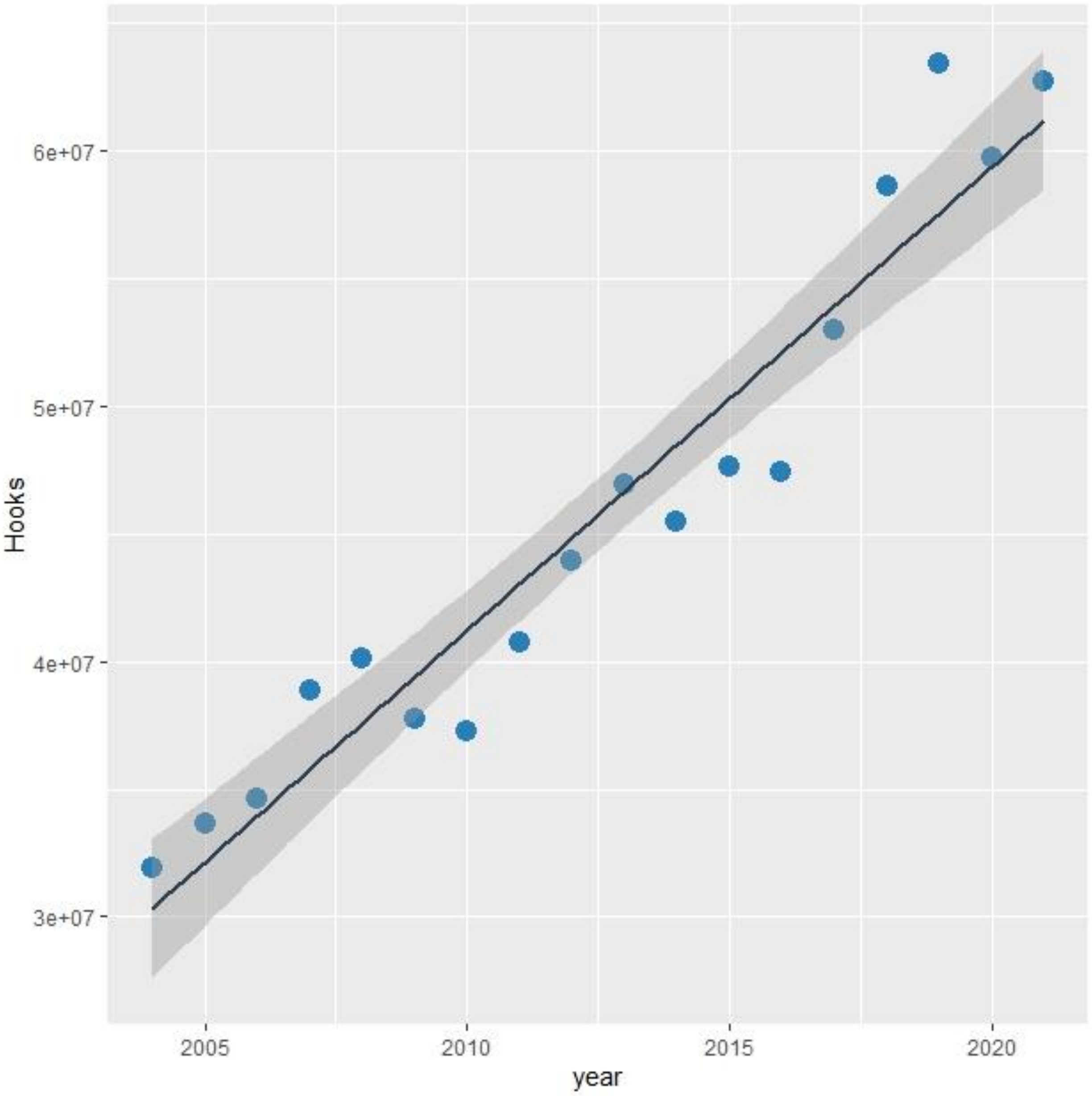


Figure 16. Trend in annual hooks per year for the DSLL from 2004 to 2021. Blue dots are the number of hooks in a given year, black line is a linear regression through the data, and the gray shaded area represents the 95% CI for the linear regression.

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.[2]

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 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[3]. 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, 2017).

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).

To summarize the historic impact of the DSLL, between 2004 and 2020, 45 giant manta rays were incidentally captured with an estimated 305 total and 5,149 oceanic whitetip sharks were observed, with an estimated 26,180 sharks incidentally captured (McCracken 2019a; McCracken and Cooper 2020a, 2020b; NMFS 2018b).The effect of this continuing action on giant manta

---

[2] 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.

[3] 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.

rays and oceanic whitetip sharks is addressed in the *Effects of the Action* section of this biological opinion in greater detail.

## 3.3 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.

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).

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 shark or giant manta ray mortalities resulting from marine debris in the *Action Area*.

It is difficult to assess the precise impact pollution and marine debris have had on oceanic whitetip sharks and giant manta ray. However, we are reasonably certain they have contributed to the declines in oceanic whitetip shark and giant manta ray.

### 3.4 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 the impact of those activities on the status, trend or the demographic processes of threatened and endangered species is largely unknown, some 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 some of the most serious and lasting effects 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.

## 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 giant manta rays and oceanic whitetip sharks. 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 giant manta rays and oceanic whitetip sharks. If hooked or entangled, may not immediately die from their wounds 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 oceanic whitetip shark, and giant manta ray populations is very difficult. Even if giant manta rays and oceanic whitetip sharks 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 oceanic whitetip sharks are likely to respond to these interactions with fishing gear.

Despite several efforts to assess the significance of unobserved or slipped catch, the number of unobserved interactions (for example, Moyes et al. 2006; Murray 2011; and Warden and Murray 2011; Gilman et al. 2013), and the difference between the number of observed interactions and the actual number of interactions remains unknown. Some species have a better opportunity to escape capture before being observed by the vessel by breaking the line either through sheer force or by biting the line.

### 4.2.1 Entanglement in Longline Gear

#### *Giant Manta Ray*

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 and applicable which is a reasonable surrogate species as, since prior to 2009, the two manta species were categorized as one a single species, giant manta ray. Surveys of the reef manta from 2005-2009 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). 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.

#### *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.

### 4.2.2 Hooking

#### 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 (Bertolini 1933; 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, fishers 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.

#### Oceanic Whitetip 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 that can last an average 10-12 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.

### 4.2.3 Trailing Gear (Line)

#### Giant Manta Ray

Given their size, giant manta rays are seldom boarded, and similar to leatherback sea turtles, observers and fishers are instructed to cut the line as close the hook as possible. 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.

#### Oceanic Whitetip 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, NMFS has implemented regulations (50 CFR 300.226) 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.

## 4.3 Species-Specific Analyses

The narratives that follow present our exposure and response analyses for giant manta rays and oceanic whitetip sharks that we previously concluded are likely to be adversely affected by the fishery. For our exposure analyses we try to estimate the number, age (or life stage), and gender of the individuals that are likely to be exposed to stressors associated with the fishery. Because almost all interactions between the fishery and imperiled species occur beneath the ocean's surface where the interactions are unobserved, as a result, we cannot estimate the actual number of exposures (for example, some animals escape before they are observed, and some individuals may interact more than once). Instead, we treat the number of animals reported as bycatch as a proxy for the number of individuals that are exposed to the fishery.

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. 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.

As noted in Section 3 – *Environmental Baseline*, there is an increasing trend in vessels, trips and sets per vessel, and hooks per set for the Hawaii DSLL fishery. While there is an upper limit on the number of vessels that can fish at any given 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 16). 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 did not find statistically significant relationships between observed hooks and observed incidental captures for giant manta rays or oceanic whitetip sharks. $R^2$ was less than 0.05 indicating that less than 5% of the variability in observed incidental captures per year is explained by the number of hooks. We acknowledge that with the limited data available (17 years) and high year-to-year variability in observed incidental captures, there may be a more significant relationship between hooks and incidental captures than we are currently able to detect for giant manta rays and oceanic whitetip sharks and we assert that this will need to carefully monitored as the fishery effort continues to increase into the future.

For our response analyses, we try to assess how animals respond after being captured in DSLL gear. For the most part these analyses are informed by published and 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.

*Interpretation of the intervals presented in these analyses*. The narrative that follows this introduction presents and discusses intervals: the exposure estimates present mean values with 95th percentiles (all from McCracken 2019a; McCracken and Cooper 2020a, 2020b) 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 are reasonably likely to occur although the point estimate (in our case, the mean value) is the value that best describes anticipated effects over a longer term.

For example, Table 12 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 12 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 1863.2 oceanic whitetip sharks with up to 9,316 over 5 years."

### 4.3.1 Giant Manta Ray

#### Exposure Analysis

Giant manta ray have been incidentally captured in the DSLL fishery in 15 of the 17 years between 2004 and 2020. During this 17-year time interval fishery observers reported that 45 giant manta ray were incidentally captured in the DSLL fishery. An additional 68 unidentified *Mobulidae* and 23 unidentified rays were incidentally captured over that timeframe. 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 2020 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 6). 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 2004 to 2020 and used the Wilson Score Method to estimate the 95% confidence interval around the proportion of giant manta ray (Table 7). We multiplied that upper 95% CI by the number of *Mobulidae* spp. and unidentified rays that McCracken (2019a) and McCracken and Cooper (2020a, 2020b) estimated were incidentally captured each year from 2004 to 2020 to estimate the proportions of each unidentified category that were likely to have been giant manta ray. In total, we estimate from 2004-2020, the DSLL fishery had 59 observed interactions with giant manta rays (45 estimated giant manta ray plus 14 from the *Mobulidae* spp. and 0 from the unidentified ray categories).

Table 6. The proportion of estimated incidental captures identified as giant manta ray and *Mobula* (devil rays) from 2004 to 2020 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 | 45 | 248 | 293 | 0.15 | 0.85 | 0.12 | 0.20 |
| *Mobula* (Devil Rays) | 248 | 45 | 293 | 0.85 | 0.15 | 0.80 | 0.88 |

Table 7. The proportion of estimated incidental captures identified as giant manta ray, *Mobula* (devil rays), and pelagic stingrays from 2017 to 2019 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 | 2.00 | 6,597.00 | 6,599.00 | 0.00 | 1.00 | 0.00 | 0.00 |
| *Mobula* (Devil Rays) | 38.00 | 6,561.00 | 6,599.00 | 0.01 | 0.99 | 0.00 | 0.01 |
| Pelagic Stingrays | 6,559.00 | 40.00 | 6,599.00 | 0.99 | 0.01 | 0.99 | 1.00 |

We used the same ratios of confirmed giant manta rays, mobula (devil ray), and pelagic sting rays to estimate the number of giant manta incidentally captured based on observed incidental captures. An estimated 228 giant manta ray and 388 unidentified *Mobulidae* spp. were incidentally captured from 2004 to 2020 (McCracken 2019a; McCracken and Cooper 2020a, 2020b). Using the upper 95th CI from

Table 6, we estimate that 20% of the latter may have been giant manta ray, or 77 individuals. Therefore about 228+77=305 giant manta ray were likely to have been incidentally captured in the fishery over the 17-year period from 2004 to 2020 (McCracken 2019a; McCracken and Cooper 2020a, 2020b).

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. 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 17 and Figure 18. 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 18).

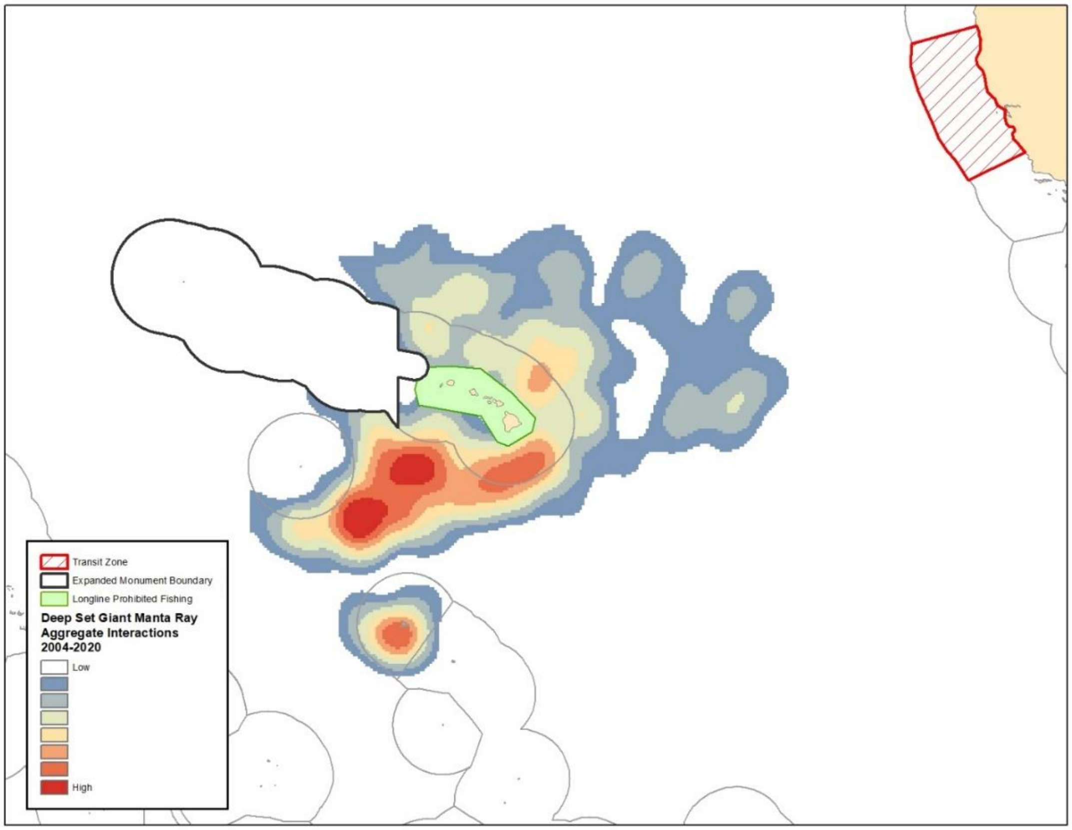


Figure 17. Geographical representation of observed aggregate giant manta ray and unidentified *Manta/Mobula* and ray interactions with the DSLL fishery between 2004 and 2018 (*n* = 312).

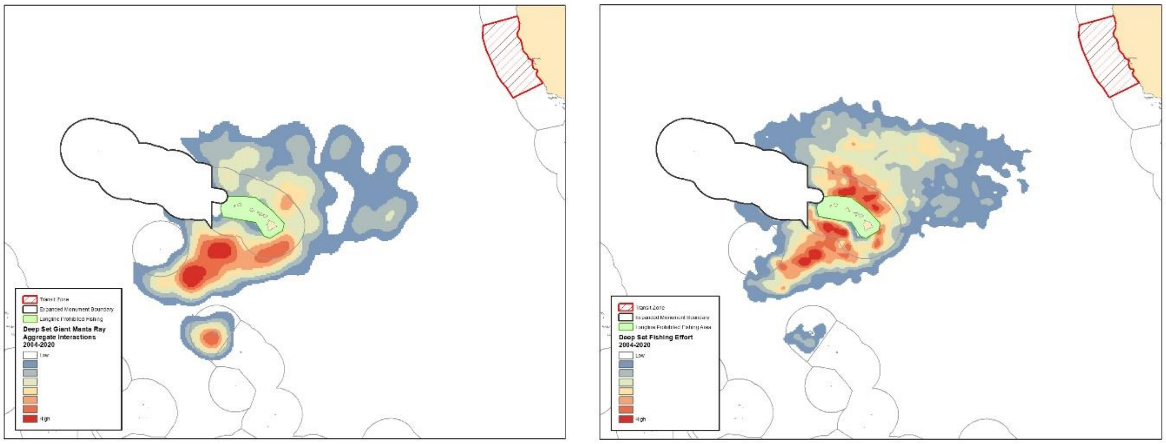


Figure 18. Heatmap comparison of observed giant manta ray and unidentified *Manta/Mobula* and ray interactions (left; *n* = 312) versus aggregated fishing effort during this same time frame (right; *n*= 65,245 from 2004 to 2018).

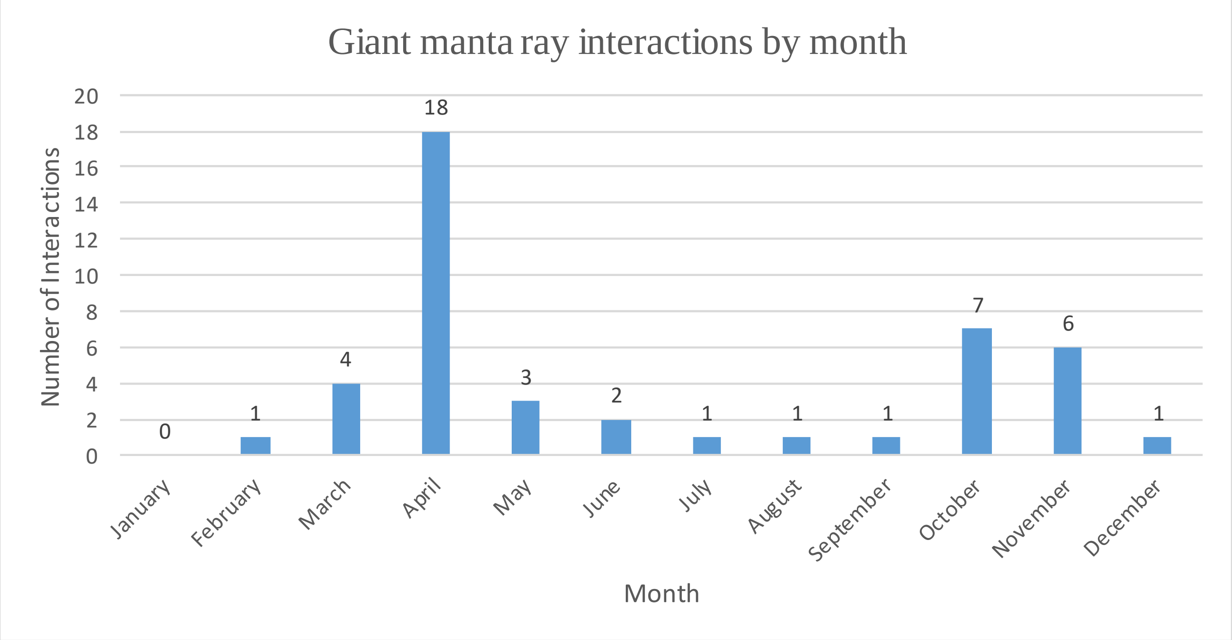


Figure 19. Numbers of observed giant manta ray incidentally captured per month in the DSLL fishery from 2004 to 2020.

Assessing aggregate annual observed interactions, 40% occurred in the month of April (Figure 19) 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 20). The incidental captures in 2010 were more than 4 times higher than observed or estimated incidental captures in any other year.

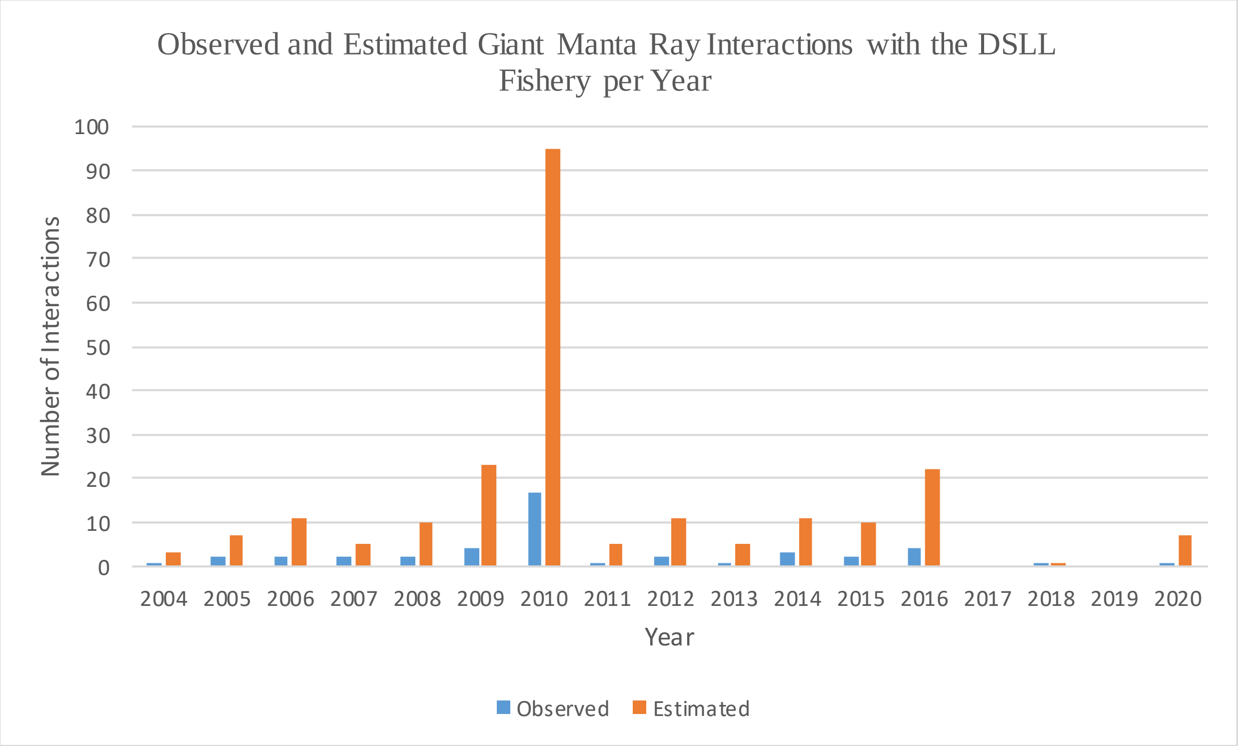


Figure 20. Giant manta ray interactions per year in the DSLL fishery between 2004 and 2020 for observed interactions (blue bars) and estimated interactions (orange bars; from McCracken 2019a and McCracken and Cooper 2020a, 2020b).

### *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 8). We also report the anticipated maximum mean number of annual mortalities associated with the 5-yr running average and the maximum cumulative number of anticipated mortalities over 5 years that have been prorated for unidentified categories (Table 8).

Table 8. Anticipated annual and 95th percentile (from McCracken 2019a and McCracken and Cooper 2020a, 2020b; 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. Values in parentheses are the numbers anticipated inclusive of unidentified *Manta/Mobula*.

| | **DSLL Encounters** | |
|---|---|---|
| **Rate** | **Mean** | **95th Percentile** |
| **Annual** | 20 (23) | 60 (70) |
| **Maximum 5-Yr Running Average** | 28.8 (35.8) | N/A |

| | **DSLL Encounters** | |
|---|---|---|
| **5-Yr Running Sum** | 144 (179) | N/A |

### Response Analysis

One of the 45 giant manta rays observed captured in the DSLL fishery was dead at capture resulting in a mean at vessel mortality rate of 2.22% (95% CI 0.39, 11.6). 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 2019).

As described above, there is available information on 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. 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 2019) 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 unentangled 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 29 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 9 details the resulting post-interaction and total (inclusive of at-vessel mortality) mortality rates. To be precautionary, we used the highest estimate of mean total mortality (0.432) in the risk analysis below but note this value is lower than the current estimate of 67.4% (mean across scenarios) to 100% assumed by Griffiths and Lezama-Ochoa (2021) for spinetail devil rays. 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 9. At-vessel, post-interaction, and total mortality rate estimates for giant manta ray interacting with the DSLL fishery from 2004 to 2020. 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** | 0.270 | 0.419 | 0.341 |
| **L95%** | 0.164 | 0.287 | 0.224 |
| **U95%** | 0.422 | 0.570 | 0.498 |

| | **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** |
|---|---|---|---|
| **At-vessel mortality rate** | 0.022 | 0.022 | 0.022 |
| **L95%** | 0.004 | 0.004 | 0.004 |
| **U95%** | 0.118 | 0.118 | 0.118 |
| **Total Mean Mortality Rate** | 0.287 | 0.432 | 0.356 |

### Risk Analysis

On average, we expect that the Hawaii DSLL fishery will interact with 23 (95th percentile= 70) giant manta ray each year and of those 9 (95th percentile= 26) 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 35.8 giant manta ray and we anticipate that up to 179 giant manta ray may be incidentally captured over 5 years. Of those, 16 (5-year running average) and 78 (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 not exceed the 95th percentile of 70 and the cumulative incidental captures over 5-years is not expected to exceed 178 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 1,432 incidental captures and we would anticipate that 623 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 10). 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 21 A&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 21C).

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 21), a circle with a minimum diameter of approximate 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 fishery incidental 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 incidental 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 10 (Figure 22) 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 2004-2020 incidental captures impacting the individual subpopulations (Table 10). 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 2020 to get the proportion of total captures expected per subpopulation. We then multiplied these proportions by the future anticipated annual incidental captures (Table 8) to get the number of annual incidental captures expected per subpopulations (Table 10).

Table 10. 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**<br>**Maximum 5-yr RA (95th percentile)** |
|---|---|---|---|---|
| Hawaii | 21.013 | -157.635 | 13 | 11 (21) |
| Johnston Atoll | 13.174 | -165.825 | 12 | 10 (19) |
| Palmyra Atoll | 6.446 | -162.358 | 23 | 19 (37) |

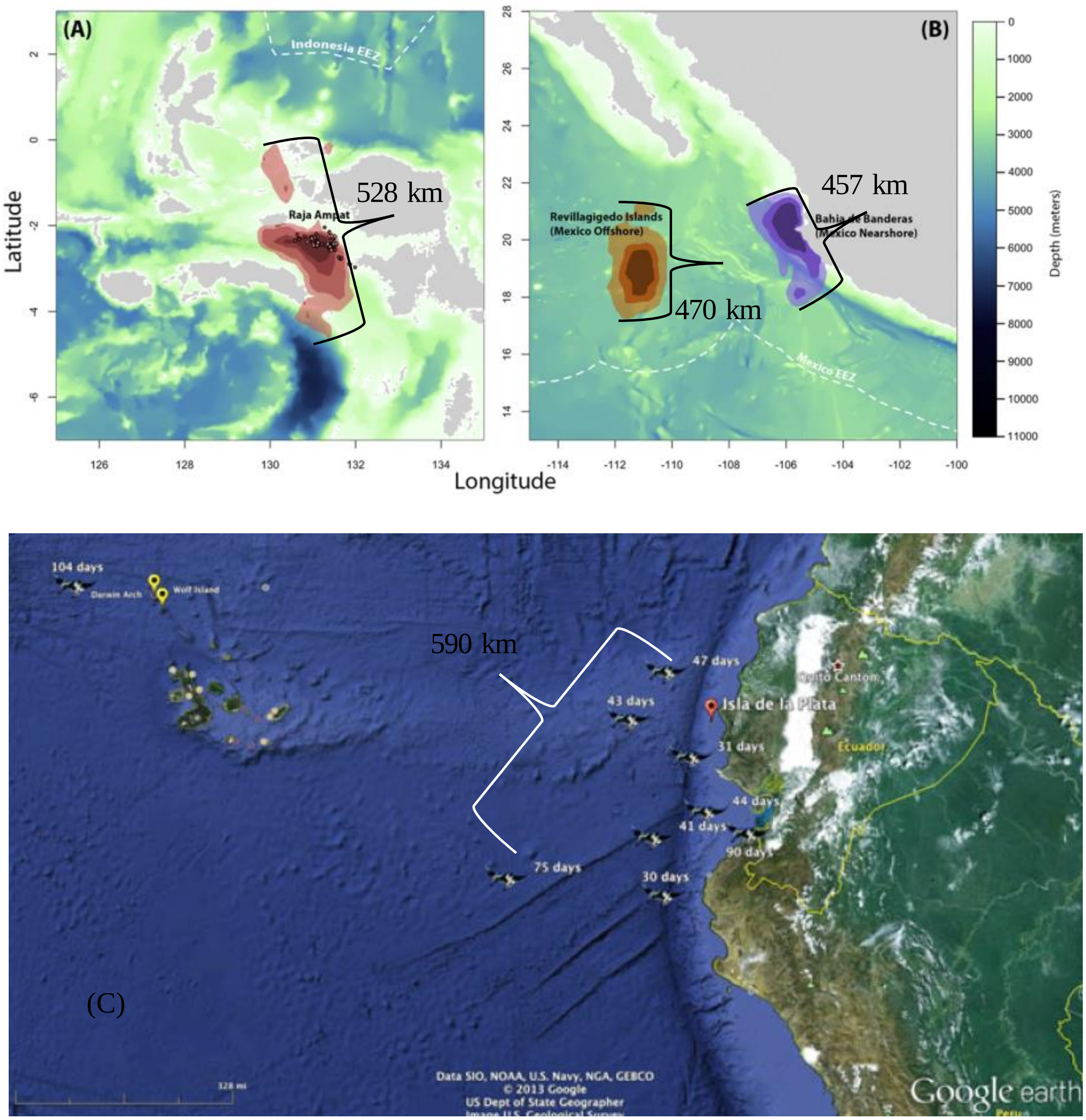


Figure 21. Extent of giant manta ray subpopulations based on tracking tagged individuals. Black brackets (white brackets in C) highlight the longest dimension of the area of occupancy with the length noted in kilometers (km). Locations: A) Raja Ampat, Indonesia (Stewart et al. 2016a); B) Revillagigedo 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 there are some concerns over the methodology because the density dependent assumptions used 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[4]. 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

---

[4] 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*, 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; Table 8) multiplied by the mean estimated post-interaction mortality rate of 0.432 (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 a worst-case analysis for the approximate maximum number of interactions we anticipate will occur in any given year.

We anticipate that misidentification of giant manta ray, in that individuals identified as giant manta ray may be different species, is an issue with the Hawaii-based DSLL fishery given that there is no observer category for reef manta rays, making it likely that at least a portion of individuals identified as giant manta rays may be reef manta rays. SFD has a high level of confidence that individual identified giant manta ray are manta ray 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 have no information as to the magnitude of any potential misidentification rate for this fishery, hence, until data can be obtained regarding this issue, we assume all giant manta rays observed incidentally captured were correctly identified.

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 11).
>
> Scenario 2: three discrete populations; 44% post-release mortality rate; 95th percentile of anticipated future annual interactions; 10, 20, and 40 years (Table 11).

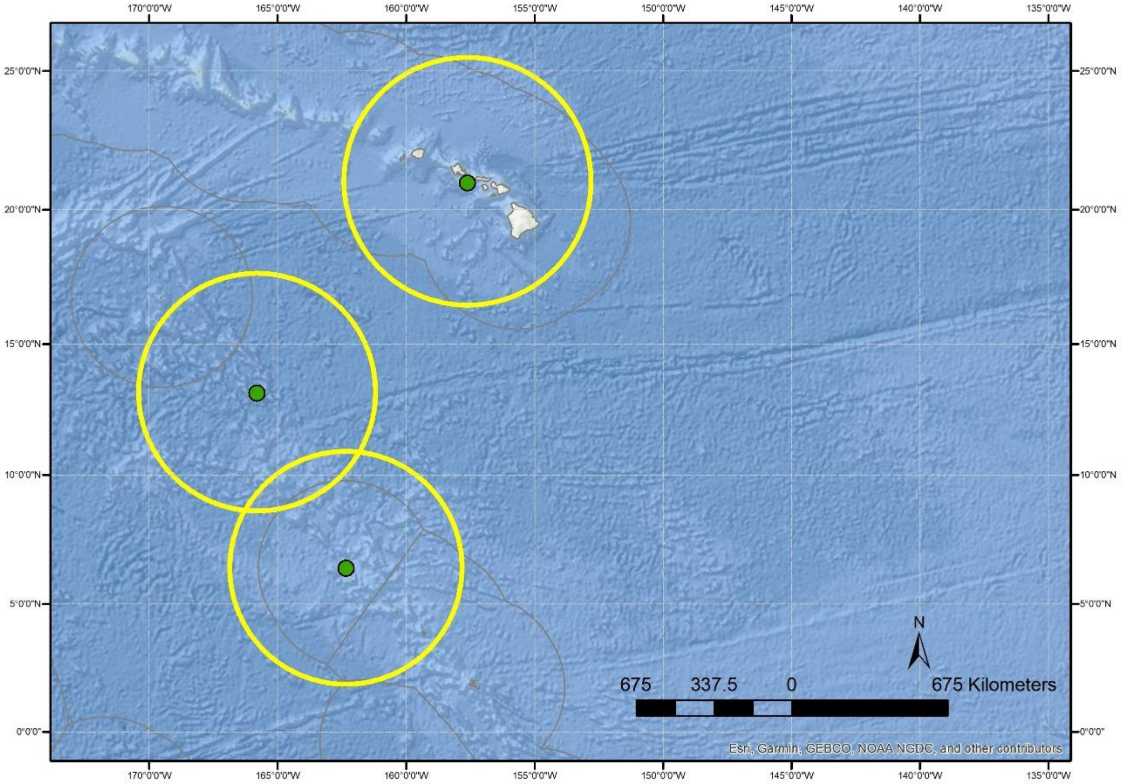


Figure 22. Map of fishing area for the DSLL fishery. Small green circles are the center points of the subpopulations in Table 10. Yellow circles represent an area around the subpopulation center points with a radius of 500 km.

Table 11. 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 $r_{max} = 0.019$, subpopulations with starting population sizes of 400 or less decline while all larger population sizes have increasing population trajectories. If $r_{max} = 0.042$, the Johnston Atoll subpopulation has an increasing population trajectory if starting population sizes are larger than 100; for Palmyra and Hawaii, increasing population trajectories occur with starting population sizes are greater than 200 (Figure 23; Appendix B). Figure 23 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 1,000 (Figure 24; Appendix B). If $r_{max} = 0.042$, the Hawaii and Palmyra subpopulations have increasing population trajectories if starting population sizes are larger than 200; for Johnston Atoll, increasing population trajectories occur with starting population sizes are greater than 100. Figure 24 summarizes the impact of the intrinsic rate of subpopulation growth on the three subpopulations for Scenario 2.

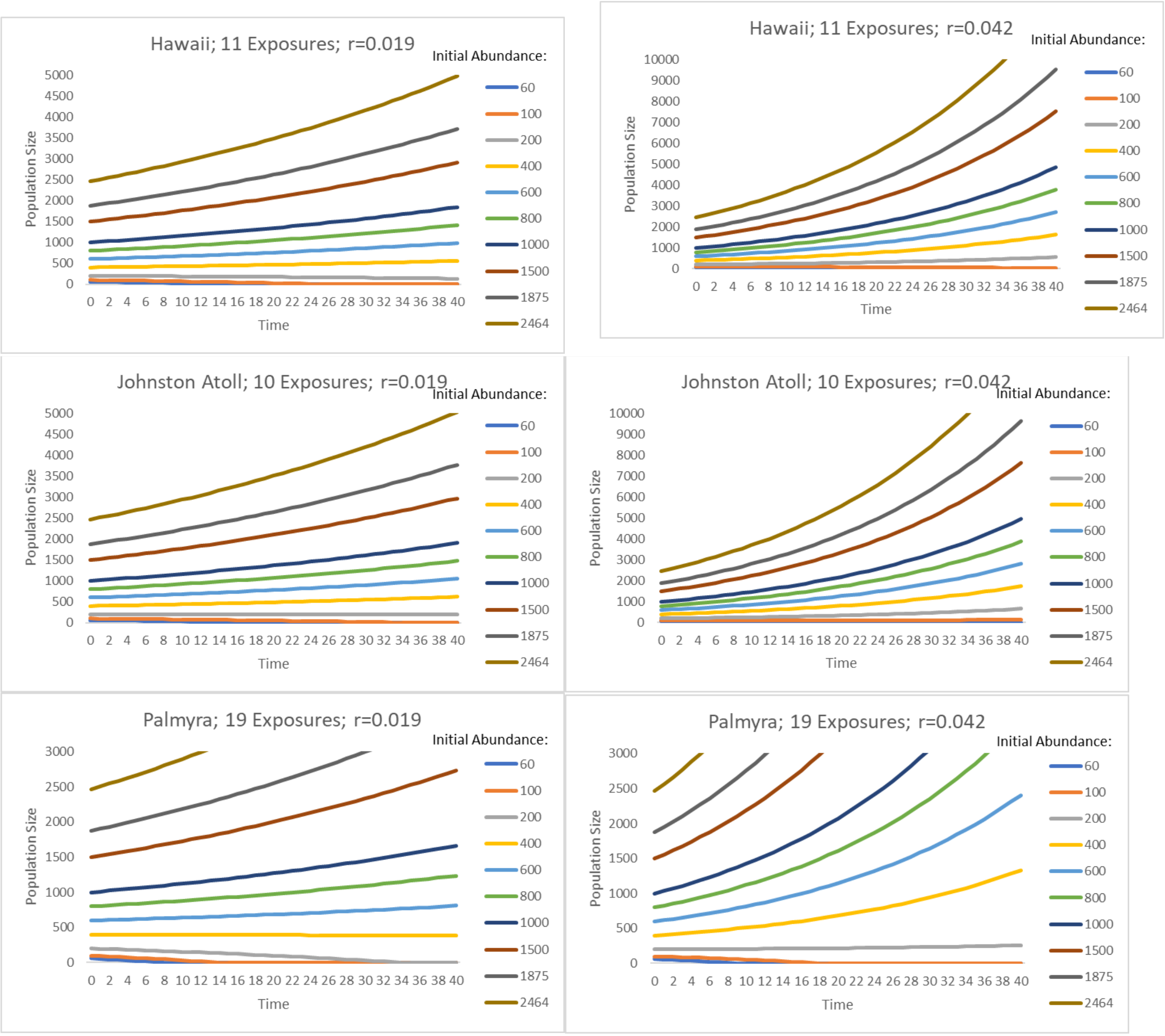


Figure 23. Results of Scenario 1 (maximum 5-year running average) for the Hawaii, Johnston Atoll and Palmyra Atoll giant manta ray subpopulations. Each graph shows population trajectories for the range of initial abundances (60 to 2,464).

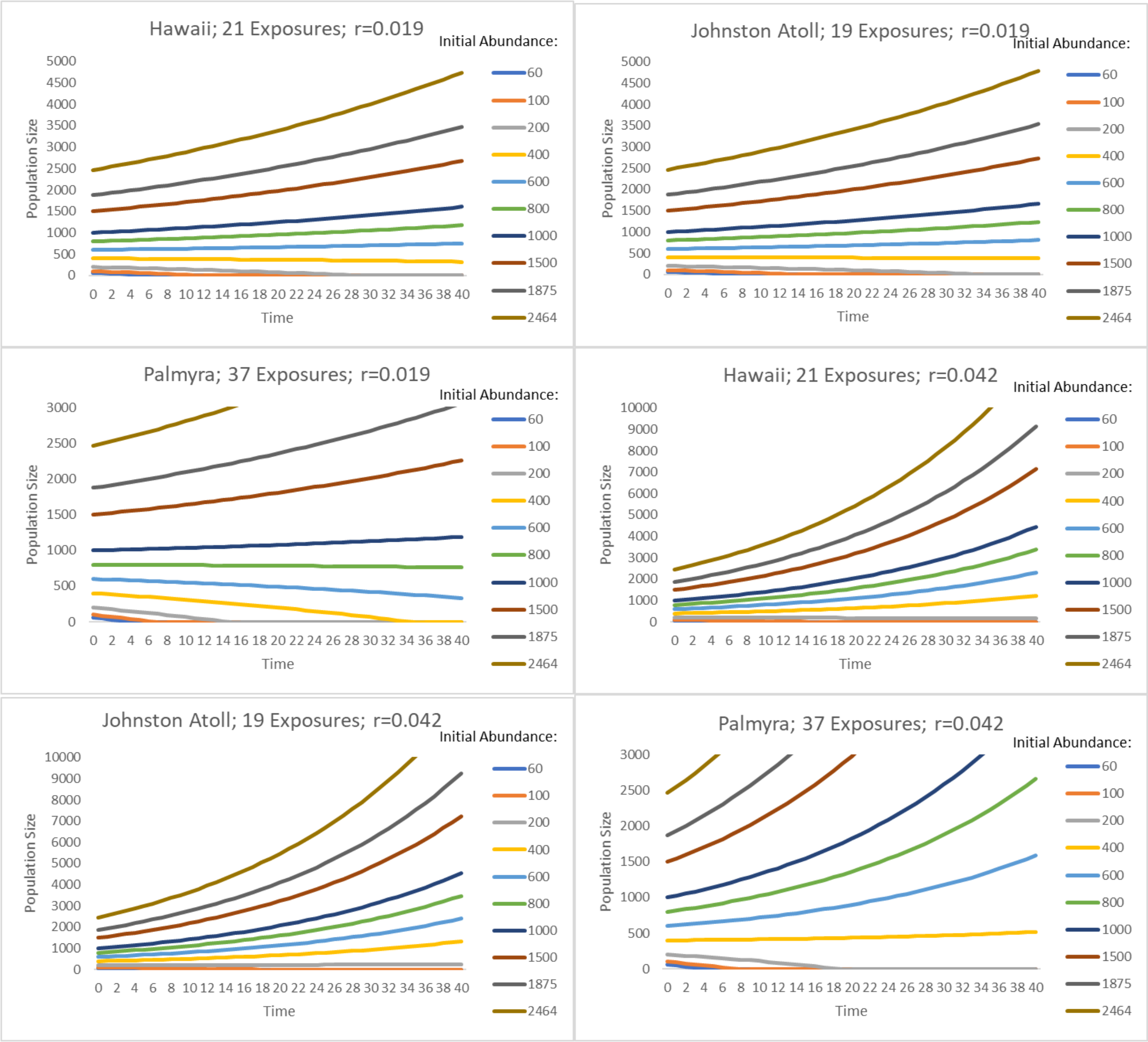


Figure 24. Results of Scenario 2 (95th percentile take levels) for the Hawaii, Johnston Atoll, and Palmyra Atoll giant manta ray subpopulations. Each graph shows population trajectories for the range of initial abundances (60 to 2,464).

The data available are not sufficient 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% in significant 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 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 2020, there have been 5,149 *observed* interactions in the DSLL fishery. When we adjusted these numbers to account for the percentage of observer coverage, approximately 26,180 oceanic whitetip sharks are likely to have been incidentally captured in the Hawaii DSLL fishery between 2004 and 2020 (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021).

We expect that the majority of sharks will be hooked rather than entangled (NMFS 2018a). Table 12 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 are from McCracken 2019a; McCracken and Cooper 2020a, 2020b; NMFS 2021b). 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% (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% of interactions are *observable*.

The estimates we present in Table 12 for the number of individual oceanic whitetip sharks are based on historical observed interactions and the use of wire leaders, and we also present our best estimate under a monofilament leader only fishery assuming a 32% reduction in *observable* interactions (numbers in parentheses; Bigelow and Carvalho 2021). Note that we anticipate the fishery to continue to interact with the same number of sharks (i.e. a mean of 1,708 per year), however with the switch to monofilament leaders, we anticipate that only 68% of these interactions will be brought to the vessel and therefore be *observable*. The remaining 32% of sharks are still anticipated to be incidentally captured by the fishery but will be *unobservable* as they are expected to bite off the line and free themselves. Although evidence suggests that the number of hooks deployed in the Hawaii DSLL fishery has been increasing overtime, we looked but could not detect a meaningful relationship between fishing effort (e.g., number of hooks in the water) and oceanic whitetip shark incidental captures. Therefore, the estimates we present in Table 12 represent our best estimates of the total number of oceanic whitetip sharks that we expect will be exposed to (incidentally captured in) the Hawaii DSLL fishery.

Table 12. 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; NMFS 2021b) and the maximum 5-year running average and cumulative sum. Numbers in parentheses represent the *observable* portion of the interactions with the fishery switching to monofilament leaders.

| **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)** | | |
|---|---|---|
| **Rate** | **Mean** | **95th Percentile** |
| **Annual** | 1,708 (1,161) | 3,185 (2,166) |
| **5-Year Running Average** | 1,863.2 (1,266.3) | N/A |
| **5-Year Cumulative Sum** | 9,316 (6,335) | N/A |

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 2020 (Figure 25). 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 54% of observed oceanic whitetip sharks were incidentally captured in April and May (~22%) and October through December (~32%) from 2004 to 2020 fishery wide.

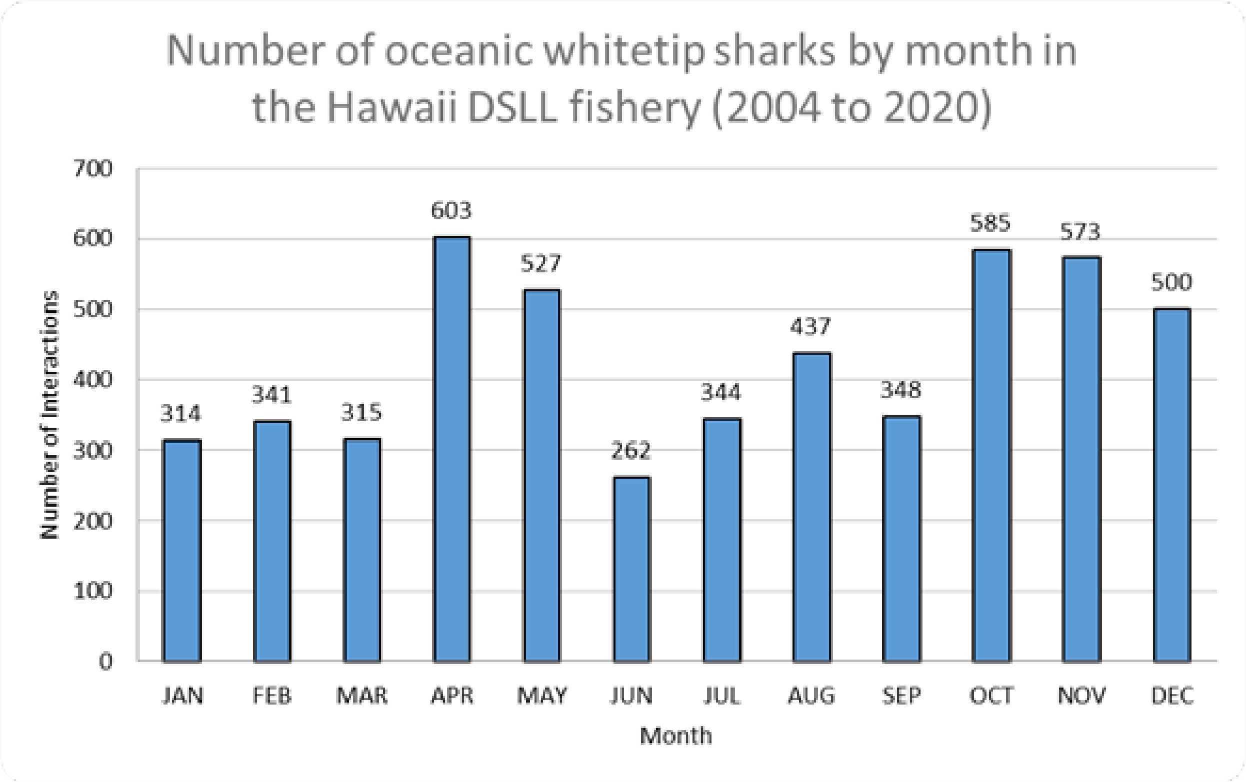


Figure 25. Number of observed oceanic whitetip sharks incidentally captured by month in the Hawaii DSLL fishery from 2004 to 2020 (*n* = 5,149).

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 26. Based on the raw data, observed interactions peaked in 2015; however, due to interannual variability in percent observer coverage, the estimated number of interactions peaked in 2004 and 2015, with estimates of interactions exceeding 2,000 oceanic whitetip individuals in these and three other years-2016, 2019, and 2020 (Figure 26).

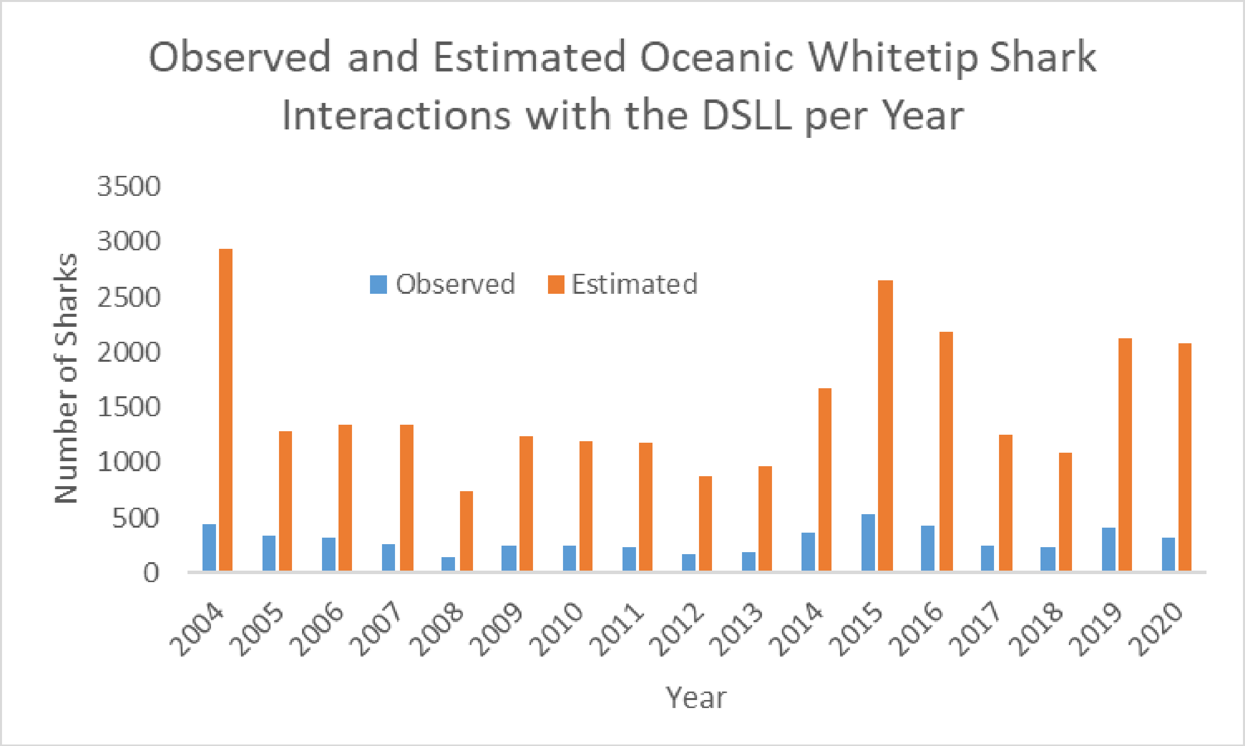


Figure 26. Oceanic whitetip shark interactions per year in the DSLL fishery between 2004 and 2020 for observed interactions (blue bars) and estimated interactions adjusted for observer coverage (orange bars; McCracken 2019a; McCracken and Cooper 2020a, 2020b).

As we did with giant manta ray, 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 27).

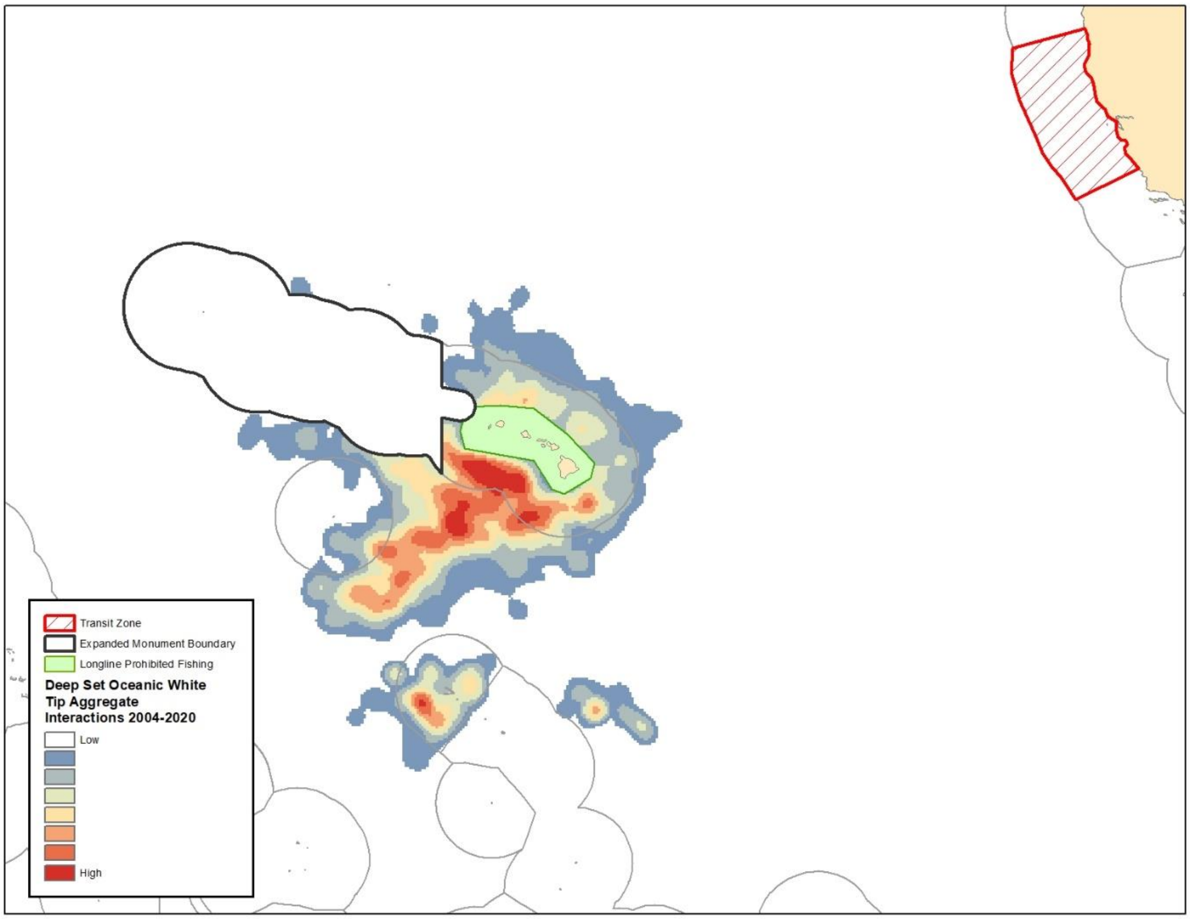


Figure 27. Geographical representation of observed aggregate oceanic whitetip shark interactions with the DSLL fishery between 2004 and 2020 (*n*= 5,149).

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 28). 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.

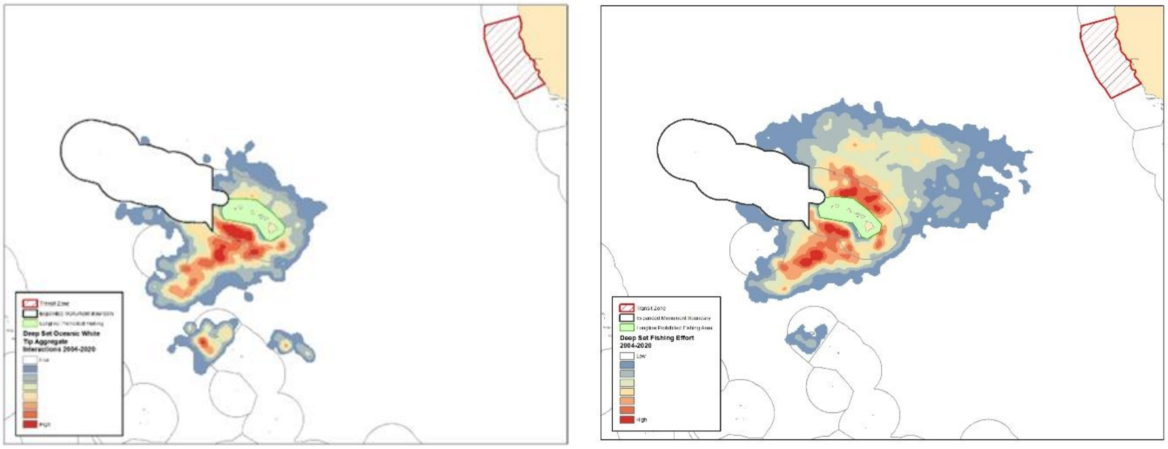


Figure 28. Heatmap comparison of observed oceanic whitetip shark interactions (left; *n* = 5,149) versus aggregated fishing effort during this same time frame (right; *n*= 65,245 from 2004 to 2020).

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. 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 location. 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, we focus on the mortality and injury that leads to latent mortality of oceanic whitetip sharks because this is the response for which we have the most information.

Herein, 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.

We present our best estimates of future mortality of oceanic whitetip sharks resulting from the Hawaii DSLL fishery in Table 13. 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 13. Estimates of number of oceanic whitetip sharks that will be incidentally captured and killed within the Hawaii DSLL fishery.

| Metric | Total Ints. | Individuals that are Observable (adjusted) | | Individuals that Bite Free (Unobserved) | | Total Mortality |
|---|---|---|---|---|---|---|
| | | Total Observable Interactions | At Vessel + Post Interaction Mortalities | Total Unobserved Interactions | No. of Bite Off Mortalities | |
| **Annual Mean** | 1,708 | 1,161 | 402 | 547 | 21 | 423 |
| **Annual 95th Percentile** | 3,185 | 2,166 | 749 | 1,019 | 40 | 789 |
| **Max. 5-Year Running Average** | 1,863 | 1,267 | 438 | 596 | 23 | 461 |
| **5-Year Cumulative Sum** | 9,316 | 6,335 | 2,192 | 2,981 | 116 | 2,308 |

### *Estimated Total Mortality of Observable Oceanic Whitetip Sharks*

Our estimate of the total number of *observable* 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 2014 to 2020, 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 *observable* 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 observable (TO) oceanic whitetip sharks that we expect to interact with this fishery.

Our calculated best estimate of at-vessel mortality for this fishery is 24.82%. We calculated at-vessel mortality two ways - using the all observer data from 2004 to 2020, and in addition we examined the at-vessel mortality of this fishery between 2014 and 2020. We expected that the at-vessel mortality for this species in this fishery may have improved following changes in prohibitions and handling protocols for sharks that occurred just prior to 2014, and expected this shorter time period would be more representative of current practices. Interestingly, the difference between relying upon the full dataset versus the truncated data set is small (about 1%). Between 2014 and 2020, a total of 2,542 oceanic whitetip sharks were incidentally captured by the Hawaii DSLL fishery and of those, 631 were dead for an at-vessel mortality of 24.82% ([631/2,542]x100 = 24.82) with 75.18% released alive. For comparison, we looked at the at-vessel mortality for the years 2004 to 2020, and based on the 5,149 observed interactions that occurred during this time calculated an at vessel mortality of 23.66% with 76.34% observed oceanic whitetip sharks were released alive. While these calculations do not appear to reflect an improvement in at vessel survival within the Hawaii DSLL fishery post implementation of CMMs that were designed to improve oceanic whitetip survival as we might have expected, we stand by our decision to rely on the truncated data set to calculate at vessel mortality for this fishery for two reasons: more recent data is still more likely to represent current practices, and the more recent and shorter data set provides a slightly higher estimate of at-vessel mortality for the species (and will result in a slightly higher estimate of mortalities for our jeopardy analysis, thus making it more protective of the species).

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* species 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:

$$TMO= (0.2482+0.13(1-0.2482))\ \ TO$$

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 14; 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 14. 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 |

| Capture Condition | Release Condition | Number | Percent |
|---|---|---|---|
|  | **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 15. 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 15. 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[5]. 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).

### *Estimated Total Mortality of Unobservable Oceanic Whitetip Sharks*

As noted previously, based on the work of Bigelow and Carvalho (2021) we expect that the change to monofilament leaders in this fishery will lead to about 32% fewer sharks being brought to the vessel after their capture because they would bite through the leader and self-release. These still represent interactions and some individual oceanic whitetip sharks are likely to experience injury and mortality from these interactions. How many and their actual fate we do not know.

Based on our review of the literature, there appears to be very little information on the fate of sharks that bite-free from fishing gear. Bayesian survival curves developed by Hutchinson et al. (2021) suggests that most oceanic whitetip sharks would survive if they escaped handling (survival estimated from Figure 5E in Hutchinson et al. 2021), whereas only about half would survive if the line were still attached (dragged and cut). In contrast, Harley et al. (2015) relied upon the probability of mortality given bite-off as 0.03 if the oceanic whitetip shark was lip-hooked, and 0.06 if the animal was gut-hooked. Harley et al. (2015) acknowledged that their values are based on expert opinion. Given that the Hutchinson et al. (2021) numbers are within the range of Harley et al. (2015) we relied upon Harley et al. (2015) to calculate the mortality of animals that bite off the leader. Based on Afonso et al. (2011), the probability of an oceanic whitetip shark would be externally hooked ($p = 0.7$) is greater than being internally hooked (e.g. swallowed or lodged in mouth; $p = 0.3$) when c-hooks are used (values estimated from Figure 4 in Afonso et al. 2011). Given this, we calculated the weighted average of these values to use for our mortality estimate for those sharks that would be expected to die after biting off the leader. This calculation is represented by:

$$0.039 = ((0.03*0.7)+(0.06*0.3))/(0.7+0.3)$$

Together these values and calculations give us our best estimates for the total mortality of *unobservable* oceanic whitetip sharks in this fishery, which we presented at the beginning of this response analysis in Table 13. Given our estimated number of *unobservable* interactions, we estimate that on average 21 oceanic whitetip sharks will die in the Hawaii DSLL fishery each year after biting through the monofilament line (95P=40).

In comparison the maximum 5-year running average suggests that about 597 *unobservable* interactions with oceanic whitetip sharks are expected in this fishery, and that 23 of those

---

[5] 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.

animals will die from their interaction (Table 15). Based on the maximum 5-year running average this suggests that the fishery would interact with a total of 74,528 oceanic whitetip sharks in the next 40 years (10 yrs=18,632), and 18,248 would die over the same time period (10 yrs = 4,562 mortalities).

### Risk Analysis

In this section of this biological opinion, we examine the probable consequences of capturing nearly 75,000 oceanic whitetip sharks in the Hawaii DSLL fishery for the next 40 years (N=1843*40), and killing nearly 20,000 individuals during the same time period (N=18,444=461*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 423 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 1,863 oceanic whitetip sharks will be incidentally captured each year (N=9,316 individuals every five years) in the Hawaii DSLL fishery. From these animals, we estimate that 456 individuals will die as a result of the interaction (N=2,281 mortalities in 5 years). On average, we expect only about 232 of these oceanic whitetip sharks would be recorded as an observed interaction (N=observable oceanic whitetip sharks multiplied by standard 20% observer coverage [not adjusted for actual observer coverage] =1,161*0.20).

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. 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 0.05% 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 16, Table 17). 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 with about 0.25% 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 0.21% of the population with 0.05% being killed.

Accumulating incidental captures over the initial 10 years suggests that up to 0.57% of the population would be killed over 10 years if the population is increasing; 0.61% 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 0.03% 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 with about 0.15% of the population with 0.04% 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 with about 0.12% of the population with 0.03% being killed.

Accumulating incidental captures over the initial 10 years suggests that up to 0.34% of the population would be killed over 10 years if the population is increasing; 0.36% if the population is declining (Table 16, Table 17). Our estimates assume that the estimated population trends in the WCPO is indicative of the trend for the entire Pacific population.

Table 16. Estimated effect of Hawaii DSLL fishery on the Pacific population of oceanic whitetip sharks incidentally captured and killed by the fishery 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 | 0.05 | 0.03 |
| **Max. 5-Year Running Average** | 2022 | 772,196 | 1,286,995 | 0.24 | 0.14 | 0.06 | 0.04 |
| **Max. 5-Year Running Average** | 2032 | 762,222 | 1,270,372 | 0.24 | 0.15 | 0.06 | 0.04 |
| **10-year based on 5-yr running average** | 2032 | 762,222 | 1,270,372 | 2.44 | 1.47 | 0.61 | 0.36 |
| **Max. 5-Year Running Average** | 2042 | 752,377 | 1,253,965 | 0.25 | 0.15 | 0.06 | 0.04 |
| **10-year Mean Projected 20 years** | 2042 | 752,377 | 1,253,965 | 2.48 | 1.49 | 0.61 | 0.37 |
| **Max. 5-Year Running Average** | 2052 | 742,660 | 1,237,769 | 0.25 | 0.15 | 0.06 | 0.04 |
| **10-year Mean projected 30 years** | 2052 | 742,660 | 1,237,769 | 2.51 | 1.51 | 0.62 | 0.37 |
| **Max. 5-Year Running Average** | 2062 | 733,067 | 1,221,782 | 0.25 | 0.15 | 0.06 | 0.04 |
| **10-year Mean projected 40 years** | 2062 | 733,067 | 1,221,782 | 2.54 | 1.52 | 0.63 | 0.38 |
| **95th Percentile** | 2062 | 733,067 | 1,221,782 | 0.43 | 0.26 | 0.11 | 0.06 |

Table 17. Estimated effect of Hawaii DSLL fishery on the Pacific population of oceanic whitetip sharks incidentally captured and killed by the fishery 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 | 0.05 | 0.03 |
| **Max. 5-Year Running Average** | 2022 | 783,632 | 1,306,052 | 0.24 | 0.14 | 0.06 | 0.04 |
| **Max. 5-Year Running Average** | 2032 | 812,356 | 1,353,925 | 0.23 | 0.14 | 0.06 | 0.03 |
| **10-year based on 5-yr running average** | 2032 | 812,356 | 1,353,925 | 2.29 | 1.38 | 0.57 | 0.34 |
| **Max. 5-Year Running Average** | 2042 | 842,134 | 1,403,557 | 0.22 | 0.13 | 0.05 | 0.03 |
| **10-year Mean Projected 20 years** | 2042 | 842,134 | 1,403,557 | 2.21 | 1.33 | 0.55 | 0.33 |
| **Max. 5-Year Running Average** | 2052 | 873,002 | 1,455,005 | 0.21 | 0.13 | 0.05 | 0.03 |
| **10-year Mean projected 30 years** | 2052 | 873,002 | 1,455,005 | 2.13 | 1.28 | 0.53 | 0.32 |
| **Max. 5-Year Running Average** | 2062 | 905,002 | 1,508,338 | 0.21 | 0.12 | 0.05 | 0.03 |
| **10-year Mean projected 40 years** | 2062 | 905,002 | 1,508,338 | 2.06 | 1.24 | 0.51 | 0.31 |
| **95th Percentile** | 2062 | 905,002 | 1,508,338 | 0.35 | 0.21 | 0.09 | 0.05 |

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 about 0.4% 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 423 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 0.06% annually. We consider the implications of this effect on OWT, as a species, in the *Integration and Synthesis* section of this opinion.

## 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. 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 giant manta rays and oceanic whitetip sharks 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, which we expect would continue in perpetuity since longlining has a history of more than 100 years in Hawaii (Kaneko 2015). 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 threatened oceanic whitetip sharks and giant manta rays, 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 threatened oceanic whitetip sharks, and threatened giant manta, it has very small effects on the dynamics of the populations those individuals represent or the species those populations comprise. As a result, we believe it does not appreciably reduce these species’ likelihood of survival and recovery in the wild.

We explain the basis for this conclusion for each species in the following sections. These summary 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 giant manta ray and oceanic whitetip shark 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: 70) threatened giant manta ray each year, although, in some years, higher or lower numbers of giant manta ray are likely to be incidentally captured. On average, 10 of the manta rays incidentally captured are likely to die as a result of their capture; although this estimate does not consider possible post-release mortalities which would include giant manta rays that die as a result of their injuries or that are killed by shark. Over a 40 year analytical time frame we would anticipate a cumulative total of 1,432 incidental captures and 623 mortalities. These numbers do not consider reductions in adult recruitment, reductions in reproductive success (for example, adult females that increase the interval between reproductive events), or spontaneous abortions resulting from capture myopathy, injury, or stress pathology.

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 2019). 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, personal communication 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 22). 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 the maximum 5-year running average as the best estimate of the effects of this fishery on giant manta rays and therefore the results of the scenario (Scenario 1) using this value is our best estimate for the impacts to the species. 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; accordingly, this scenario likely overestimates the impacts of the fishery on 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 our 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 400 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 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 is reasonably certain the proposed action will allow increasing population trajectories in the putative subpopulations of giant manta ray in the

*Action Area* under both scenarios with 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 400, 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 1.2.2 (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.

Our analysis is based on a number of assumptions, which we have attempted to ensure are clearly explained in this opinion and the consultation record. As stated by Kuljis (2014) "there will likely always be some available science that contradicts the agency's ultimate conclusions." While it is not concerning necessarily that information may be available to contradict the agency, our concern is largely focused on: (1) did we identify and consider available information; and (2) did we consider it in a way that is reasoned, given the available evidence to help the reader understand what data or information is best, and why. We believe that we have done this.

## 5.2 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 0.345, we expect 423 of the oceanic whitetip sharks incidentally captured are likely to die as a result of their capture each year. Over a 40 year analytical time frame we would anticipate a cumulative total of about 75,000 incidental captures and almost 20,000 mortalities. This estimate does not consider reductions in adult recruitment, reductions in reproductive success (for example, adult females that increase the interval between reproductive events), or spontaneous abortions resulting from capture myopathy, injury, or stress pathology.

As discussed in the *Status of Listed Resources* and the *Effects of the Action* section of this biological opinion, the best available information suggest that oceanic whitetip sharks in the Pacific Ocean are likely comprised of one 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). Although oceanic whitetip sharks have low fecundities for sharks (between 0 and 15 pups) and a biennial reproductive cycle, the effect of the deaths on a population that size would be undetectable if we consider reproduction.

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 (Futerman 2018; Rice and Harley 2012). 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 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. 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 a mean of 1,708 (95th percentile: 3,185) oceanic whitetip sharks and the resulting death of 423 sharks per year. At projected abundance levels (1,292,023 individuals), this constitutes death of 0.03% of the population. 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 0.05% 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. This means without the proposed action, the population trends would range from -0.08% to 0.41% per year.

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 1.2.2 (*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.

# 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 threatened giant manta ray or threatened oceanic whitetip shark.

# 7 INCIDENTAL TAKE STATEMENT

The proposed action results in the incidental take of threatened giant manta ray and threatened oceanic whitetip shark. Currently there are no take prohibitions for giant manta rays or oceanic whitetip sharks, so an exemption from the take prohibitions of section 9 of the ESA is neither necessary nor appropriate for either of these two species. However, 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 so the action does not jeopardize the species 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 ITS." The measures described below are nondiscretionary, and must be undertaken by NMFS. NMFS has a continuing duty to regulate the activity covered by this incidental take statement.

## 7.1 Amount or Extent of Take

The following levels of incidental take are 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, this 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).

We anticipate the amount of take from the Hawaii DSLL in Table 18.

Table 18. The anticipated mean annual captures and mortalities from the Hawaii DSLL. These are the estimated number of captures, expanded from observed number of captures.

| **Species** | **Annual** | |
|---|---|---|
| | **Mean Number Captured** | **Mean Number Killed** |
| Oceanic whitetip shark[a] | 1,161 | 402 |
| Giant manta ray | 20 | 9 |

[a] The historical estimated and observed captures and mortalities for oceanic whitetip shark have been reduced by 32% to represent the anticipated captures and mortalities with the fishery using monofilament leaders.

As provided in the text of the statute and legislative history, an additional purpose of an ITS is to serve as a reinitiation trigger (see Chapter 12 "Reinitiation of Consultation") that provide clear signals that the level of anticipated take has been exceeded and, therefore, would require reexamination of the Federal agency action through a reinitiated consultation. Because of high variability in annual interactions, the mean annual numbers above are not appropriate for triggering reinitiation. Instead, we rely on the maximum 5-year running sum and 5-year running average to monitor the action's incidental take of threatened and endangered species. 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. Over any five consecutive years the number of interactions are not to exceed the defined 5-year running sum, which is five times the 5-year running average, rounded up to the nearest integer (Table 19).

Table 19. Reinitiation triggers for the Hawaii DSLL. These are the estimated number of captures, expanded from observed number of captures. Exceeding the maximum 5-year running average/sum over any five consecutive years is a reinitiation trigger.

| **Species** | **Maximum 5-Year Max Running Average** | **Maximum 5-Year Max Running Sum** |
|---|---|---|
| Oceanic whitetip shark[a] | 1,266.3[a] | 6,335 |
| Giant manta ray | 28.8 | 144 |

[a] The historical estimated and observed captures and mortalities for oceanic whitetip shark have been reduced by 32% to represent the anticipated captures and mortalities with the fishery using monofilament leaders.

## 7.2 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 species and to monitor the level and nature of any incidental takes. These measures are non-discretionary.

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 the extent of take is not exceeded, and that the terms and conditions in this incidental take statement are effective in minimizing incidental take.

## 7.3 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.

1. The following terms and conditions implement Reasonable and Prudent Measure No. 1:
    a. Minimize the amount of trailing gear left on giant manta rays and oceanic whitetip sharks to the maximum extent possible to reduce the amount of injury and harm and improve post release mortality.
2. The following terms and conditions implement Reasonable and Prudent Measure No. 2:
    a. Observers shall collect standardized information regarding the incidental capture, injury, and mortality of giant manta rays and oceanic whitetip sharks for each interaction including gear, and set information, as well as the presence or absence of tags on these species and not as a random sample of individuals as currently practiced.

       To the maximum extent possible, observers shall identify hooking location, and estimated length of gear left on giant manta rays and oceanic whitetip sharks at release and when they cannot be boarded. These data are intended to allow NMFS to assign these interactions into the categories developed through NMFS’ most current post-hooking mortality guidelines and improve estimates of harm, injury, and mortalities within this fishery.
    b. NMFS’ SFD shall provide an annual report to the NMFS’ PRD that details the results of its monitoring of bycatch reports during each fishing season. These reports shall be submitted in writing within one month of the new fishing year and will summarize all statistical information based on a January 1 through December 31 fishing year.

## 7.4 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.

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 threatened elasmobranchs. Specifically, NMFS should:
    a. Monitor the amount and extent of trailing gear that is left on oceanic whitetip sharks and giant manta rays to understand fishery performance with this management measure and improve understanding of the effects of the Hawaii DSLL fishery on these species.
    b. Implement 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 for ESA-listed sharks as outlined in WCPFC CMM 2010-07.

In order to keep NMFS' Protected Resources Division informed of actions minimizing or avoiding adverse effects or benefitting listed species or their habitats, the Sustainable Fisheries Division should notify the Endangered Species Division of any conservation recommendations they implement in their final action.

## 7.5 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

Acevedo-Whitehouse K, and Duffus AL. 2009. Effects of environmental change on wildlife health. Philosophical Transactions of the Royal Society B: Biological Sciences. 364(1534):3429-3438.

Adams DH, and Amesbury E. 1998. Occurrence of the manta ray, *Manta birostris*, in the Indian River Lagoon, Florida. Florida Scientist.7-9.

Afonso, A. S., F. H. V. Hazin, F. Carvalho, J. C. Pacheco, H. Hazin, D. W. Kerstetter, D. Murie, and G. H. Burgess. 2011. Fishing gear modifications to reduce elasmobranch mortality in pelagic and bottom longline fisheries off Northeast Brazil. Fisheries Research. 108(2-3):336-343.

Allen, CD, Lemons GE, Eguchi T, LeRoux RA, Fahy CC, Dutton PH, Peckham SH, and Seminoff JA. 2013. Stable isotope analysis reveals migratory origin of loggerhead turtles in the Southern California Bight. Marine Ecology Progress Series. 472:275-285.

Anderson RC, Adam MS, and Goes JI. 2011. From monsoons to mantas: seasonal distribution of *Manta alfredi* in the Maldives. Fisheries Oceanography. 20(2):104-113.

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).

Backus RH, Springer S, and Arnold Jr EL. 1956. A contribution to the natural history of the white-tip shark, *Pterolamiops longimanus* (Poey). Deep Sea Research (1953). 3(3):178-188.

Baum J, Medina E, Musick JA, and Smale M. 2015. *Carcharhinus longimanus*. The IUCN Red List of Threatened Species 2015: e.T39374A85699641. (Downloaded on 30 June 2018)doi:http://dx.doi.org/10.2305/IUCN.UK.2015.RLTS.T39374A85699641.en.

Baum JK, and Myers RA. 2004. Shifting baselines and the decline of pelagic sharks in the Gulf of Mexico. Ecology Letters. 7(2):135-145.

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.

Beerkircher LR, Cortés E, and Shivji M. 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.

Bermudez R, Winder M, Stuhr A, Almen AK, 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.

Beverly S, and Chapman L. 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., J. Rice and F. Carvalho. 2022. Future stock projections of oceanic whitetip sharks in the Western and Central Pacific Ocean. Eighteenth Regular Session of the Scientific Committee of the Western and Central Pacific Fisheries Commission; WCPFC-SC18-2022/EB-WP-0. 19 pp.

Bigelow, K., Rice, J. 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.

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.

Bignami S, Sponaugle S, and Cowen RK. 2013. Response to ocean acidification in larvae of a large tropical marine fish, Rachycentron canadum. Global Change Biology. 19(4):996-1006.

Bleckmann H, and Hofmann MH. 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.

Bonfil R, Clarke S, Nakano H, Camhi MD, Pikitch EK, and Babcock EA. 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., Mardhiah, U., Siregar, H., Hunter, J., Putra, M. I. H., Marlow, J., Cahyana, A., Boysandi, Demoor, A.Y.L., Lewis, S., Adhiasto, D., Adrianto, L., and Yulianto, I. 2021. An integrated approach to tackling wildlife crime: Impact and lessons learned from the world's largest targeted manta ray fishery. Conservation Science and Practice. 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.

Briscoe DK, Parker DM, Balazs GH, Kurita M, Saito T, Okamoto H, Rice M, Polovina JJ, and Crowder LB. 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 DK, Parker DM, Bograd S, Hazen E, Scales K, Balazs GH, Kurita M, Saito T, Okamoto H, Rice M *et al*. 2016b. Multi-year tracking reveals extensive pelagic phase of juvenile loggerhead sea turtles in the North Pacific. Movement Ecology. 4(1):23.

Brodziak J, Walsh WA, and Hilborn R. 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.

Burgess KB. 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 DS, Harvey CJ, and McElhany P. 2013. Potential impacts of ocean acidification on the Puget Sound food web. ICES Journal of Marine Science. 70(4):823-833.

Camargo SM, Coelho R, Chapman D, Howey-Jordan L, Brooks EJ, Fernando D, Mendes NJ, Hazin FH, Oliveira C, Santos MN *et al*. 2016. Structure and Genetic Variability of the Oceanic Whitetip Shark, *Carcharhinus longimanus*, Determined Using Mitochondrial DNA. PLoS One. 11(5):e0155623.

Campana SE, Joyce W, and Manning MJ. 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, J. R., and D. A. Roach. 2020. Biodemography: An Introduction to Concepts and Methods. Princeton University Press.

Chan F, Barth JA, Blanchette CA, Byrne RH, Chavez F, Cheriton O, Feely RA, Friederich G, Gaylord B, Gouhier T *et al*. 2017. Persistent spatial structuring of coastal ocean acidification in the California Current System. Science Reports. 7(1):2526.

Chin A, Kyne PM, Walker TI, and McAuley RB. 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.

CITES. 2013. Consideration of proposals for amendment of Appendices I and II: Manta Rays. Bangkok, Thailand, March 3-14.

Clark TB. 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. 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, Harley S, Hoyle S, and Rice J. 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, Yokawa K, Matsunaga H, and Nakano H. 2011b. Analysis of North Pacific Shark Data from Japanese Commercial Longline and Research/Training Vessel Records. Pohnpei, Federated States of Micronesia. p. 89.

Clarke SC, Harley SJ, Hoyle SD, and Rice JS. 2012. Population trends in Pacific Oceanic sharks and the utility of regulations on shark finning. Conservation Biology. 27(1):197-209.

Clarke SC, Magnussen JE, Abercrombie DL, McAllister MK, and Shivji MS. 2006. 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.

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, Hazin FHV, Rego M, Tambourgi M, Oliveira P, Travassos P, Carvalho F, and Burgess G. 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, Lino PG, and Santos MN. 2011. At-haulback mortality of elasmobranchs caught on the Portuguese longline swordfish fishery in the Indian Ocean. IOTC–2011–WPEB07–31. 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.

Compagno LJV. 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.

Cortes E. 2002. Incorporating uncertainty into demographic modeling: application to shark populations and their conservation. Conservation Biology. 16(4):1048-1062.

Cortes E, Brown CA, and Beerkircher L. 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 LI, Jaine FR, and Kashiwagi T. 2015. First photographic records of the giant manta ray *Manta birostris* off eastern Australia. PeerJ. 3:e742.

Couturier LIE, Marshall AD, Jaine FRA, Kashiwagi T, Pierce SJ, Townsend KA, Weeks SJ, Bennett MB, and Richardson AJ. 2012. Biology, ecology and conservation of the Mobulidae. Journal of fish biology. 80(5):1075-1119.

Croll DA, Dewar H, Dulvy NK, Fernando D, Francis MP, Galvan-Magana F, Hall M, Heinrichs S, Marshall AD, McCauley D *et al.* 2016. 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.

Deakos MH, Baker JD, and Bejder L. 2011. Characteristics of a manta ray *Manta alfredi* -population off Maui, Hawaii, and implications for management. Marine Ecology Progress Series. 429:245-260.

Dewar H, Mous P, Domeier M, Muljadi A, Pet J, and Whitty J. 2008. Movements and site fidelity of the giant manta ray, *Manta birostris*, in the Komodo Marine Park, Indonesia. Marine Biology. 155(2):121-133.

Domingo A, Pons M, Jimenez S, Miller P, Barcelo C, and Swimmer Y. 2012. Circle Hook Performance in the Uruguayan Pelagic Longline Fishery. Bulletin of Marine Science. 88(3):499-511.

Doney SC, Ruckelshaus M, Duffy JE, Barry JP, Chan F, English CA, Galindo HM, Grebmeier JM, Hollowed AB, Knowlton N *et al*. 2012. Climate change impacts on marine ecosystems. Annual Review of Marine Science. 4:11-37.

Duffy CAJ, 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., Pardo, S.A., Simpfendorfer, C.A., and J.K. Carlson. 2014. Diagnosing the dangerous demography of manta rays using life history theory. PeerJ. 2:e400.

EPA. 2010. Climate Change Indicators in the United States. 80 p.

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 Stevens G. 2011. A study of Sri Lanka’s manta and mobula ray fishery. The Manta Trust.

Fossi MC, Baini M, Panti C, Galli M, Jimenez B, Munoz-Arnanz J, Marsili L, Finoia MG, and Ramirez-Macias D. 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 MC, Coppola D, Baini M, Giannetti M, Guerranti C, Marsili L, Panti C, de Sabata E, and Clo S. 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.

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.

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.

Futerman, A. M. 2018. At the Intersection of Science & Policy: International Shark Conservation & Management. Duke Environmental Law & Policy Forum. 28:259-306.

Germanov ES. 2015a. From manta rays to mass spectrometry. In: Foundation MM, editor. http://www.marinemegafauna.org/manta-rays-mass-spectrometry/.

Germanov ES. 2015b. Microplastics & Megafauna.

Germanov ES, Marshall AD, Bejder L, Fossi MC, and Loneragan NR. 2018. Microplastics: No small problem for filter-feeding megafauna. Trends in Ecology & Evolution. 33(4):227-232.

Gilman E, Suuronen P, Hall M, and Kennelly S. 2013. Causes and methods to estimate cryptic sources of fishing mortality. Journal of Fish Biology. 83(4):766-803.

Graham RT, Witt MJ, Castellanos DW, Remolina F, Maxwell S, Godley BJ, and Hawkes LA. 2012. Satellite tracking of manta rays highlights challenges to their conservation. PLoS One. 7(5):e36834.

Griffiths, S.P. and N. Lezama-Ochoa. 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. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *31*(10), pp.2910-2925.

Haigh R, Ianson D, Holt CA, Neate HE, and Edwards AM. 2015. Effects of ocean acidification on temperate coastal marine ecosystems and fisheries in the northeast Pacific. PLoS One. 10(2):e0117533.

Hall MA, and Roman M. 2013. Bycatch and non-tuna catch in the tropical tuna purse seine fisheries of the world. FAO Fisheries and Aquaculture Technical Paper. (568):244.

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.

Hawaii State Department of Land and Natural Resources. 2019. Commercial marine landings summary report: Calendar year 2019. https://dlnr.hawaii.gov/dar/files/2021/04/cmlstr2019.pdf

Hawaii Longline Association. 2020. RE: Hawaii-Based Commercial Longline Deep-Set Fishery Section 7 Consultation. 7 p.

Hazen EL, Jorgensen SJ, Rykaczewski RR, Bograd SJ, Foley DG, Jonsen ID, Shaffer SA, Dunne JP, Costa DP, Crowder LB *et al*. 2012. Predicted habitat shifts of Pacific top predators in a changing climate. Nature Climate Change. 3(3):234-238.

Hazin FH, Hazin HG, and Travassos P. 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 AR, Acuna D, Ketchum JT, Penaherrera C, Green J, Marshall A, Guerrero M, and Shillinger G. 2014. Elasmobranchs of the Galapagos marine reserve. The Galapagos Marine Reserve. Springer. p. 23-59.Heinrichs S, O'Malley M, Medd H, and Hilton P. 2011. Manta Ray of Hope: Global Threat to Manta and Mobula Rays. Manta Ray of Hope Project.

Heinrichs S, O’Malley MP, Medd H, Hilton P. 2011. The global threat to Manta and Mobula Rays. Manta Ray of Hope, WildAid, San Francisco, California.

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.

Homma K, Maruyama T, Itoh T, Ishihara H, and Uchida S. 1999. Biology of the manta ray, *Manta birostris Walbaum*, in the Indo-Pacific. Proceedings of the 5th Indo-Pacific Fish Conference. p. 209.

Howey-Jordan LA, Brooks EJ, Abercrombie DL, Jordan LK, Brooks A, Williams S, Gospodarczyk E, and Chapman DD. 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 LA, Tolentino ER, Papastamatiou YP, Brooks EJ, Abercrombie DL, Watanabe YY, Williams S, Brooks A, Chapman DD, and Jordan LKB. 2016. Into the deep: the functionality of mesopelagic excursions by an oceanic apex predator. Ecology and Evolution. 6(15):5290-5304.

Hutchinson M, Poisson F, and Swimmer Y. 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.

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.

IPCC (Intergovernmental Panel on Climate Change). 2007. Climate Change 2007: Synthesis Report. p. 23-73.

IPCC. 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.

Jambeck JR, Geyer R, Wilcox C, Siegler TR, Perryman M, Andrady A, Narayan R, and Law KL. 2015. Plastic waste inputs from land into the ocean. Science. 347(6223):768-771.

Jaine F, Rohner C, Weeks S, Couturier L, Bennett M, Townsend K, Richardson A (2014) Movements and habitat use of reef manta rays off eastern Australia: Offshore excursions, deep diving and eddy affinity revealed by satellite telemetry. Mar Ecol Prog Ser 510: 73-86

Jones TT, Martin S, Eguchi T, Langseth B, Baker J, and Yau A. 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 SJ, Chen NF, Hsu HH, and Liu KM. 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.

Kajiura SM. 2001. Head morphology and electrosensory pore distribution of carcharhinid and sphyrnid sharks. Environmental Biology of Fishes. 61(2):125-133.

Kaneko JJ. 2015. Hawaii Pelagic Longline Fishery Fact Sheet. Hawaii Seafood Council. p. 3.

Kashiwagi T, Ito T, and Sato F. 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.

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.

Last PR, and Stevens JD. 2009. Sharks and rays of Australia. CSIRO Publishing. p. 656.

Lawson JM, Fordham SV, O'Malley MP, Davidson LN, Walls RH, Heupel MR, Stevens G, Fernando D, Budziak A, and Simpfendorfer CA. 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 JA, MacLeod CD, Santos MB, Pierce GJ, Crick HQP, and Robinson RA. 2006. Potential effects of climate change on marine mammals. Oceanography and Marine Biology - an Annual Review. 44:431-464.

Lessa R, Paglerani R, and Santana F. 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 SA, Setiasih N, Fahmi F, Dharmadi D, O'Malley MP, Campbell SJ, Yusuf M, and Sianipar AB. 2015. Assessing Indonesian manta and devil ray populations through historical landings and fishing community interviews. Peer J PrePrints 6:e1334v1.

Liu K-M, and Tsai W-P. 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 MV, and Channel R. 1998. Range collapse, re-introductions, and biogeographic guidelines for conservation. Conservation Biology. 12(2):481-484.

Luiz OJ, Balboni AP, Kodja G, Andrade M, and Marum H. 2009. Seasonal occurrences of *Manta birostris* (Chondrichthyes: Mobulidae) in southeastern Brazil. Ichthyological Research. 56(1):96-99.

Lyons K, Carlisle A, Preti A, Mull C, Blasius M, O'Sullivan J, Winkler C, and Lowe CG. 2013. Effects of trophic ecology and habitat use on maternal transfer of contaminants in four species of young of the year lamniform sharks. Mar Environ Res. 90:27-38.

Macfadyen G, Huntington T, and Cappell R. 2009. Abandoned, lost or otherwise discarded fishing gear. Food and Agriculture Organization of the United Nations (FAO). p. 21.

MacLeod CD. 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.

MantaMatcher. 2016. Facebook: Manta Matcher - the Wildbook for Manta Rays. https://www.facebook.com/mantamatcher/.

Marshall A, and Conradie J. 2014. Manta Fishery Solor. Marine Megafauna Foundation.

Marshall A, Kashiwagi T, Bennett MB, Deakos M, Stevens G, McGregor F, Clark T, Ishihara H, and Sato K. 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 AD, and Bennett MB. 2010. Reproductive ecology of the reef manta ray *Manta alfredi* in southern Mozambique. Journal of fish biology. 77(1):169-190.

Marshall AD, Compagno LJV, and Bennett MB. 2009. Redescription of the genus Manta with resurrection of *Manta alfredi* (Krefft, 1868)(Chondrichthyes; Myliobatoidei; Mobulidae). Zootaxa. 2301:1-28.

Mas F, Forselledo R, and Domingo A. 2015. Mobulid ray by-catch in longline fisheries in the south-western Atlantic Ocean. Marine and Freshwater Research. 66(9):767-777.

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. PIFSC Data Report DR-20-

004. Pacific Island Fisheries Science Center. Honolulu, HI. PIFSC Data Report DR-21-005. Pacific Island Fisheries Science Center. Honolulu, HI. 4 p.

McCracken, M. L. and B. Cooper. 2021. 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.

Medeiros AM, Luiz OJ, and Domit C. 2015. Occurrence and use of an estuarine habitat by giant manta ray *Manta birostris*. Journal of fish biology. 86(6):1830-1838.

Milessi AC, and Oddone MC. 2003. Primer registro de *Manta birostris* (Donndorff 1798)(Batoidea: Mobulidae) en el Rio de La Plata, Uruguay. Gayana (Concepción). 67(1):126-129.

Miller MH, and Klimovich C. 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.

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.

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 ABM. 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.

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 CD, Fragoso N, Musyl MK, and Brill RW. 2006. Predicting postrelease survival in large pelagic fish. Transactions of the American Fisheries Society. 135(5):1389-1397.

Munday PL, Leis JM, Lough JM, Paris CB, Kingsford MJ, Berumen ML, and Lambrechts J. 2009. Climate change and coral reef connectivity. Coral Reefs. 28(2):379-395.

Murray KT. 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.

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, Okazaki M, and Okamota H. 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.

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.

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.

NMFS. 2001. Final Environmental Impact Statement for Fishery Management Plan, Pelagic Fisheries of the Western Pacific Region. Pacific Islands Region.

NMFS. 2004. Biological Opinion on Proposed Regulatory Amendments to the Fisheries Management Plan for the Pelagic Fisheries of the Western Pacific Region. p. 281.

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. 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. 2015. 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. 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. In: Commerce USDo, editor. p. 1-8.

NMFS. 2017. 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. 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. 2019. Biological Opinion. Continued Authorization of the Hawaii Shallow-set Longline Fishery. NMFS, Pacific Island Regional Office, Honolulu, HI. 506 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. 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.

NOAA (National Oceanic and Atmospheric Administration). 2015. Pacific Islands Marine Debris Encounter Reports 2007-2015. p. 2.

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

Notarbartolo-di-Sciara G, and Hillyer EV. 1989. Mobulid rays off eastern Venezuela (Chondrichthyes, Mobulidae). Copeia.607-614.

O'Malley MP, Lee-Brooks K, and Medd HB. 2013. The global economic impact of manta ray watching tourism. PLoS One. 8(5):e65051.

O'Shea OR, Kingsford MJ, and Seymour J. 2010. Tide-related periodicity of manta rays and sharks to cleaning stations on a coral reef. Marine and Freshwater Research. 61:65-73.

Paig-Tran EW, Kleinteich T, and Summers AP. 2013. The filter pads and filtration mechanisms of the devil rays: Variation at macro and microscopic scales. Journal of Morphology. 274(9):1026-1043.

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.

Peatman T, Allain V, Caillot S, Park T, Williams P, Tuiloma I, Smith N, Panizza A, and Fukofuka S. 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, Bell L, Allain V, Caillot P, Williams S, Tuiloma I, Panizza A, Tremblay-Boyer L, Fukofuka S, and Smith N. 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.

Poeta G, Staffieri E, Acosta ATR, and Battisti C. 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.

Portnoy DS, McDowell JR, Heist EJ, Musick JA, and Graves JE. 2010. World phylogeography and male-mediated gene flow in the sandbar shark, *Carcharhinus plumbeus*. Molecular Ecology. 19(10):1994-2010.

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.

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, 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.

Rochman CM, Kurobe T, Flores I, and Teh SJ. 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.

Rohner CA, Pierce SJ, Marshall AD, Weeks SJ, Bennett MB, and Richardson AJ. 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 CL. 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.

Ryder CE, Conant TA, and Schroeder BA. 2006. Report of the Workshop on Marine Turtle Longline Post-Interaction Mortality. U.S. Dep. Commerce, NOAA Technical Memorandum NMFS-F/OPR-29. p. 36.

Sales G, Giffoni BB, Fiedler FN, Azevedo VG, Kotas JE, Swimmer Y, and Bugoni L. 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 FM, Duarte-Neto PJ, and Lessa RP. 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. Laboratorio de Dinâmica de Populacoes Marinhas - DIMAR.

Schwartz FJ. 1984. A blacknose shark from North Carolina deformed by encircling monofilament line. Florida Scientist.62-64.

Scott, M., Cardona, E., Scidmore-Rossing, K., Royer, M., Stahl, J. and Hutchinson, M., 2022. What's the catch? Examining optimal longline fishing gear configurations to minimize negative impacts on non-target species. *Marine Policy*, *143*, p.105186.

Seki T, Taniuchi T, Nakano H, and Shimizu M. 1998. Age, Growth and Reproduction of the Oceanic Whitetip Shark from the Pacific Ocean. Fisheries Science. 64(1):14-20.

Setala O, Fleming-Lehtinen V, and Lehtiniemi M. 2014. Ingestion and transfer of microplastics in the planktonic food web. Environmental Pollution. 185:77-83.

Silber GK, Lettrich MD, 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.

Silverman BW. 1986. Density estimation for statistics and data analysis. Chapman and Hall, London.

Simmonds MP, and Eliott WJ. 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 Mandelman, J.W. 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 IM, van der Ouderaa IBC, and Tibirica Y. 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 SE, Au DW, and Show C. 2008. Intrinsic rates of increase in pelagic elasmobranchs. Sharks of the open ocean: biology, fisheries conservation. p. 288-297.

Sreelekshmi, S., Sukumaran, S., Kishor, T.G., Sebastian W., and Gopalakrishnan A. 2020. Population genetic structure of the oceanic whitetip shark, *Carcharhinus longimanus,* along the Indian coast. Marine Biodiversity 50:78

Stevens, G., D. Fernando, and G. N. Di Sciara. 2018. Guide to the Manta and Devil Rays of the World. Princeton University Press.

Stevens JD, Bonfil R, Dulvy NK, and Walker PA. 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.

Stewart JD, Beale CS, Fernando D, Sianipar AB, Burton RS, Semmens BX, and Aburto-Oropeza O. 2016a. Spatial ecology and conservation of *Manta birostris* in the Indo-Pacific. Biological Conservation. 200:178-183.

Stewart JD, Hoyos-Padilla EM, Kumli KR, and Rubin RD. 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).Stewart, K., Johnson, C. & Godfrey, M.H. (2007) The minimum size of leatherbacks at reproductive maturity, with a review of sizes for nesting females from the Indian, Atlantic and Pacific Ocean basins. Herpetological Journal, 17:123-128.

Strasburg DW. 1958. Distribution, abundance, and habits of pelagic sharks in the central Pacific Ocean. Fisheries. 1:2S.

Swimmer Y, and Gilman E. 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.

Swimmer Y, Gutierrez A, Bigelow K, Barcelo C, Schroeder B, Keene K, Shattenkirk K, and Foster DG. 2017. Sea Turtle Bycatch Mitigation in U.S. Longline Fisheries. Frontiers in Marine Science. 4.

The Hawaii Association for Marine Education and Research Inc. 2005. Manta rays. http://www.hamerinhawaii.org.

Tolotti M, Bauer R, Forget F, Bach P, Dagorn L, and Travassos P. 2017. Fine-scale vertical movements of oceanic whitetip sharks (*Carcharhinus longimanus*). Fishery Bulletin. 115(3):380-395.

Tolotti MT, Bach P, Hazin F, Travassos P, and Dagorn L. 2015. Vulnerability of the Oceanic Whitetip Shark to Pelagic Longline Fisheries. PLoS One. 10(10):e0141396.

Tolotti MT, Travassos P, Fredou FL, Wor C, Andrade HA, and Hazin F. 2013. Size, distribution and catch rates of the oceanic whitetip shark caught by the Brazilian tuna longline fleet. Fisheries Research. 143:136-142.

Tremblay-Boyer L, and Brouwer S. 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.

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.

Van Houtan KS. 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 KS, and Halley JM. 2011. Long-term climate forcing in loggerhead sea turtle nesting. PLoS One. 6(4):e19043.

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.

Warden ML, and Murray KT. 2011. Reframing protected species interactions with commercial fishing gear: Moving toward estimating the unobservable. Fisheries Research. 110(3):387-390.

WCPFC (Western and Central Pacific Fisheries Commission). 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 (Western and Central Pacific Fisheries Commission). 2021. Science and Scientific Data Functions. Available at: https://www.wcpfc.int/node/29966 Accessed last on: June 20, 2020.

Wegner NC, and Cartamil DP. 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 ER, Myers MC, Flemming JM, and Baum JK. 2015. Shifting elasmobranch community assemblage at Cocos Island—an isolated marine protected area. Conservation Biology. 29(4):1186-1197.

Wilkinson C, and Souter D. 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.

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.

Worm B, Lotze HK, Jubinville I, Wilcox C, and Jambeck J. 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 DJ, Fahey DW, and Hibbard K. 2017. Climate Science Special Report (CSSR). Washington, DC: US Global Change Research Program.

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 https://doi.org/10.1007/s11160-020-09601-3.

Young CN, Carlson J, Hutchinson M, Hutt C, Kobayashi D, McCandless CT, and Wraith J. 2017. Status review report: oceanic whitetip shark (*Carcharhinius longimanus*). Final Report to the National Marine Fisheries Service, Office of Protected Resources. December 2017. p. 170.

# 9 APPENDIX A: EXPOSURE PROFILES

## 9.1 Stressors Not Likely to Adversely Affect Listed Resources

### 9.1.1 Vessel Noise

The proposed action would expose giant manta rays and oceanic whitetip sharks under NMFS' jurisdiction 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 giant manta rays and oceanic whitetip sharks 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 harm or harassment on giant manta rays and oceanic whitetip sharks. Thus, NMFS is reasonably certain this stressor will only have insignificant effects on giant manta rays and oceanic whitetip sharks.

### 9.1.2 Collision with vessels

The proposed action would expose giant manta rays and oceanic whitetip sharks under NMFS' jurisdiction 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.

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, these species remain below the surface of the water the vast majority of the time.

Because giant manta rays and oceanic whitetip sharks 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.

### 9.1.3 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 giant manta rays and oceanic whitetip sharks would be widely scattered throughout the proposed *Action Area*, the small chance that giant manta rays and oceanic whitetip sharks 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 giant manta rays and oceanic whitetip sharks.

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 giant manta rays and oceanic whitetip sharks based on the low number of participants in the fishery, and is therefore insignificant.

### 9.1.4 Vessel Groundings

A vessel grounding could have severe consequences to the giant manta ray or oceanic whitetip shark. Environments were groundings may occur are used for feeding, resting, or various stages of the species respective life cycles. Effects to the overall area and habitat will depend on the specific site, condition, and situation of the vessel, environmental conditions, exposure interval, and many other variables. The amount of habitat potentially affected, given the size of a vessel, is inconsequential in relation to the amount of unaffected habitat available. Therefore, we are reasonably certain habitat loss from vessel grounding will not rise to the scale of harm to individual, and thus is insignificant.

These are vagile species which are extremely unlikely to have any direct exposure to this stressor, as they can vacate the area. Nevertheless, in the highly unlikely event an animal were trapped between the bottom and the sinking vessel; this could result in pinning, broken bones, crushing injuries, or death. The likelihood of this scenario affecting any individual giant manta ray or oceanic whitetip shark given their distribution in the *Action Area*, and the 4.90x$10^{-3}$ chance a vessel may potentially ground and trap an individual is extremely unlikely, and therefore discountable.

## 10 APPENDIX B: FULL RESULTS FOR THE GIANT MANTA RAY ANALYSIS

Table 1. 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 | 0.66 | 0.23 | 0.01 | 0.06 | 0.10 | 0.12 | 0.15 | 0.16 | 0.17 |
| Johnston | 0.52 | 0.23 | 0.01 | 0.10 | 0.14 | 0.15 | 0.17 | 0.18 | 0.19 | 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 | 0.66 | 0.23 | 0.08 | 0.01 | 0.03 | 0.09 | 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 2. 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 | 0.50 | 0.02 | 0.14 | 0.22 | 0.27 | 0.33 | 0.36 | 0.38 |
| Johnston | 1.00 | 0.50 | 0.02 | 0.22 | 0.30 | 0.34 | 0.37 | 0.40 | 0.41 | 0.42 |

| **r=0.019** | **Initial Abundance** | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| 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 | 0.50 | 0.18 | 0.02 | 0.08 | 0.21 | 0.26 | 0.31 |
| 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 3. 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 | 0.05 | 0.35 | 0.54 | 0.66 | 0.82 | 0.89 | 0.95 |
| Johnston | 1.00 | 1.00 | 0.05 | 0.54 | 0.74 | 0.84 | 0.90 | 0.98 | 1.01 | 1.04 |
| 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 | 0.44 | 0.05 | 0.19 | 0.51 | 0.63 | 0.75 |
| 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 4. 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 | 0.45 | 0.04 | 0.28 | 0.36 | 0.40 | 0.42 | 0.46 | 0.47 | 0.48 |
| Johnston | 0.29 | 0.04 | 0.28 | 0.40 | 0.44 | 0.46 | 0.47 | 0.49 | 0.50 | 0.50 |
| Hawaii | 0.49 | 0.09 | 0.22 | 0.37 | 0.42 | 0.45 | 0.46 | 0.48 | 0.49 | 0.50 |
| **95th Percentile Future Annual Interactions; Scenario 2; 10 years** | | | | | | | | | | |
| Palmyra | 1.00 | 1.00 | 0.45 | 0.04 | 0.20 | 0.28 | 0.33 | 0.39 | 0.42 | 0.44 |
| 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 5. 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 | 0.09 | 0.70 | 0.91 | 1.01 | 1.07 | 1.15 | 1.19 | 1.22 |
| Johnston | 0.73 | 0.09 | 0.70 | 1.01 | 1.11 | 1.16 | 1.19 | 1.23 | 1.25 | 1.27 |
| 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 | 1.00 | 0.09 | 0.50 | 0.70 | 0.83 | 0.99 | 1.05 | 1.12 |
| 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 6. 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 | 0.29 | 2.33 | 3.01 | 3.35 | 3.55 | 3.82 | 3.93 | 4.04 |
| Johnston | 1.00 | 0.29 | 2.33 | 3.35 | 3.69 | 3.86 | 3.96 | 4.09 | 4.15 | 4.20 |
| 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 | 0.29 | 1.65 | 2.33 | 2.74 | 3.28 | 3.50 | 3.70 |
| 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 |