6.C.1(2)
146th SSC

Endangered Species Act (ESA) 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 American Samoa 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-2022-02106
PIRO Reference No.:	I-PI-22-2059-DG
Approved By:	MALLOY.SARAH.JOAN.1262526743 Digitally signed by MALLOY.SARAH.JOAN.1262526743 Date: 2022.10.27 09:21:15 -10'00' Sarah Malloy Acting Regional Administrator, Pacific Islands Region
Date Issued:	10/27/22

Contents

List of Figures

List of Tables

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 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 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 United States 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 American Samoa longline (ASLL) fishery, as currently managed under the existing regulatory framework of the Pelagic Fishery Ecosystem Plan (FEP) and other applicable laws. The consulting agency for this proposal is PIRO's Protected Resources Division (PRD). This document represents NMFS' final biological opinion on the effects of the proposed action on 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

---

[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." 50 C.F.R. 222.102. 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.

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 2015 American Samoa longline fishery 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 SFD's biological evaluation (NMFS 2019a), NMFS' status reviews on the giant manta ray and oceanic whitetip shark (Miller and Klimovich 2017 and Young et al. 2017) and the Western Pacific Fishery Management Council (Council) annual stock assessment and fishery evaluation reports for Pacific Island pelagic FEP for 2017 (WPRFMC 2018), and other sources of information as cited herein.

1.1 Consultation History

Prior to 2008, NMFS consulted on the ASLL fishery as part of the Fisheries Management Plan for the Pelagic Fisheries (pelagic FMP) which included the Hawaii shallow-set longline (SSLL), the Hawaii deep-set longline (DSLL), the ASLL, and the regional non-longline pelagic fisheries. Consultations on the pelagic FMP were conducted in 1998, 2001, and 2004. Consultation histories for earlier consultations on the FMP and the ASLL fishery can be found in the 1998, 2001, 2004, 2008, 2010, and 2015 biological opinions.

The 2004 Opinion (NMFS 2004a) included an Incidental Take Statement (ITS) estimating that six sea turtle interactions (cumulatively resulting in one mortality) for green, loggerhead, olive ridley, or hawksbill sea turtle species combined would occur annually in the ASLL fishery and the regional non-longline pelagic fisheries combined. The ITS of six sea turtles was exceeded, and on September 16, 2010, NMFS completed a no-jeopardy Opinion (2010 Opinion; NMFS 2010) under ESA Section 7 on the implementation of Amendment 5 to the Fishery Ecosystem Plan for Pelagic Fisheries of the Western Pacific (Pelagics FEP; WPFMC 2009a). The Council developed Amendment 5, which established measures to reduce interactions between the fishery and green sea turtles by requiring vessels to deploy all longline hooks to fish at least 100 meters (m) deep. Additional regulations included requirements for a minimum of 15 branchlines between each float, and a maximum of 10 swordfish retained on each trip. The Secretary of Commerce approved Amendment 5 and NMFS issued final regulations on August 24, 2011, that were effective on September 23, 2011 (76 FR 52888).

In the 2010 Opinion, NMFS determined that the proposed action is likely to adversely affect green sea turtles, hawksbill sea turtles, leatherback sea turtles, and olive ridley sea turtles, but not likely to jeopardize the continued existence or recovery of these species. NMFS anticipated and authorized a three-year ITS of 45 green sea turtles, one hawksbill sea turtle, one leatherback sea turtle, and one olive ridley sea turtle in the fishery.

SFD reinitiated formal consultation in 2015 because the fishery exceeded the ITS set in the 2010 Opinion for leatherback sea turtles and olive ridley sea turtles and NMFS published a final rule

(79 FR 53852) that listed 20 new species of reef-building corals as threatened under the ESA. NMFS also listed four Distinct Population Segments (DPS; hereinafter referred to as species) of scalloped hammerhead shark as threatened under the ESA (79 FR 38213). The threatened Indo-West Pacific scalloped hammerhead shark is the only scalloped hammerhead species that occurs in the *Action Area* that may be affected by the ASLL fishery. The NMFS 2015 BiOp concluded the action was not likely to jeopardize green, hawksbill, leatherback, olive ridley, and the Indo-West Pacific scalloped hammerhead shark, and not likely to adversely affect the six species of reef-building corals found in the *Action Area*. In the 2015 BiOp, NMFS also included a conference opinion for the green turtle species, which became effective at the time of the final listing in 2016 (81 FR 20058, April 5, 2016). In addition, the Council recommended management changes to the fishery that could result in effects not previously analyzed in prior consultations, including modifications to the American Samoa limited access permit program, an exemption to certain portions of the large vessel prohibited areas (LVPA), changes in retention limits for swordfish, among other potential measures.

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 March 12, 2019, NMFS SFD requested reinitiation of formal consultation to include giant manta ray and oceanic whitetip shark consistent with 50 CFR 402.16.

On March 12, 2019, NMFS PRD initiated consultation.

On October 7, 2021 NMFS SFD published a final rule (86 FR 55743) modifying the ASLL fishery limited entry program consolidating vessel class sizes, permit eligibility requirements, and reducing the minimum harvest requirements for small vessels. The final rule became effective on November 8, 2021.

On August 10, 2022, to ensure Section 7(a)(2) compliance, NMFS determined to complete a supplemental opinion to the 2015 American Samoa Longline Biological Opinion 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 American Samoa longline fishery, the 2015 biological opinion, remains valid and effective until replaced.

1.2 Description of the Proposed Action

NMFS SFD proposes to authorize the pelagic ASLL fishery as currently managed. NMFS regulates the fishery to meet the requirements of the Endangered Species Act (16 U.S.C. 1531 et seq.), Marine Mammal Protection Act (16 U.S.C. 1361 et seq.), Magnuson-Stevens Fishery Conservation and Management Act (16 U.S.C. 1801 et seq.), and other applicable laws. These measures include an exemption to certain portions of the LVPA (86 FR 36239), modifications to the limited access program (86 FR 55743), and retention limits for swordfish (85 FR 71577). This document represents NMFS' biological opinion (Opinion) on the effects of the operation of the ASLL fishery, under the existing regulatory framework.

The United States pelagic longline fishery based in American Samoa is a limited access fishery with a maximum of 60 vessels under the federal permit program. Vessels range in size from less than 40 to over 70 ft. long. The fishery primarily targets albacore tuna for canning in the local

Pago Pago cannery, although the fishery also catches and retains other tunas (e.g., bigeye, yellowfin, and skipjack), and other pelagic management unit species (e.g., billfish, dolphinfish, wahoo, oilfish, opah, and sharks) for sale and home consumption.

Around 2000-2001, the longline fishery began to expand rapidly, principally through the influx of large (vessels greater than 50 ft.) conventional monohull vessels similar to the type used in the Hawaii longline fisheries. Longline fishing employs a type of fishing gear consisting of a mainline that exceeds approximately one nautical mile (6,076 ft.) in length suspended horizontally in the water column, from which branch lines with baited hooks are attached (NMFS 2019a). 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%). Longline fishing involves setting (deploying) a mainline horizontally at a preferred depth in the water column using floats spaced at regular intervals. Crewmembers usually attach three to five radio buoys at regular intervals along the mainline so the line may be easily located for retrieving (hauling) the gear and retrieving line segments if the mainline breaks during fishing operations. Crewmembers clip branch lines to the mainline at regular intervals, and each branch line has a single baited hook. Mainline lengths can be 30 to 100 kilometers (km) (18 to 60 nautical miles (nm)) long. After deploying the mainline, the gear fishes (soaks) for several hours before being hauled. In longlining, a “set” is the deployment and retrieval of a discrete unbroken section of mainline, floats, and branch lines. Deep-set longline gear is set at night and hauled during the day and generally, vessels make one set per day.

The fishing gear in the ASLL fishery is required to fish at least 100 m deep (76 FR 52888) and this is accomplished by requiring a minimum float line and branch line length of 30 m and 10 m respectively, together with a minimum of 70 m of blank mainline (no hooks) between each float line and the first branch line in either direction along the mainline. In addition, there must be at least 15 branch lines between floats (Figure 1). The branchlines are constructed of monofilament (no wire leaders) with 13/0 or 14/0 circle hooks. Mackerel, sardines, and sanma are common bait. An ASLL trip lasts an average of 43 days.

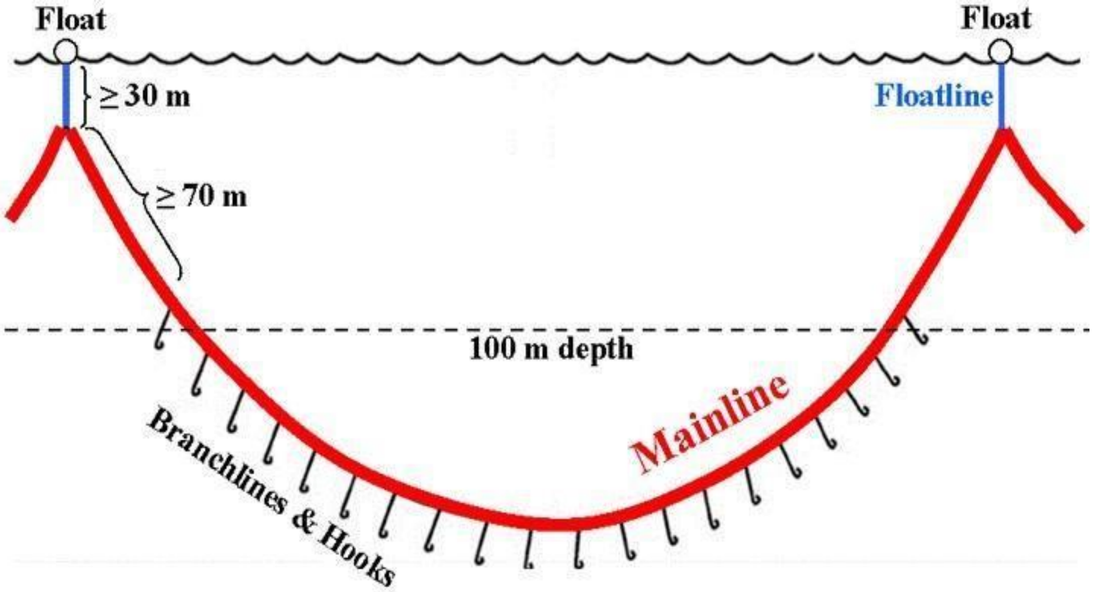


Figure 1. Gear configuration in the ASLL fishery (NMFS 2019a).

The Pelagic FEP and its implementing regulations at 50 CFR 665 contain a number of requirements to prevent and mitigate the effects of the longline fishery on protected species—sea turtles, marine mammals, and seabirds. The specific requirements governing the ASLL fishery are grouped into the following categories, and each category is summarized below:

Fishing Permits and Certificates required on board the vessel:

- American Samoa Longline Limited Access Permit.
- Marine Mammal Authorization Program Certificate.
- High Seas Fishing Compliance Act Permit for fishing on the high seas.
- WCPFC Convention Area Endorsement for fishing on the high seas in the convention area.
- Protected Species Workshop (PSW) certificate.
- Western Pacific Receiving Vessel Permit, if applicable.

Area Restrictions:

- Pelagic fishing vessels 50 ft. and longer are prohibited from fishing within the LVPA, which extends approximately 25-50 nm around the islands of American Samoa. Longliners 50 ft. and longer holding American Samoa Limited Access permits are granted an exemption to fish in portions of this area, as explained below.
- All commercial fishing is prohibited within the boundary of the Rose Atoll Marine National Monument.

Protected Species Workshop (PSW):

- Each year, longline vessel owners and operators (captains) must attend a PSW conducted by NMFS PIRO and receive a certificate. The workshops teach fishermen about mitigation, handling, and release techniques for sea turtles, seabirds, and marine mammals. Fishermen must carry and use specific equipment, and follow certain procedures for handling and releasing sea turtles, seabirds, and marine mammals that may be caught incidentally while fishing.
- A valid PSW certificate is required to renew an American Samoa longline permit.
- The operator of a longline vessel must have a valid PSW certificate on board the vessel while fishing.

Other Mitigation Measures:

- Regulations require all hooks must be placed at least 100 m deep. This is accomplished by requiring a minimum float line length of 30 m, together with a minimum of 70 m of blank mainline (no hooks) between each float line and the first branch line in either direction along the mainline.
- Regulations also prohibit longline vessels from retaining more than 10 swordfish per trip to discourage targeting of swordfish, which are generally found shallower than 100 m where sea turtle interactions are more likely to occur.

Reporting, Monitoring, and Gear Requirements:

- Logbook for recording effort, catch, and other data.
- Transshipping Logbook, if applicable.
- Marine Mammal Authorization Program Mortality/Injury Reporting Form.
- Vessel monitoring system unit.
- Vessel and fishing gear identification.
- Owners and operators of vessels longer than 40 ft. must use longline gear that is configured according to the following requirements:
    - Each float line must be at least 30 m long.
    - At least 15 branch lines must be attached to the mainline between any two float lines attached to the mainline.
    - Each branch line must be at least 10 m long.
    - No branch line may be attached to the mainline closer than 70 m to any float line.
    - No more than 10 swordfish may be possessed or landed during a single fishing trip.

Notification Requirement and Observer Placement:

- Notify NMFS before departure on a fishing trip to declare the trip.
- Carry a fishery observer on board if requested by NMFS; since 2010, NMFS placed observers on approximately 20% or greater percentage of all longline trips annually.

Elasmobranch Handling:

- 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 to this measure, the amount of trailing gear shall be minimal as to cause as little harm as possible.

Recent Changes to Fishery Regulations

Management of the ASLL fishery is dynamic due to the cross-jurisdictional and pelagic nature of the fishery, as well as the participation by the United States in international fisheries management organizations. Management revisions recommended by the Council at the time of initiation, have since been approved by the Secretary of Commerce and are part of the proposed action.

Exemptions from the Large Vessel Prohibited Areas for Longline Vessel

Regulatory provisions of the Pelagic FEP prohibits vessels ≥50 ft. from operating within the United States Exclusive Economic Zone (EEZ) 3-50 nm around Swains Island and generally within 3-50 nm around Tutuila and the Manua Islands (the northern boundary of the LVPA around Tutuila and Manua is approximately 32 nm seaward from the islands). At its 171st meeting (October 2017) the Council recommended exempting eligible American Samoa limited entry permitted longline vessels ≥50 ft. from certain areas of the American Samoa LVPA to improve fishing efficiency (see 50 CFR 665.806). The LVPA longline exempted area would be defined as the area seaward of 12 nm from Tutuila, Manua Islands, Swains Island, and 2 nm around the offshore banks (Figure 2).

On July 9, 2021, following several years of litigation, NMFS published a final rule (86 FR 36239) during the course of the consultation implementing the regulatory exemption which allowed eligible U.S. longline vessels 50 feet and larger to fish in portions of the American Samoa LVPA. NMFS SFD also considered the regulatory exemption as part of this action.

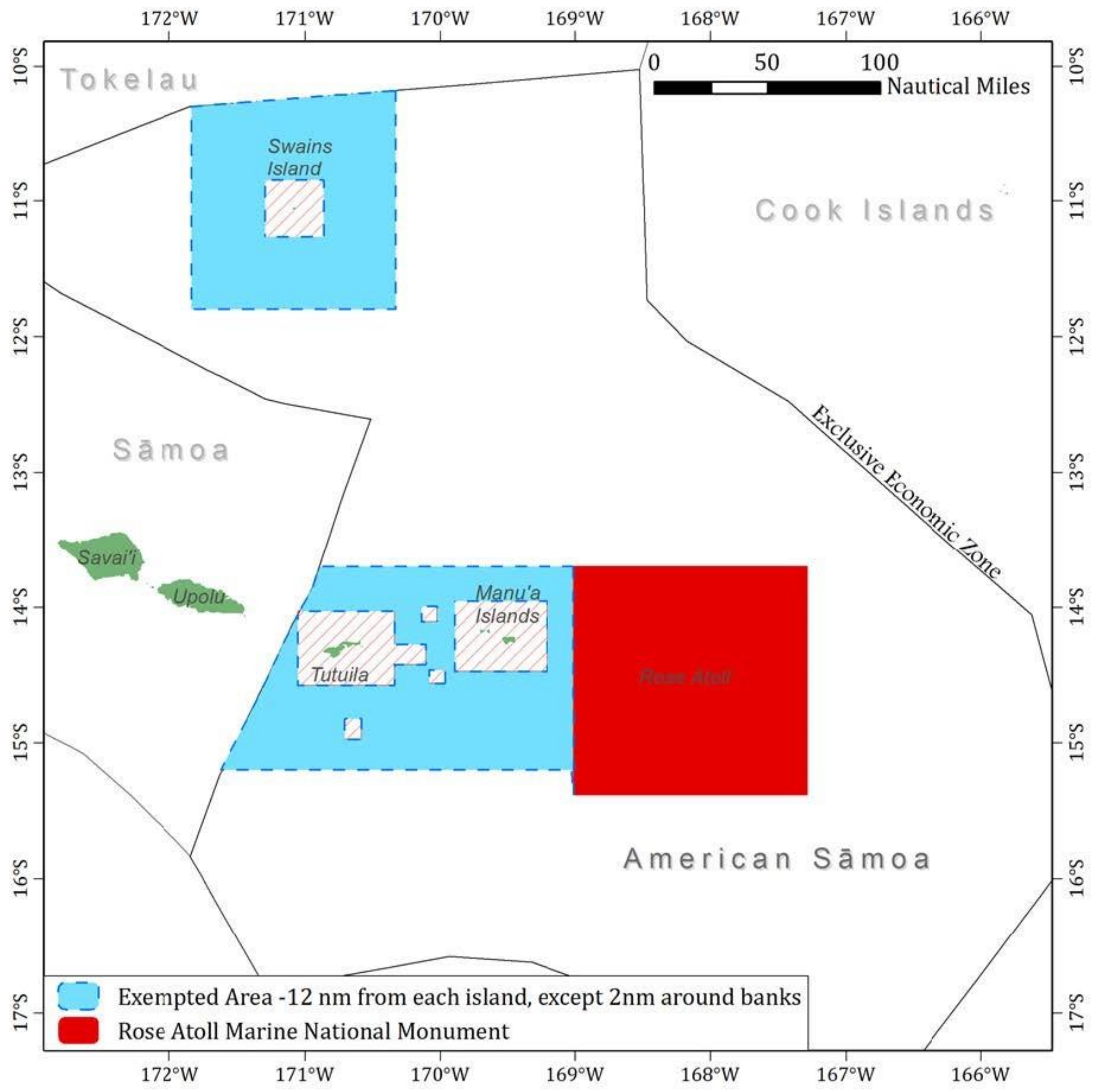


Figure 2. Current LVPA boundaries in waters around American Samoa and the longline vessel exempted areas (NMFS 2019a).

Modification to the Retention Limits for Swordfish

Regulatory provisions of the Pelagic FEP require hooks deployed in the ASLL fishery to be set below 100 meters (m) in depth in order to minimize the incidental catch of green sea turtles. Regulatory provisions also include a trip limit of 10 swordfish per trip to discourage fishermen from setting their gear shallow to target swordfish on the same trip. The swordfish trip limit was instituted as a safeguard to prevent targeting swordfish with shallow-set longline gear and thus reduce the potential for incidental interactions with sea turtles, which typically occur shallower in the water column. The limit mirrored regulations applicable to the Hawaii DSLL longline fishery. At its 172nd meeting (March 2018), the Council took final action and recommended removing the 10 swordfish per trip retention limit, deeming it an unnecessary measure to ensure compliance with the 100 m longline hook setting requirement implemented in 2011, and that it may cause unnecessary waste of swordfish resources. On November 10, 2020, NMFS published a final rule to remove the swordfish retention limit in the American Samoa deep-set longline fishery (85 FR 71577). The action includes NMFS' regulation to remove the 10 swordfish per trip limit (85 FR 71577).

Fishing Participation and Effort

Amendment 11 to the FMP for the Pelagic Fisheries of the Western Pacific Region established the American Samoa Longline Limited Entry Program, and NMFS implemented the program on May 24, 2005 (70 FR 29646). On November 7, 2021, NMFS published a final rule modifying the ASLL fishery limited entry program to consolidate vessel class sizes, modify permit eligibility requirements, and reduce the minimum harvest requirements for small vessels (86 FR 55743). Although the American Samoa longline limited entry program allows for up to 60 vessels, the number of vessels actively participating in the fishery in the last 12 years has ranged from a high of 29 vessels in 2007, to a low of 11 in 2019/2020. In 2007, 29 vessels made 377 trips, 5,910 sets, and deployed approximately 17,524,000 hooks. In contrast, in 2021, 11 vessels made 40 trips, 1,484 sets, and deployed 4,247,000 hooks. This represents an approximate decrease, -76% difference, in effort from hooks deployed in 2007 to 2021. In the last 15 years, there have been annual increases in effort, in 2012, 2015, 2019, and 2021. Otherwise, the fishery statistics across all categories have generally declined. The fishery is strongly seasonal with a low period in the Austral summer between December and April. Typically, vessels experience lower catches in these months and fishing effort is much lower than the rest of the year.

Despite these long-term declines, NMFS believes that the fishery has potential for increased participation, although not necessarily as high as the levels observed from 2007 to 2010. Factors that might influence increased participation include, but are not limited to:

- Increased regional availability of south Pacific albacore tuna resulting from expected improvements in Western and Central Pacific Fisheries Commission (WCPFC) management decisions;
- Decreases in operating costs and improved economic conditions of the fishery; and
- The implementation of an amendment to the Pelagic FEP to exempt longline vessels ≥50 ft. from certain areas of the LVPA, with expected improvements in fishing efficiency. *See* 86 FR 36239 (July 9, 2021).

Accordingly, NMFS believes that the fishery's average effort over the last 5-year period provides a more reliable estimate of the ASLL fleet's likely future effort. Therefore, NMFS conservatively estimates the fishery may operate up to the level seen on average in the last 5-year period (2014-2018); i.e., 18 vessels, 2,269 sets, and 6,369,788 hooks. Table 1 illustrates the number of active vessels and fishing effort (active vessels, trips, sets, and hooks deployed) in the fishery from 2007 to 2018.

Table 1. Number of active vessels and fishing effort in the ASLL fishery, 2008-2021 (2008 02-18 from NMFS 2019a; 2019-2021 from WPRFMC 2020, 2021, 2022).

Year	**Active Vessels**	**Number of Trips**	**Number of Sets**	**Number of Hooks Set**	**Percent Change from Previous Year (Hooks)**
2007	29	377	5,910	17,524,000	---
2008	28	287	4,730	14,372,000	▼ -17.99%
2009	26	175	4,601	14,207,000	▼ -1.15%
2010	26	264	4,496	13,067,000	▼ -8.02%
2011	24	274	3,776	10,767,000	▼ -17.60%
2012	25	195	4,099	11,800,000	▲ 9.59%
2013	22	96	3,324	9,917,000	▼ -15.96%
2014	23	194	2,720	7,601,111	▼ -23.35%
2015	20	202	2,718	7,643,530	▲ 0.56%
2016	20	214	2,412	6,789,400	▼ -11.17%
2017	15	135	2,333	6,623,450	▼ -2.44%
2018	13	68	1,162	3,191,449	▼ -51.82%
2019	17	114	1,695	4,769,000	▲ 49.43%
2020	11	90	1,227	3,401,000	▼ -28.68%
2021	11	40	1,484	4,247,000	▲ 24.88%
2007 to 2021 Percent change (Hooks)=					▼ -75.76%

1.3 Overview of NMFS Assessment Framework

Biological opinions address two central questions: (1) has a Federal agency insured that an action it proposes to authorize, fund, or carry out is not likely to jeopardize the continued existence of endangered or threatened species and (2) has a Federal agency insured that an action it proposes to authorize, fund, or carry out is not likely to result in the destruction or adverse modification of critical habitat that has been designated for such species. 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 ASLL 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 3). Although Figure 3 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 3); 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 3); this is alternatively called a counterfactual because it describes the world as it might exist if a particular action did not occur. Although consultations do not address it explicitly, the future without the project is implicit in almost every effects analysis.

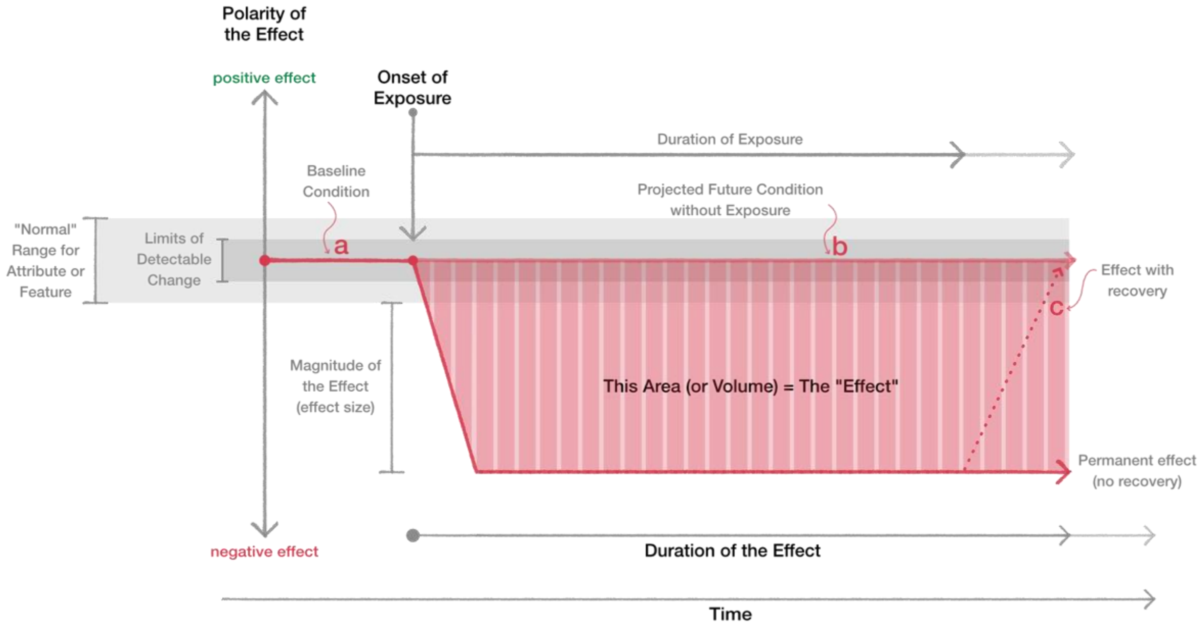


Figure 3. A schematic of the various elements encompassed by the word "effect." The vertical bars in the figure depict a series of annual "effects" (negative changes from a pre-existing or "baseline" condition) that are summed over time to estimate the action's full effect. See text for a more complete explanation of this figure.

As Figure 3 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). The recovery rate allows us to estimate how long it would take for a coral reef and associated benthic communities would take 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.

1.3.1 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 (D.D.C. 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 Listed Resources* sections of biological opinions. Those sections of biological opinions also present descriptions of the number of populations that comprise the species and their geographic distribution. Then NMFS' assessments must consider the status of those populations in a particular *Action Area* based on how prior activities in the *Action Area* have affected them. The *Environmental Baseline* sections of biological opinions contain these analyses; 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 populations 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.

1.4 Application of this Approach in this Consultation

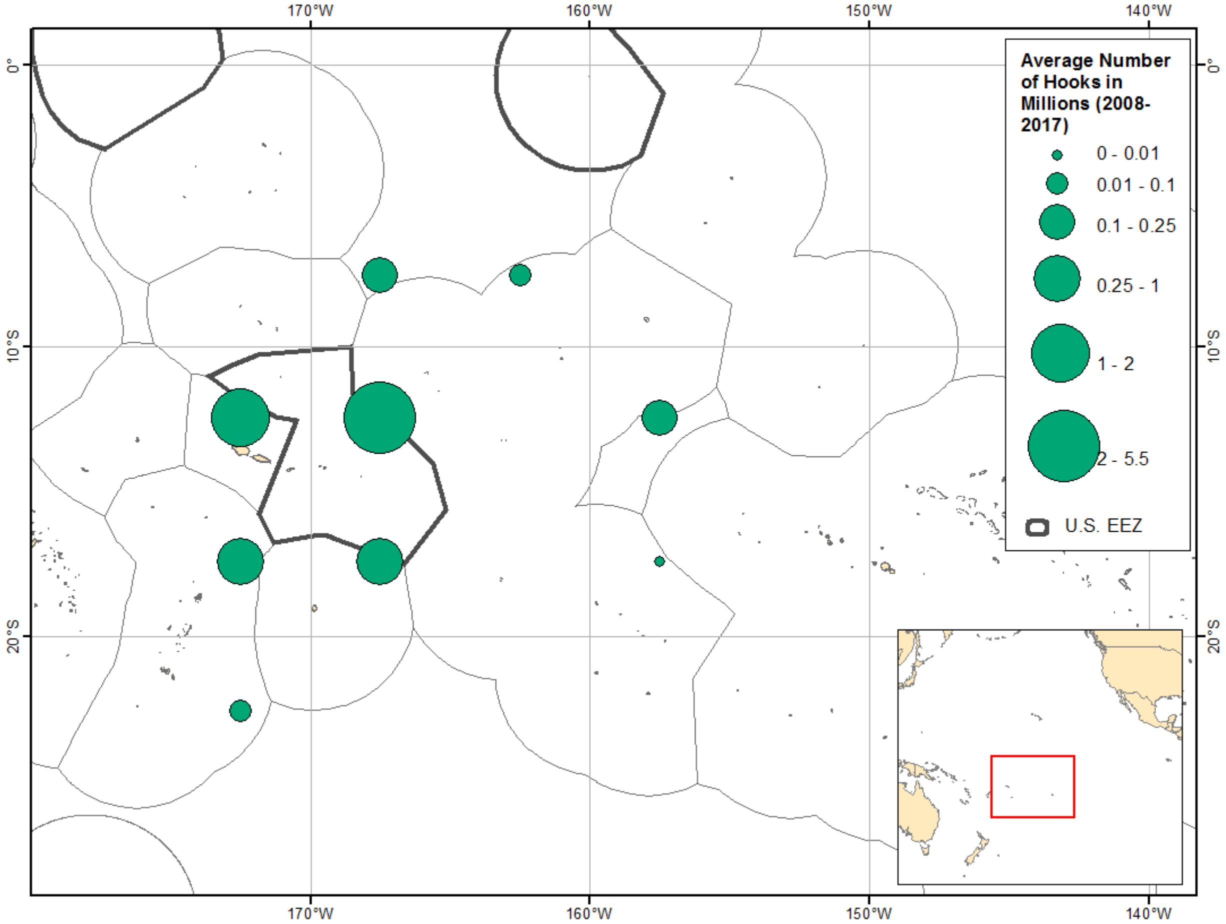
NMFS has identified several aspects of the ASLL fishery and fishing vessels that represent potential stressors to threatened oceanic whitetip sharks and giant manta rays. The term *stressor* means any physical, chemical, or biological entity that can induce a direct or indirect effect on the environment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

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

1.4.1 Action Area

The *Action Area* includes all areas to be affected directly or indirectly by the Federal action and not merely the immediate area involved in the action (50 CFR §402.02). The *Action Area* for this proposed action is based both on where the proposed actions occur, and the area where physical, chemical, and biotic consequences of those actions will occur, including fishing areas and vessel paths during transits.

Accordingly the *Action Area* is identified as all areas where vessels permitted by the ASLL fishery operate, including transiting and fishing. This generally includes the EEZ around American Samoa, the EEZs of countries adjacent to American Samoa and on the high seas. Fishermen may operate in Tokelau to the north, Niue to the south, Cook Islands to the east, and Samoa to the west, as well as areas of the high seas (Figure 4). The fishery operates longline gear at depths of approximately 100–300 m.



Map by: R. O'Conner and R. Walker, NOAA Fisheries, Pacific Islands Regional Office.

Figure 4. Location of sets made by the ASLL fishery from 2008-2017. Note: The figure depicts non-confidential levels of effort, that is, where three or more vessels fished (NMFS 2019a).

1.5 Approach to Evaluating Effects

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

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

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

We used available data to describe the ASLL fishery location and its stressors. Interactions by hooking, entanglements, and landings represent the best data available on the ASLL fishery because it has been collected under approximately 20% observer coverage. 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.

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 ASLL fishery produce responses that range from exposed but not likely adversely affected (such as opportunistic successful depredation of bait or catch) with no hook interaction; interactions with predators and prey; accidentally being hooked and then 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 5 presents our conceptual model of how we translate an interaction between an animal's exposure and the ASLL 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 ASLL 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 oceanic whitetip sharks and giant manta rays.

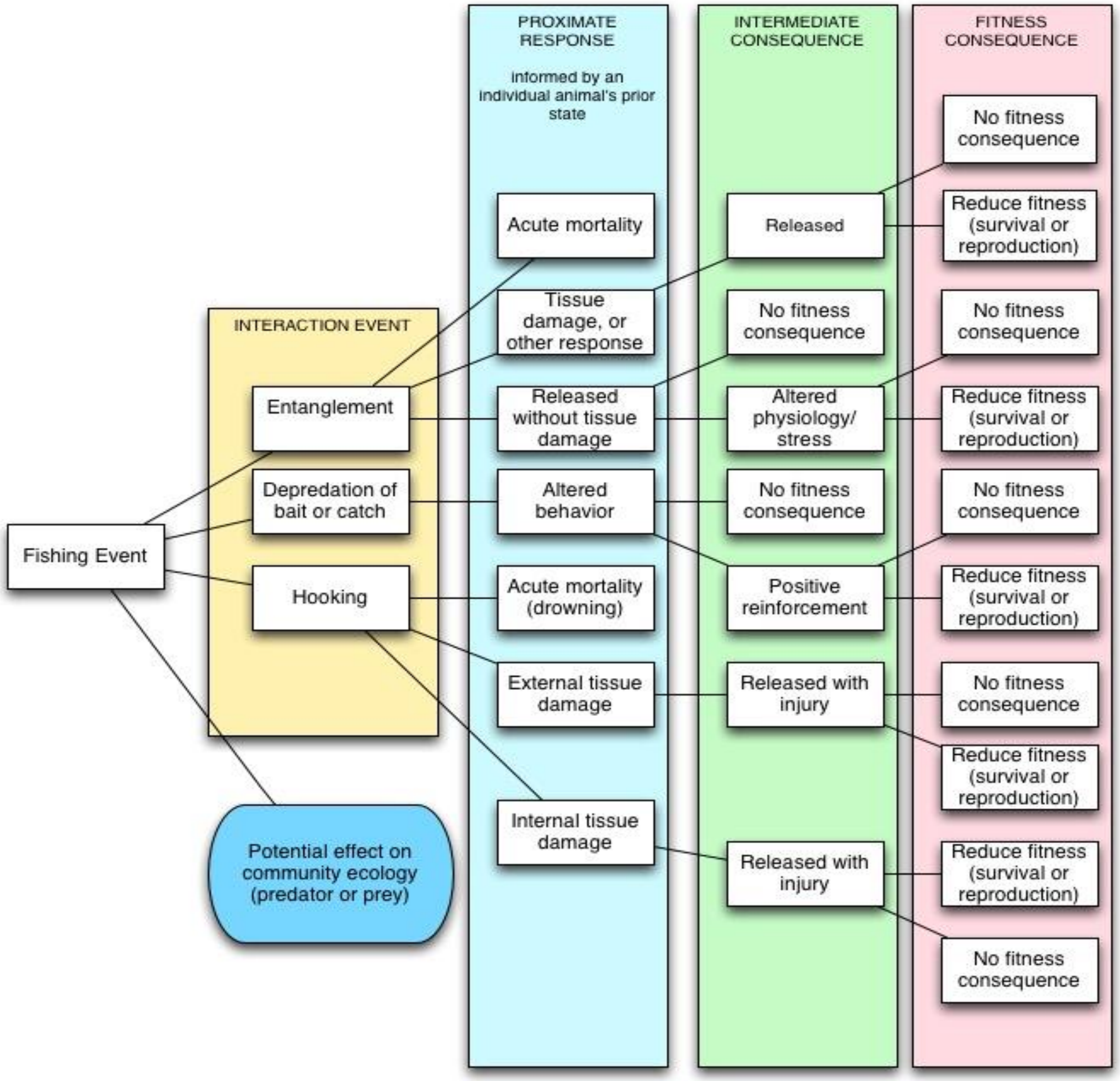


Figure 5. Conceptual model of the potential responses of an animal's exposure in the ASLL fishery and the pathways between exposure (the interaction event the individual animal has with the fishing gear), responses, and potential fitness consequences to the individual.

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 analysis presented in the biological evaluation (NMFS 2019a) used data from 2006 to 2018. We now have observed incidental captures in the ASLL through 2019, hence our analysis of the demographic, spatial and temporal effects of the fishery in this biological opinion focus on observed captures from 2006 to 2019 as these are the years for which we have data.

From there, we developed mortality estimates for both unobserved historic interactions and anticipated future interactions. Species like oceanic whitetip sharks contain a range as the best available science was reviewed to estimate a range for post release mortality. We acknowledge that this is a limitation of the approach, however, this is a robust statistical analysis incorporating available data on observed incidental captures and we anticipate that the results will be representative of interaction rates in the near future (10 to 15 years).

1.5.1 Tipping Points

The Ninth Circuit Court of Appeals has said that in some circumstances, the Services are required to identify a ‘‘tipping point’’ beyond which the species cannot recover in making section 7(a)(2) determinations. Specifically, the Ninth Circuit has said that ‘‘when a proposed action will have significant negative effects on the species’ population or habitat, the duty to consider the recovery of the species necessarily includes the calculation of the species’ approximate tipping point.’’ *Oceana, Inc. v. Nat’l Marine Fisheries Serv.*, 705 F. App’x 577, 580 (9th Cir. 2017) (citing *Nat’l Wildlife Fed’n v. Nat’l Marine Fisheries Serv.,* 524 F.3d 917 (9th Cir. 2008)); *see also Wild Fish Conservancy v. Salazar,* 628 F.3d 513, 527 (9th Cir. 2010) (overturning jeopardy analysis based on purported NMFS failure to determine ‘‘when the tipping point precluding recovery…is likely to be reached’’). Neither the Act nor our regulations state any requirement for the Services to identify a ‘‘tipping point’’ as a necessary prerequisite for making section 7(a)(2) determinations. Section 7(a)(2) provides the Services with discretion as to how it will determine whether the statutory prohibition on jeopardy or destruction or adverse modification 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. See USFWS-NMFS Joint Final Rule, 84 FR 44976 (August 27, 2019). NMFS has considered the Ninth Circuit’s guidance on this topic, and has determined that the circumstances in this consultation do not warrant the identification of tipping points for the subject species.

1.5.2 Climate Change

Future climate will depend on warming caused by past anthropogenic emissions, future anthropogenic emissions and natural climate variability. NMFS’ policy (NMFS 2016) 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 2. 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 2 (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 2. 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've 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 effects the species' ability to feed, reproduce, and carry out normal life functions, including movements and migrations.

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

1.5.3 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., expected number of interactions) 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., expected number of interactions) 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 ensuring that the action satisfies the requirements of Section 7(a)(2) of the ESA, our assessment uses the mean and the 95th percentile which corresponds to the upper end of the 90% credible interval (CI), 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. 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. By selecting the 95th percentile, we can be confident that there is a 95% probability that, given the data, the true population parameter is contained within the credible interval. 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 2012 to 2020 on the time series of estimated annual incidental captures from McCracken (2019a, 2020a) and McCracken and Cooper (2022). In other words, our first 5-year running average is the average of the estimated incidental captures from 2012 to 2016, 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 various ITS'. At this meeting it was acknowledged that an ITS 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, 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.

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 of 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 ASLL since the 1990s.

1.5.4 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 ASLL fishery and giant manta rays and oceanic whitetip sharks collected by as bycatch between 2006 and 2019 (2nd quarter) and (2) the Pacific Islands Fisheries Science Center's analyses of these data (McCracken 2019a). We supplemented the data from these two sources with information contained in SFD's 2019 *Biological Evaluation on the American Samoa Pelagic Longline Fishery*, and the Western Pacific Regional Fishery Management Council's *Stock Assessment and Fishery Evaluation Report* (WPRFMC 2019, 2020, 2021). As a result, these represent the best scientific and commercial data available at the time of consultation.

To support our status assessments, assessments of the expected impacts of the environmental baseline on endangered and threatened species and designated critical habitat, response analyses, and risk assessments, we relied on information from the 2016 *Report of the Rare Events Bycatch Workshop Series* (WPRFMC 2016), and the Bycatch Management Information System (BMIS). 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 literature searches to address a set of specific questions:

1. What is the population structure of oceanic whitetip sharks and giant manta rays this consultation considered? Specifically, we targeted information that would allow us to identify the number of populations and sub-populations that comprise the oceanic whitetip shark and giant manta rays.

2. What effects have been reported for the oceanic whitetip sharks and giant manta rays 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 oceanic whitetip sharks and giant manta rays after an interaction with 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 oceanic whitetip sharks and giant manta rays 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: "Pacific," "*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, we conducted separate searches for data on environmental variables that explain or are correlated with their pelagic distribution, using the keywords "environmental correlates" and "manta ray" or "*Mobulid*" or 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.

Finally, we relied our own count data from NMFS' observer program of past interactions between the ASLL recorded as bycatch from 2006 through 2019. These data represent credible and relevant commercial data. In response to the emerging COVID-19 crisis, and to ensure the safety and protect the health of fishermen, observers, and others, NMFS issued an emergency action on March 27, 2020 (85 FR 17285), extended on September 21, 2020 (85 FR 59199), to provide the authority, on a case-by-case basis, to waive observer coverage. As a result, the observer coverage in 2020 was 2.13% (one trip). In the absence of observer data, McCracken and Cooper (2022) developed an estimator protected species bycatch that relied solely on vessel (name and permit number), port and departure and return dates. As these estimates are not based on observed captures, it is not clear that they represent historical levels of captures, therefore we focus our analysis for this biological opinion on the observer data from 2010 to 2019.

Importantly, observer data are 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 typically 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 ASLL 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.

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.

2 STATUS OF LISTED SPECIES

NMFS has determined that the action that NMFS SFD proposes to authorize, the operation of the ASLL fishery as currently managed, may affect the threatened oceanic whitetip shark and giant manta ray (Table 3).

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

Following the narratives that summarize information on these three topics, each species' narrative provides information on the diving and social behavior of the affected species because that behavior helps assess a species' probability of being incidentally captured by longline fishing gear. 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 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 western central Pacific Ocean, 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.

Abundance of oceanic whitetips appears to be the highest in pelagic waters in a 10° band centered on the equator (Figure 6); 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 that have 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.

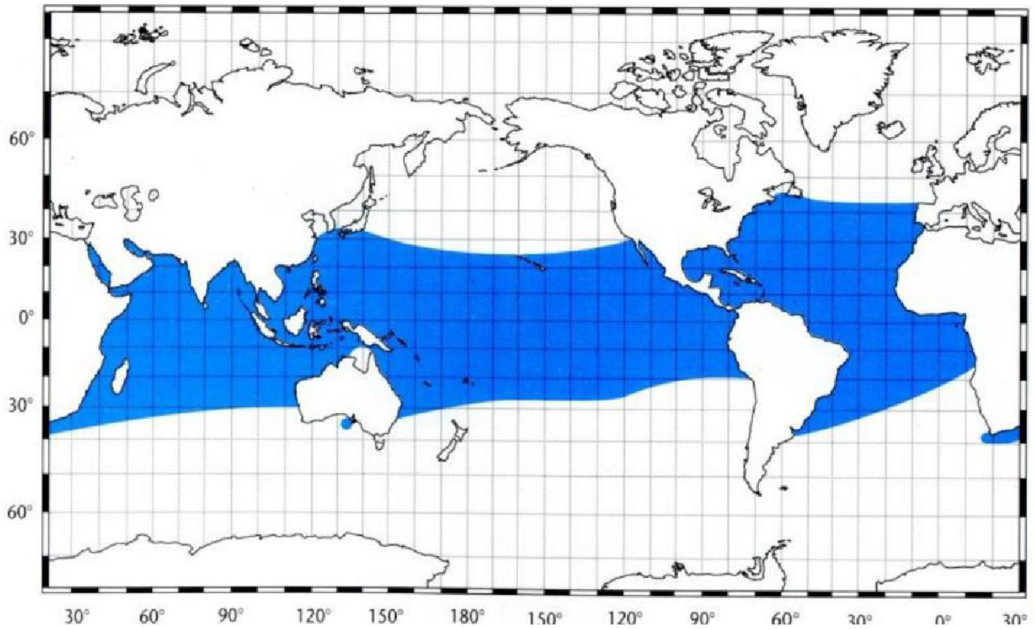


Figure 6. Geographical distribution of the oceanic whitetip shark (Last and Stevens 2009).

Status and Trends

Oceanic whitetip sharks were globally listed 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 (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 by the Food and Agriculture Organization of the United Nations (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 over 648 models in their structural uncertainty grid in Tremblay-Boyer et al. (2019), the authors subsequently estimated the median value of the current total number of individuals in the WCPO (*n*= 775,214; see NMFS 2020). We used this 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 and 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 7; see Table 3 of Bigelow et al. 2022). These results are based on very 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 7). 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 ASLL as currently operated.

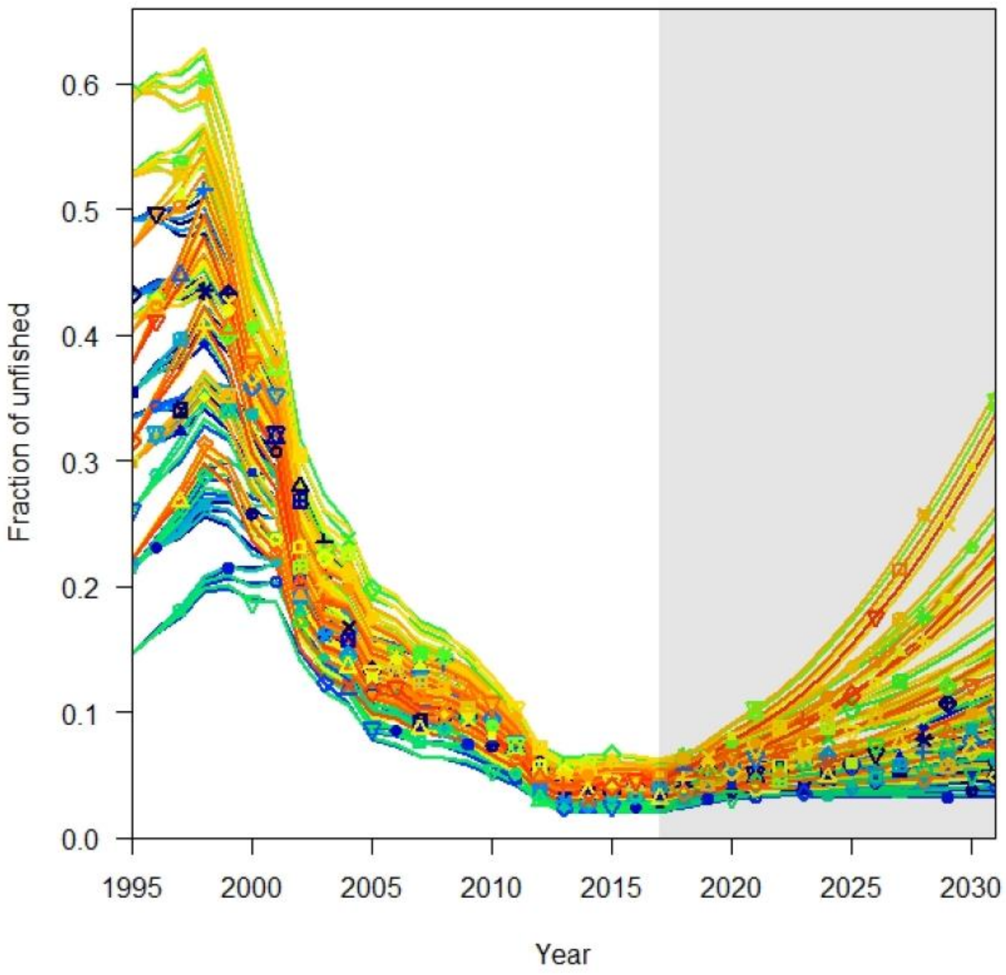


Figure 7. Projected ratios of spawning biomass (projected to 2031) to the equilibrium unfished spawning biomass for WCPO oceanic whitetip sharks with updated at-vessel and post-release mortality rates and the prohibition of wire branchlines and shark line (Figure 7 in Bigelow et al. 2022).

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.

Population Dynamics

Oceanic whitetip sharks are a 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 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 (United States and foreign) that overlap the *Action Area* will be discussed in the *Environmental Baseline*, as appropriate.

Overall, the species has experienced significant historical and potentially ongoing abundance declines in all three ocean basins due to overutilization from fishing pressure and inadequate regulatory mechanisms to protect the species (based on CPUE; Hazin et al. 2007; Lawson 2011; Clarke et al. 2012; Hasarangi et al. 2012; Brodziak et al. 2013; Hall and Roman 2013).

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 incidental 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 (outside of the *Action Area*) that incidentally capture oceanic whitetip sharks include the bottomfish fisheries in Guam, CNMI, and the MHI; the SSLL, and DSLL fisheries. The bottomfish fisheries are estimated to interact with 1 (Guam), 4 (CNMI), and 2 (MHI) oceanic whitetip sharks over a 5 year period, respectively (NMFS 2022a). The SSLL fishery is estimated to interact with 102 oceanic whitetip sharks in a given year (95th percentile) with a total of 875 sharks caught between 2004 and 2018 (NMFS 2019b). The DSLL fishery is estimated to interact with on average, 1,708 (95th percentile: 3,185) oceanic whitetip sharks annually (McCracken 2019b; NMFS 2018a). In total, there were 5,149 observed interactions in the DSLL fishery from 2004 to 2020. When these data are adjusted to account for the percentage of observer coverage, approximately 26,180 oceanic whitetip sharks are likely to have been incidentally captured in the DSLL fishery in the 17-year period from 2004 to 2020 (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021).No interactions have been noted with oceanic whitetip sharks in any West Coast highly migratory species fisheries management plan to date (C. Villafana and C. Fahy pers. comm. to J. Rudolph; March 7, 2019). Lastly, the United States fisheries in Alaska are not expected to overlap with the species range.

Overall, the species has experienced significant historical and potentially ongoing 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 of the Oceanic Whitetip Shark

In this section of this biological opinion, we explained that the oceanic whitetip shark is 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). 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 Secretariat of the Pacific Community observer data from the larger western and central Pacific and the observer data from the Hawaii-based pelagic longline fishery suggest that the stock decline of oceanic whitetip sharks in this portion of its range is not just a localized trend, but rather a Pacific-wide phenomenon (Brodziak et al. 2013). Based on Bigelow et al. (2022), these trends may turn around; however, fishery bycatch, direct harvest and finning continue to be the primary threats to oceanic whitetip sharks.

2.1.2 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; 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; Corturier 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 distinct morphological differences exist 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 27). This map does not show occupancy for giant manta rays in much of the western central Pacific Ocean East of Papua New Guinea and the Solomon Islands, however, based on observed incidental captures of giant manta rays in fisheries throughout this area, the species occupancy is likely distributed throughout this area (Figures 23 and 24 in Tremblay-Boyer and Brouwer 2016; NMFS unpublished data).

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. (2016a) 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 subpopulations 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. 2016b; 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 2016), 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 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 4 below 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. Although the number of regional subpopulations is unknown, the sizes of those identified as regional subpopulations tends to be small, ranging from 600 to 25,250 (CITES 2013; Marshall et al. 2018; Beale et al. 2019; Table 4[2]). CITES (2013) highlights three giant manta ray subpopulations that have been studied and population estimates provided, and counts for more than ten aggregations (Table 4). CITES (2013) also discusses an additional approximately 25 aggregations where

---

[2] We refer the reader to these references if they have questions concerning how subpopulation estimates were determined by distinct authors. Additionally, we note that all putative subpopulations listed in Table 6 occur outside of the *Action Area.*

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, and within the *Action Area* of the United States WCPO purse seine fishery, these areas include nine sites: Indonesia, Papua New Guinea, Federated States of Micronesia, Palau, Solomon Islands, Kiribati, New Caledonia, Fiji and French Polynesia. 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 in 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 6). 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 4 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 (J. Stewart, Manta Trust pers. comm. to A. Garrett, NMFS PRD, 2021). Of the 12 studied subpopulations identified in Table 6, 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 resightings 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 4). 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 CPUE data for giant manta ray observed incidentally captured 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 2021). 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 4. Numbers of recorded individuals and subpopulation estimates of giant manta ray at identified locations adapted from CITES (2013) and updated with supplementary references as specified.

Location	Recorded Individuals	Subpopulation Estimate	Reference
Mozambique	180 - 254	600	Marshall et al. (2009) and pers. comm. cited in CITES (2013); MantaMatcher (2016)
Egypt	60	-	Marine Megafauna (2011) as cited in CITES (2013)
Republic of Maldives	716	-	J. Stewart pers. comm. to A. Garrett citing S. Hilbourne pers. comm. (2021)
Republic of Maldives	378	-	Nicholson-Jack (2020)
Kona, Hawaii (United States)	29	-	Clark (2010)
Thailand	365	-	J. Stewart pers. comm. to A. Garrett citing Manta Trust data (2021)
Raja Ampat, Indonesia	588	1,875	Beale et al. (2019)
Isla de la Plata, Ecuador	~650	1,500	M. Harding, pers. comm. cited in CITES (2013); Sanchez (2016)
Isla de la Plata, Ecuador	2,464	25,250	MantaMatch (2016); Burgess (2017); Marshall and Holmberg 2011 as cited in Burgess (2017); Subpopulation estimate from J. Stewart pers. comm. to A. Garrett (2021)
Brazil	60	-	Laje Viva Institute unpubl. cited in CITES (2013), Luiz et al. (2009)

Mexico (Revillagigedos Is.)	916	-	J. Stewart pers. comm. to A. Garrett citing pers. comm to R. Rubin and K. Kumli (2021)
Mexico (Isla Holbox)	> 200	-	R. Graham, pers. comm. cited in CITES (2013)
Jupiter, Florida (United States)	59	-	Pate and Marshall (2020)
Flower Garden Banks (United States EEZ)	>70	-	Graham and Witt (2008) cited in CITES (2013)
Flower Garden Banks (United States EEZ)	95 (52 proposed *M. cf. birostris*)	-	Stewart et al. (2018)
Japan (Ogasawara Islands)	42	-	Kashiwagi et al. (2010)
Azores, Portugal	31	-	J. Stewart pers. comm. to A. Garrett citing A. Sobral pers. comm. (2021).
Myanmar	201	-	J. Stewart pers. comm. to A. Garrett citing Manta Trust data (2021)
Costa Rica	52	-	J. Stewart pers. comm. to A. Garrett citing Manta Trust data (2021)

Population Dynamics

Giant manta rays are a long-lived, late maturing species with productivity that is among the lowest of all elasmobranchs. Rambahiniarison et al. (2018) estimated that giant manta ray off the Philippine Islands matured at about 9 years and had their first pregnancy at about 13 years of age. Overall, age at maturity estimates range from three to more than 15 years. Giant manta rays typically give birth to only one pup every two to three years, but this can range from annual to 5 years (Notarbartolo-Di-Sciara 1988; Marshall and Bennett 2010; Dulvy et al. 2014; Rambahiniarison et al. 2018). Rambahiniarison et al. (2018) reported that the proportion of pregnant females in subpopulations of giant manta ray in the Philippine Islands averaged about 9 out of every 100 females (9%), but they suggested this might depend on the length of the interpregnancy 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 increasing 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. 2011a).

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 addition to this fishery, giant manta rays have been identified in United States bycatch data for the Hawaii DSLL and SSLL fisheries which occur outside of the *Action Area* in the Pacific. Observer data (20% coverage) for the Hawaii DSLL fishery recorded 56 interactions with giant manta rays from 2004-2018 with an estimated 218 interactions (NMFS unpublished data). Observer data (100% coverage) for the Hawaii SSLL fishery recorded 21 interactions with giant manta rays from 2004-2018 (NMFS 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 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 incidentally 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.

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 for a biological opinion 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.

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 have likely already been impacted by this threat through the pathways described above. Also, the past and present impacts of human and natural factors leading to the status of these two species within the *Action Area* include fishery interactions, vessel strikes, climate change, pollution, marine debris, and entanglement. Although most of the fishing effort in the *Action Area* is primarily in the American Samoa EEZ, fishing has occurred in the EEZs of countries adjacent to American Samoa and on the high seas. Fishermen have operated in Tokelau to the north, Niue to the south, Cook Islands to the east, and Samoa to the west, as well as areas of the high seas (see Figure 4). The environmental baselines for these two species are described below.

Information in this section is summarized from the several past biological opinions on the Hawaii longline fisheries, the United States WCPO purse seine fishery and other international fisheries that occur within the same *Action Area* as the ASLL fishery (NMFS 2004a, 2004b, 2015a, 2021). We also used the 2017 pelagics report (WPRFMC 2018), the status reviews for oceanic whitetip sharks and giant manta rays (Miller and Klimovich 2017; Young et al. 2017), and the other sources as cited in subsequent subsections.

3.1 Threats Posed by a Changing Global Climate

Global annually averaged surface air temperature has increased by about 1.8 ºF (1.0 ºC) over the last 115 years (1901 to 2016) (USGCRP 2017). This period is now the warmest in the history of modern civilization. It is extremely likely that human activities, especially emissions of greenhouse gases, are the dominant cause of the observed warming since the mid-20th century. For the warming over the last century, there is no convincing alternative explanation supported by the extent of the observational evidence (USGCRP 2017). These global trends are expected to continue over climate timescales. The magnitude of climate change beyond the next few decades will depend primarily on the amount of greenhouse gases (especially carbon dioxide) emitted globally. 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 (USGCRP 2017). With significant reductions in emissions, the increase in annual average global temperature could be limited to 3.6 ºF (2 ºC) or less (USGCRP 2017). The global atmospheric carbon dioxide concentration has now passed 400 parts per million, a level that last occurred about three million years ago, when both global average temperature and sea level were significantly higher than today. There is broad consensus that the further and the faster the earth warms, the greater the risk of potentially large and irreversible negative impacts (USGCRP 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 (USGCRP 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 (USGCRP 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).

Marine species ranges are expected to shift as they align their distributions to match their physiological tolerances under changing environmental conditions (Doney et al. 2012). 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.

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. 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.

Because giant manta rays are migratory and considered ecologically flexible (e.g., low habitat specificity), they may be less vulnerable to the impacts of climate change compared to other sharks and rays (Chin et al. 2010). However, as giant manta rays frequently rely on coral reef habitat for important life history functions (e.g., feeding, cleaning) and depend on planktonic food resources for nourishment, both of which are highly sensitive to environmental changes (Brainard et al. 2011; Guinder and Molinero 2013), climate change is likely to have an impact on the distribution and behavior of these animals. Decreased access to cleaning stations may negatively impact the fitness of the giant mantas by hindering their ability to reduce parasitic loads and dead tissue, which could lead to increases in diseases and declines in reproductive fitness and survival rates.

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 Subsistence Hunting/Human Consumption

Some species, despite their protected status under the ESA, continue to be killed for consumption either in subsistence hunting, or to be included in the food or traditional medicine market place, including elasmobranchs.

As noted above not many directed fisheries for sharks exist, however, the demand for shark products, including liver oil, hides, meat, teeth, jaws, and especially fins, has resulted in sharks caught as bycatch in other fisheries being retained rather than released. Recent measures have likely decreased the intensity of this threat. In 2008, the WCPFC adopted CMM 2008-2006 (most recently replaced with CMM 2010-07), calling for commission members, cooperating non-members, and participating territories to develop National Plans of Action or other relevant policies for sharks that include measures to minimize waste and discards from shark catches and encourage the live release of incidental catches of sharks, include key shark species, such as oceanic whitetip and scalloped hammerhead sharks, in their annual reporting, and limit the ratio of fins to full shark carcasses for retained sharks. The overall effectiveness of this measure in reducing shark finning and increasing live releases is unclear (Clarke et al. 2016).

Protections for oceanic whitetip sharks were further enhanced in response to a recent population assessment report indicating that oceanic whitetip sharks are overfished (Rice and Harley 2012). The WCPFC adopted CMM 2011-04 (most recently revised in CMM 2019-04), prohibiting the retention of oceanic whitetip sharks, in whole or in part, for vessels flying the flags of and vessels under charter arrangements to participating commission members, non-members, and territories in the fisheries covered by WCPFC. The measure further requires the release of oceanic whitetip sharks that are 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. Additionally, NMFS published a final rule requiring the removal of fishing gear from any oceanic whitetip shark caught in all of the region's domestic longline fisheries (87 FR 25153; 04/28/2022).

In the South Pacific, overall longline effort trend south of 20°S indicates that average fishing effort over the most recent 10 years has been substantially higher than in previous years (Figure 9).

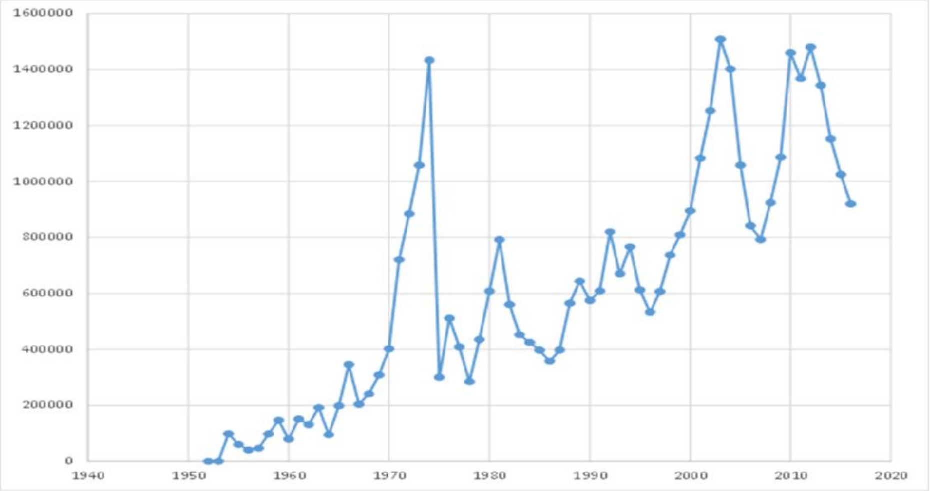


Figure 9. Pacific longline effort levels south of 20˚S in hundred hooks, 1952–2016 (SPC 2019).

Similar to sharks, giant manta rays are valued for their meat and gill rakers, leading them to be retained rather than released when caught as non-target bycatch in fisheries. In addition, many countries within the *Action Area* have artisanal fisheries that target mobulids (Croll et al. 2016), with mobulids traditionally targeted for their meat; however, since the 1990s, a market for mobulid gill rakers has increased. We have not found sufficient information to understand the full effect of these fisheries on the species.

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). This commercial trade of manta ray products, particularly gill plates, coupled with emerging technological advances (e.g., motorized vessels) and an increase in the number of boats in the fishery, greatly increased fishing pressure and harvest of manta rays in the 1990s and 2000s (Dewar 2002). In Lamakera, Indonesia, one of the main landing sites for mobulids, and particularly manta rays, Dewar (2002) estimates that the total average harvest of "mantas" during the 2002 fishing season was 1,500 (range 1,050- 2,400), a significant increase from the estimated historical levels of around 200-300 mantas per season; however, Lewis et al. (2015) note that this estimate likely represents all mobulid rays, not just mantas. Fishermen from Lamalera, whose fishing grounds overlap with the Lamakera fishing fleet, reported landings of around 200-300 per season but noted that very few mantas were caught from 1998-2001, and attributed the low catch to the presence and competition of Taiwanese fishing ships, which also began fishing off Lamalera in large numbers in the 1990s (Barnes 2005).

Although fishing for manta rays was banned within the Indonesian 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 (Marshall and Conradie 2014). It is unlikely that fishing effort and associated utilization of the species will significantly decrease in the foreseeable future as interviews with fishermen indicate that many are excited for the new prohibition on manta rays in Indonesian waters because it is expected to drive up the price of manta ray products, significantly increasing the current income of current resident fishermen (Marine Megafauna Foundation 2016 as cited in Miller and Klimovich 2017).

In the Philippines, fishing for manta rays mainly occurs in the Bohol Sea and dates back to at least the late 19th century. By 1997, there were 22 active mobulid ray fishing sites in the Bohol Sea (Acebes and Tull 2016). In Pamilacan, 18 boats were fishing for mobulids in 1993, increasing to 40 by 1997, and in Jagna, at least 20 boats were engaged in mobulid hunting in the 1990s (Acebes and Tull 2016). Catches from this time period, based on the recollection of fishermen from Pamilacan and Baclayon, Bohol, were around 8 manta rays (for a single boat) in 1995 and 50 manta rays (single boat) in 1996 (Alava et al. 2002). Although a ban on hunting and selling giant manta rays was implemented in the Philippines in 1998, this has not seemed to impact the mobulid fishery in any way. In Pamilacan, there were 14 mobulid hunting boats reported to be in operation in 2011 (Acebes and Tull 2016). In the village of Bunga Mar, Bohol, there were 15 boats targeting mobulids in 2012, and out of 324 registered fishermen, over a third were actively engaged in ray fishing (Acebes and Tull 2016). Due to their size, the boats can only catch a maximum of 4 giant manta rays per trip (Acebes and Tull 2016). Acebes and Tull (2016) monitored the numbers of manta rays landed at Bunga Mar over a period of 143 days from April 2010 to December 2011 (during which there were around 16-17 active fishing boats targeting mobulids), and in total, 40 giant manta were caught. In 2013, records from a single village (location not identified) showed over 2,000 mobulids landed from January to May, of which 2% (n= 51 individuals) were giant manta (Verdote and Ponzo 2014). As there is little evidence of enforcement of current prohibitions on manta ray hunting, and no efforts to regulate the mobulid fisheries, it is unlikely that fishing for mantas will decrease in the future, particularly since fishing is the primary source of income for the people of Jagna and Pamilacan and a "way of life," with mobulid fishing providing the greatest profit (Acebes and Tull 2016).

Opportunistic hunting of manta rays has been reported in Tonga and Micronesia (B. Newton and J. Hartup pers. comm. cited in CMS 2014). While the extent of this fishing and associated impacts on the local manta ray populations are unknown, given the reportedly opportunistic nature of the practice, it is unlikely that fishing pressure is significant on the species.

3.3 Fisheries and Effects on ESA-listed Species

A significant factor affecting all species considered in this Opinion within the *Action Area* are United States and international fishing fleets other than the ASLL fishery. In this section, we present an overview of the different types of fisheries, associated level of effort, and their effect on ESA-listed species. The following fisheries are in the *Action Area*:

1. *The American Samoa-based bottomfish (NMFS 2015b, 2022) and pelagic troll fishery* (NMFS 2009) are managed under the United States Pelagics Fisheries Management Plan. The American Samoa bottomfish Biological Opinion found the action is not likely to adversely affect oceanic whitetip sharks or giant manta rays (NMFS 2022a). We note, other United States bottomfish fisheries adversely affects oceanic whitetip sharks, but do not overlap with the *Action Area*, thus were discussed in the *Status of the Species* as appropriate.
2. *Foreign. WCPO Longline Fisheries* - There were roughly 5.5-7.5 million hooks deployed annually by the international longline fleet operating in the WCPO between 2008 and 2015 (Figure 10; WCPFC 2021). There are two types of vessels: (1) large distant-water freezer vessels that undertake long voyages (months) and operate over large areas of the region; and (2) smaller offshore vessels with ice or chill capacity that typically undertake trips of about one month (like the Hawaii deep-set longline fleet). The total annual number of longline vessels in the western central Pacific region has fluctuated between 3,000 and 6,000 for the last 30 years. The four main target species are yellowfin, bigeye, and albacore tuna, and swordfish. The WCPO longline fishery as a whole was observed at rates between 1.8-3.0% from 2013–2017.
3. *United States and foreign WCPO Purse Seine Fisheries* - Purse seine is conducted in two management structures: United States purse seine managed under the Pelagics Fisheries Management Plan, and foreign purse seine. Between 2008 and 2015, there were approximately 68,000 to 142,000 annual sets made by the international purse seine fleet operating in the WCPO exclusive of sets made by the United States fleet (Figure 11; WCPFC 2021). While the United States WCPO purse seine fishery is typically observed at 100%, under the SPTT, the observer program is administered by FFA. and therefore NMFS has not consistently received 100% of the observer data. From 2013 to 2017, NMFS received observer data from between 44% and 69% of the fishing effort. Between 2008 and 2015, there were approximately 63,419 sets by the United States WCPO purse seine fleet operating in the WCPO (NMFS unpublished data). Observer

coverage for the fishery from 2020 to 2022 has been substantially lower than 100% due to the COVID-19 pandemic.

Purse seine can be conducted by setting the net, or purse, below objects that are known to concentrate fish – these objects may be natural or manmade and are called Fish Aggregating Devices (FADs). Because there are known interactions between ESA listed species and FADs, we present that information as a subsection to the section on Purse Seine. *FADs used in Purse Seine Fisheries* (see page 72) -Purse seine sets are categorized as either "unassociated" or as "associated" when sets are made on FADs. These are can be man-made, either anchored (aFADs) or drifting (dFADs) FADs, or they can be sets made on natural objects such as logs, dead whales, or other floating debris (Hare et al. 2015). Up until the mid-1990s, unassociated sets accounted for the majority of purse seine fishing activity. Since that time sets have been generally split somewhat evenly between the two types; the percentage of associated sets has ranged from 41-67% in the WCPO between 2007 and 2015. Exposed surfaces of FADs are frequently covered with netting, and, beneath the surface structure, FADs generally have submerged "appendages". The depth and extent of appendages are used to control the drifting speed of dFADs, to provide bio-fouling opportunities, and shelter and shade associated non-tuna finfish, all of which are felt to enhance tuna aggregation. The depth of those appendages can vary, from 10 to 120 m depth, depending on fleet, ocean and season, with a tendency for that depth to increase in recent years (Pilling et al. 2017).

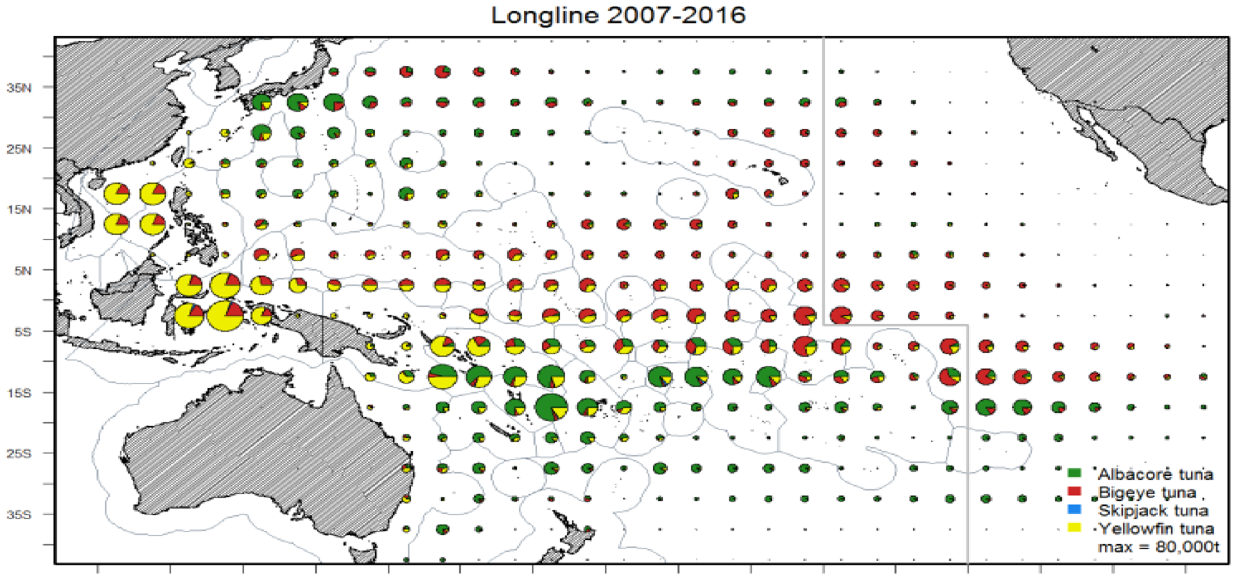


Figure 10. Spatial distribution of tuna catch for the West Central Pacific longline fleets (from Brouwer 2017).

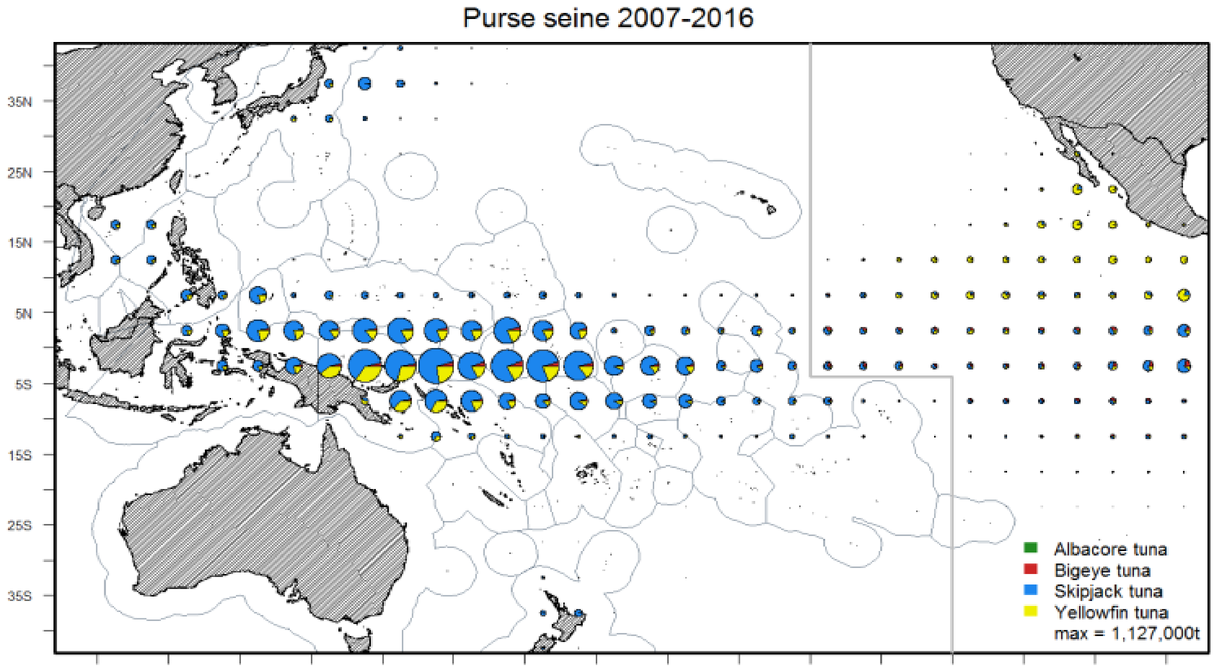

Purse seine 2007-2016

Figure 11. Spatial distribution of tuna catch for the West Central Pacific purse seine fleets (from Brouwer 2017).

We present the baseline effects of these three fishery practices by species.

The American Samoa-based bottomfish and pelagic troll fisheries

There are no expected interactions from these United States fisheries with elasmobranchs as no interactions have occurred to date with these species. Therefore, these fisheries will not be discussed further.

Foreign WCPO longline Fisheries

In the western Pacific, annual reports provided to the Commission from the member countries, lack species-specific data for many of the elasmobranchs. Median shark and ray catch estimates for some species were modeled by Peatman et al. (2018) for multiple regions of the WCPFC using longline observer data. Median shark and ray catch estimates were modeled by Peatman et al. (2018) for multiple regions of the WCPFC using longline observer data. Table 5 displays statistics relevant to the oceanic whitetip shark South of 10°S from 2003 to 2017 which includes a portion of the purse seine fisheries' *Action Area* and Table 5 is an excerpt from Peatman et al. (2018). It should be noted that these data already include United States data and the proportion of overlap from other international fisheries with the *Action Area* is unknown. Additionally, caveats apply as observer coverage range from 1 to 4.5% of the total hooks set, particularly north of 10°N; and has a wide confidence interval for key shark species (Peatman et al. 2018). We accessed the public domain bycatch data served by the WCPFC (2021) to estimate the numbers of annual interactions and mortalities of oceanic whitetip sharks from 2013 to 2018 (Table 6).

Table 5. Median oceanic whitetip shark catch estimates (1,000 individuals) between South of 10°S (Peatman et al. 2018).

Oceanic Whitetip Shark (reported in 1,000 individuals) South of 10°S	
Year	**Individuals**
2003	**32.6**
2004	**25.8**
2005	**2.03**
2006	**18.3**
2007	**15.5**
2008	**14.4**
2009	**18.1**
2010	**25.2**
2011	**21.5**
2012	**17.2**
2013	**12.4**
2014	**10.1**
2015	**10.7**
2016	**11.1**
2017	**10.3**
Total	**263.5**

Table 6. Mean and (95% confidence interval) of annual numbers of oceanic whitetip reported captured/killed by participating countries reporting catch data to the Western and Central Fisheries Commission for longline fisheries operating in the *Action Area* from 2013 to 2020. Data were reported in 5° x 5° bins, and data were restricted to those that overlap with the *Action Area* as closely as possible (WCPFC 2021).

Species	Observed Interactions	Observed Mortalities	Estimated Interactions	Estimated Mortalities	Total Est Captures 2013 to 2020	Total Est Mortalities 2013 to 2020
Oceanic Whitetip Shark	748 (636 – 860)	181 (159 – 202)	27,782 (25,110-30,455)	6,895 (5,750 – 8,038)	222,257	55,161
Percent Observer Coverage (2013-2018): 2.8% (2.2 – 3.4%)						

Rays are currently not considered key bycatch species, and therefore have not been included in the WCPFC bycatch summaries discussed above. However, their bycatch rates were summarized by Tremblay-Boyer and Brouwer (2016; Table 7).

Table 7. Numbers of observed giant manta ray interactions reported by participating countries to the WCPFC for longline fisheries operating in WCPO (observed interaction data from Tremblay-Boyer and Brouwer 2016).

Year	Observed Interactions
2012	96
2013	134
2014	99
2015	19
Total	348

Peatman et al. (2018), summarize observer data from 2003 to 2017 to estimate the catch and catch composition of the longline fisheries of the western and central Pacific Ocean. Observer coverage levels in the region are generally less than 5% and observer coverage can be expressed in a variety of units (e.g., trips with observers on board, hooks with observer onboard, hooks observed, Peatman et al. 2018). Observer coverage over the whole Convention Area (Figure 12) tends to be consistent from 2003 – 2010 (1 to 1.5%) before reaching a maximum of ~4.5% in 2013 and then varying between 2 and 4% up to 2017 (Peatman et al. 2018). Since the United States is a cooperating commission member, the data from the American Samoa, Hawaii DSLL and SSLL fisheries is included in this summarized observer data report.

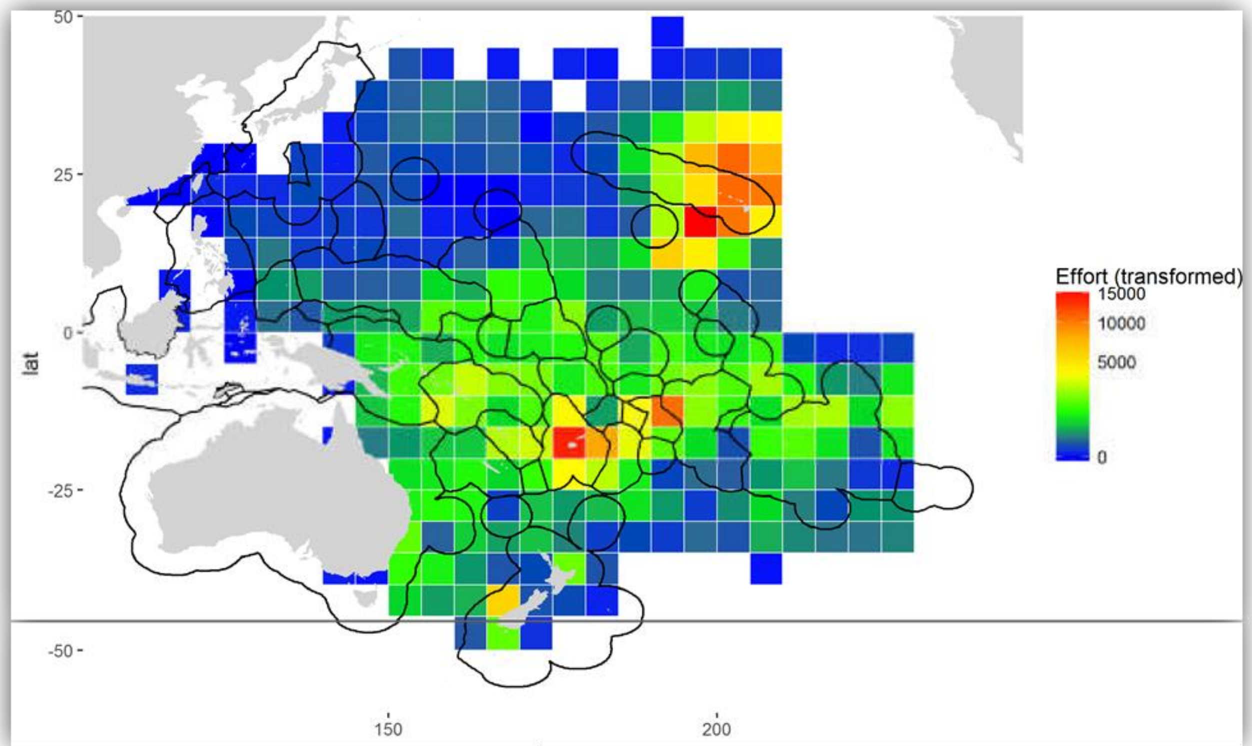


Figure 12. Observed effort in number of hooks (square root transformed) for longliners during the 2003-2017 time period in the WCPFC convention area (Peatman et al. 2018).

Longline observers record catch data specific to each individual caught. As such, the natural catch unit for the estimation of catches is numbers of individuals. Currently, the observers are not instructed to distinguish between mantas and mobulids. In Figure 13, the total number of manta and mobula ray interactions documented by observers from 2003-2017 totals 1,800.

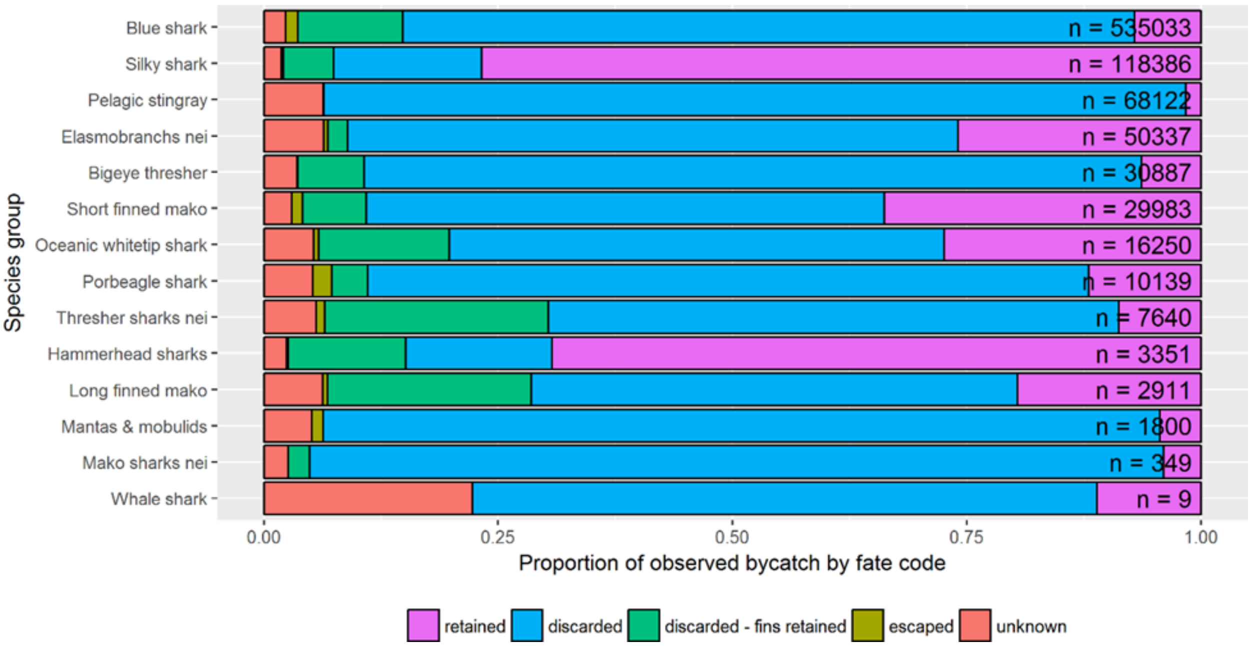


Figure 13. Recorded fate of observed sharks and rays catch by species/species group, as a proportion of total observed catch (number of fish) for the species/species group in the longline fisheries. The number of records is provided (Peatman et al. 2018).

As evident in the above figure, very few mantas or mobulas were retained, the majority incidentally captured were discarded. In Figure 14 below, of the mantas and mobulas discarded, the majority were discarded either alive-healthy-injured or alive-unknown. Many were discarded in an unknown condition while a smaller proportion were discarded alive-dying or dead.

According to Tremblay-Boyer and Brouwer (2016), giant manta rays are observed at a rate of 0.001-0.003 individuals per 1,000 hooks in the longline fisheries. The longline standardized CPUE data, while short, provides a more accurate representation of the species' abundance trend (due to traditional focus on species in longline observer programs) and indicate that giant manta rays are observed less frequently in recent years compared to 2000-2005 (Tremblay-Boyer and Brouwer 2016).

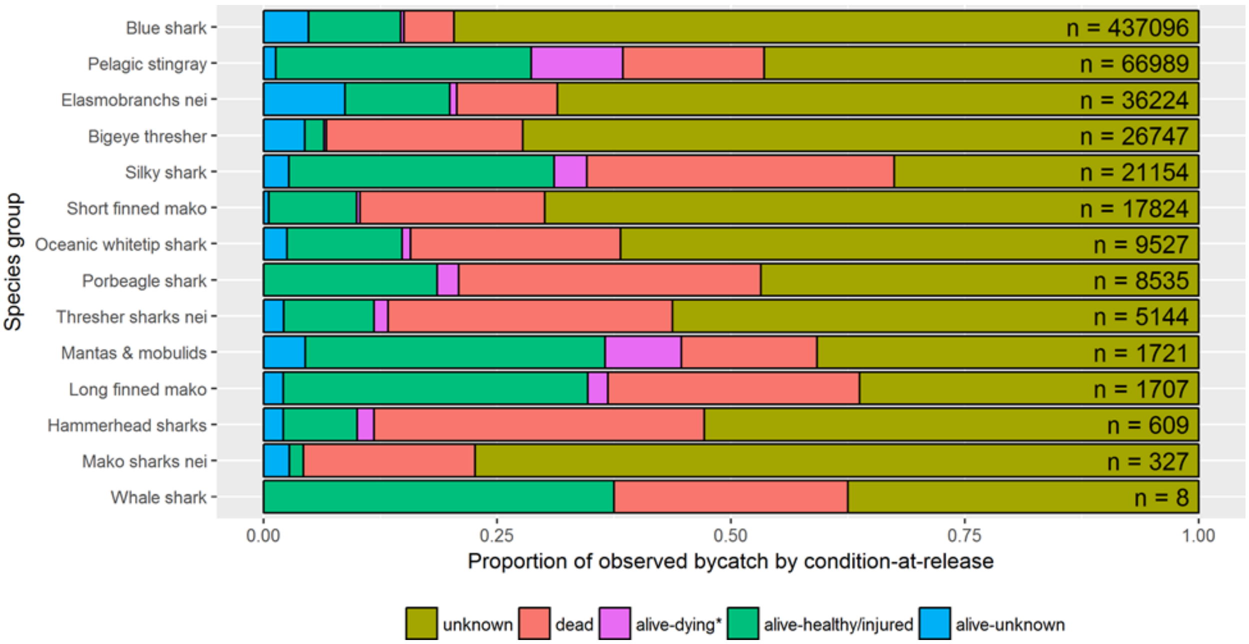


Figure 14. Recorded condition at release of observed sharks and rays catch by species/species group, as a proportion of total observed catch (number of fish) for the species/species group in the longline fisheries. The number of records is provided for each species/group. Note – alive-dying* is individuals that were alive but considered unlikely to survive (Peatman et al. 2018).

United States WCPO Purse Seine

The observed interactions, mortalities and anticipated number of interactions and mortalities of oceanic whitetip sharks and giant manta rays that occurred from 2008-2015 in United States WCPO purse seine fishery are listed below. As noted above, NMFS (2021) calculated annual interaction rates by dividing the number of observed incidental captures by the number of observed sets. They then conducted a nonparametric bootstrap of the annual interaction rates from 2008 to 2018 to estimate the 95th percentile of the distribution. They also calculated a maximum 5-year running average of the interaction rates, and used those along with the anticipated number of sets (3,100 per year) to estimate the maximum 5-year running average for captures. Separately, NMFS developed predictions of annual bycatch using Bayesian statistical inference techniques (NMFS unpublished data). The annual bycatch rates resulting from this analysis were used to estimate the total number interactions with the United States WCPO purse seine fishery from 2008 to 2018 for oceanic whitetip sharks, and 2010 to 2018 for giant manta rays.

Oceanic Whitetip Shark

In the United States WCPO purse seine fishery, sharks may not be boarded for a number of reasons, including poor weather, the shark's size, condition at landing, efficiency of maintaining fishery operations, safety and stress considerations for the animal, and for the crew's safety. However, sharks may be boarded if the crew is unable to identify a shark's presence within the catch. Currently, demographic data is not collected in the purse seine fishery. At this time we cannot conclude if any specific size, age, or gender classes of oceanic whitetip sharks are being caught in this fishery. Table 8 below lists the number of oceanic whitetip interactions and mortalities and Table 9 lists the anticipated number of exposures and mortalities.

Table 8. Observed interactions of oceanic whitetip sharks described by the observer data from 2008-2018 (NMFS 2021).

Year	Observed Interactions	Mortalities
2008	232	95
2009	188	78
2010	279	115
2011	214	98
2012	253	103
2013	170	70
2014	207	98
2015	187	62
2016	180	72
2017	152	58
2018	222	76
Total	2284	925

Table 9. Maximum 5-year running average and 95th percentile of oceanic whitetip shark posterior estimated exposure and mortality estimates in terms of anticipated annual interactions with fishing gear (NMFS 2019c, 2021).

	Maximum 5-year Running Average	**95th percentile**
Predicted Annual Exposures	102.8	160
Predicted Mortality Estimates	87	135

Giant Manta Ray

Due to their large size and removal practices, giant mantas are injured or killed when separated from the catch or during the process to return them to the ocean. Giant mantas were not consistently recorded by observers across the whole observer data set used. Observers are instructed to document interactions with rays by species when possible, but observations can include two unidentified groupings which may include giant manta rays; "*Mobula* nei" and "Sharks, rays, skates, etc. nei". The observer may not be able to distinguish whether the ray species is a giant manta ray. In this situation, observers are instructed to use the classification "*Mobula* nei". United States purse seine vessels in the WCPO have carried observers on all trips (100% observer coverage) since 2010; prior to 2010, the target observer coverage was 20% of purse seine trips.

NMFS International Fisheries Division (IFD) combined the two categories (Mobula nei and Sharks, rays, skates, etc. nei) for the purpose of that analysis (NFMS 2021). Through the Bayesian analysis approach of Martin et al. (2015) IFD estimated the number of interactions and mortalities for all years (Table 10). In addition, IFD used the ratio of confirmed giant manta rays and mobula (devil ray) to estimate the potential giant manta rays included in the *Mobula* spp. category. The estimates of the interaction rates, across different interaction classes were used to estimate the number of interactions that occurred in fishing activities that were not observed, or for which we did not have complete observer data in the 2008 – 2018 dataset. NMFS (2021) estimates that there is at least a 75% misidentification rate for giant manta ray in the United States WCPO purse seine fishery and applied this misidentification rate to determine the anticipated incidental captures for giant manta ray (Table 11).

Table 10. Number of estimated interactions* with giant manta rays from 2008 to 2018 with estimated number of mortality events based on the observed data (NMFS 2021).

Year	**Observed Interactions**	**Mortalities**
2008	299	299
2009	335	335
2010	503	503
2011	466	466
2012	558	558
2013	300	300
2014	473	473
2015	277	277
2016	293	293
2017	172	172
2018	272	272
Total	3,551	3,676

*Observed interactions are included in this evaluation for years when the data is present, and includes an estimate of those unidentified animals which would be expected to be giant manta rays.

Table 11. Maximum 5-year running average and 95th percentile of giant manta ray posterior estimated exposure and estimated mortality in terms of anticipated annual interactions with fishing gear (NMFS 2021).

Rate	Maximum 5-year Running Average	95th percentile
Predicted Annual Exposures (no misidentification)	187.4	199
Predicted Mortality Estimates (no misidentification)	179.9	191
Predicted Annual Exposures (75% misidentification)	47.0	50
Predicted Mortality Estimates (75% misidentification)	45.2	48

Foreign purse seine fisheries in the WCPO/FADs

The international purse seine fishery in the WCPO operates in a tightly concentrated area in the equatorial band, with the highest catches in the zone 5°N - 10°S. Like the IATTC, even though these fisheries occur outside of the *Action Area*, past and on-going effects of these fisheries have led to the current status of the oceanic whitetip shark and giant manta ray within the *Action Area*.

Between 2008 and 2015, there were approximately 68,000 to 142,000 annual sets by the international purse seine fleet operating in the WCPO exclusive of those by the United States fleet (data from the Western and Central Pacific Fisheries Commission website). The WCPO purse seine fishery as a whole, exclusive of the United States fleet, was observed at rates between 44-69% from 2013–2017. In contrast, the United States fleet has received 100% coverage since 2010, although not all data are available (see discussion in the *Effects Analysis section*). We note that much of the best scientific and commercial data available in the literature based on WCPO data is inclusive of United States data and we generally cannot separate the United States portion from the greater WCPO data.

The data has shown approximately 1.6% of sharks caught in the purse seine fisheries from 2010 to 2016 are oceanic whitetips (Figure 15) which are most likely a result of decreased number of FADs and are indicative of population declines in the area (Peatman et al. 2017). Since 2009, bycatch levels have been relatively low and stable (Peatman et al. 2017; Figure 15). Updates provided by Peatman and Nichols (2021) show slightly increased catch estimates from 2018-2020 (see Table 8 in Peatman and Nichols 2021).

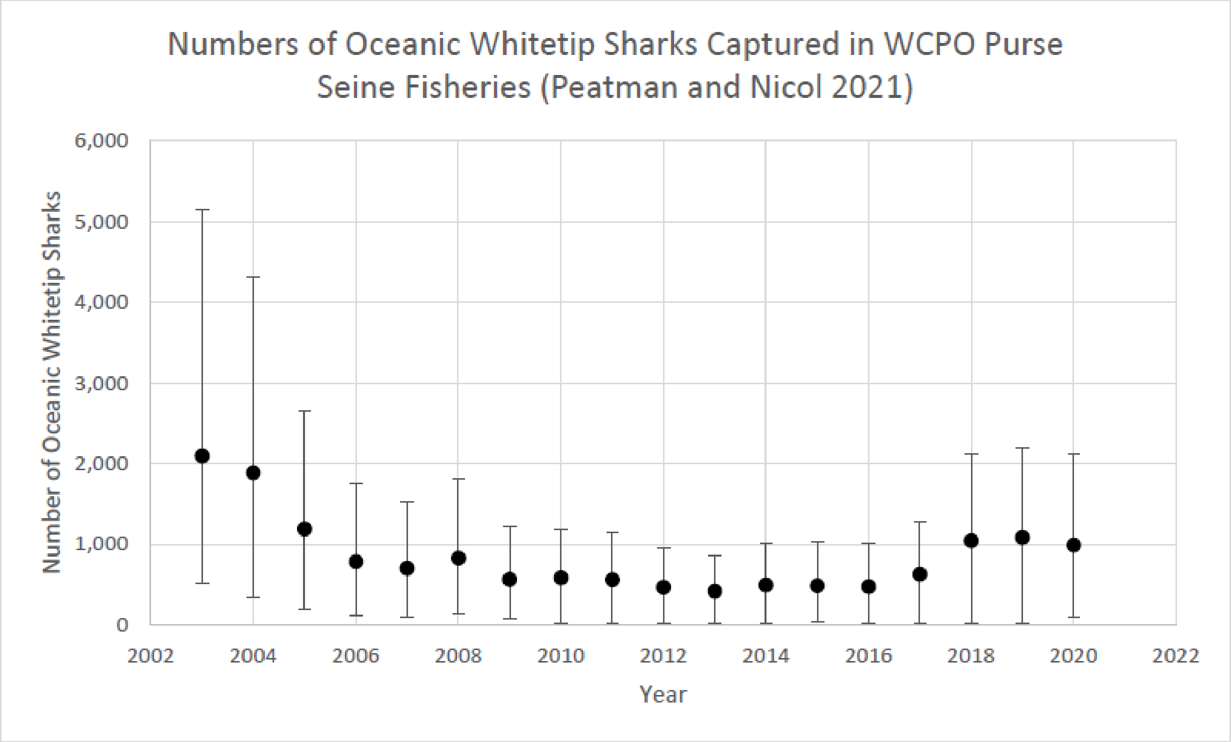


Figure 15. Predicted total annual oceanic whitetip bycatch (numbers) by year for large-scale purse seine fleets between 2002 and 2020. Ninety-five percent confidence intervals (95% CI) shown in boxplot whiskers. Source: Peatman and Nichol 2021.

Peatman et al. (2017) provided modeled estimations of oceanic whitetip shark catches due to the lack of record submissions to the WCPFC by several nations in the smaller purse seine vessel fleet, and due to lower than mandated observer coverage rates. Actual observed numbers of individuals caught for this period (*n* = 1,822), for the large-scale purse seine fleet, are provided in (Peatman et al. 2017). Peatman and Nichol (2021) later estimated that the purse seine fleet managed by the WCPFC captured about 15,267 (median estimate) oceanic whitetip sharks from 2003 to 2020. 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 and purse seine fisheries and oceanic whitetip sharks, which includes the *Action Area*. Again, as these numbers represent the entire WCPFC boundaries, we cannot parse out the number of bycaught, harmed or killed oceanic whitetip individuals in the *Action Area* by foreign fisheries. However, at this time, this is considered the best scientific data available for this fishery and Region.

Entanglement in FADs has been documented for silky sharks and for *Carcharhinus* species which may include oceanic whitetip sharks (Chanrachkij and Loog-on 2003; Filmalter et al. 2013; Murua et al. 2017). Additionally, no estimates in the literature for oceanic whitetip shark interaction rates with FADs exist, therefore, while we acknowledge that FAD entanglement is a likely stressor for oceanic whitetip sharks, we do not know the significance of this potential stressor on the species chances of survival and recovery. We do recognize that ten records of FAD entanglements with oceanic whitetip sharks exist in the WCPO purse seine fishery with one FAD accounting for 5 individuals in 2018 (NMFS unpublished data). However, more information is needed regarding these types of events.

Sharks can become entangled in the net wall or in dFAD materials. An entanglement as such, could cause the shark to die if it is unable to circulate water through its gills. Hutchinson et al. (2015) also described that some sharks which are entangled are removed from the net as it's removed from the water, thereby these sharks were landed before sacking up or brailing occurs and have better chances at survival.

Pilling et al. (2018) suggests that oceanic whitetip sharks were the second most commonly entangled shark in the WCPO in 2015, although the number of individuals was not provided. Without constant monitoring of a dFAD, large numbers of incidentally caught animals can be unobserved as they may decompose or be predated on, removing any trace of an interaction and would therefore be considered cryptic or unaccounted mortalities (Filmalter et al. 2013; Gilman et al. 2013; Pilling et al. 2018).

One set with one oceanic whitetip was documented as entangled in a FAD. However the data does not elaborate on the event. The shark was assigned A2- alive but injured by the observers. Animals entangled in FADs can naturally degrade or be depredated on before the retrieval of the device (Filmalter et al. 2013). Simple online searches provide photographic evidence of sharks entangled in FADs and Filmalter et al. (2013) has identified this issue as a major source of mortality, at least in the Indian Ocean. However, we do not know the significance of this potential stressor on the species chances of survival and recovery without additional data.

In the western Pacific fisheries, *Manta* spp. are rarely reported in the bycatch. In the tropical tuna purse seine fisheries, Hall and Roman (2013) note that *M. japonica* represents the most abundant mobulid in the fishery bycatch. Analysis of the catch of WCPFC purse seine and longline fisheries from 1995-2015 (based on observer data) showed that giant manta rays are rarely caught (Tremblay- Boyer and Brouwer 2016). In purse seine sets, the species is observed at a rate of 0.0017 individuals per associated set (sets made around a FAD) and 0.0076 individuals per unassociated set (sets on free swimming schools of tuna) (Tremblay- Boyer and Brouwer 2016). The available standardized purse seine CPUE data from the western and central Pacific Ocean show strong reporting bias trends (as observer reporting in the purse seine fisheries to species-level became more prevalent after 2008), and, therefore, are not particularly useful for accurately assessing abundance trends (Tremblay-Boyer and Brouwer 2016).

3.4 Vessel Strikes

Marine habitats of the action area often feature both heavy commercial and recreational vessel traffic. However, vessel strikes are not expected to be a threat to oceanic whitetip sharks given their depth preferences and lack of recorded strikes in the available literature. Vessel strikes represent a recognized threat to large, air breathing marine species, and is also a potential threat to the giant manta ray. This threat is increasing as commercial shipping lanes cross important breeding and feeding habitats (Swingle et al. 1993; Wiley et al. 1995). As vessels continue to become faster and more widespread, an increase in vessel interactions is expected.

Manta ray aggregation sites are sometimes in areas of high maritime traffic, and therefore are at potential risk of being struck and killed by boats (Marshall et al. 2011a; Graham et al. 2012). Internet searches also reveal photographs of mantas with injuries that are consistent with boat strikes, and manta researchers report that they may affect manta ray fitness in a significant way (The Hawaii Association for Marine Education and Research Inc. 2005; Deakos et al. 2011; Heinrichs et al. 2011; Couturier et al. 2012; CMS 2014; Germanov and Marshall 2014; Braun et al. 2015), potentially similar to the impacts of shark or orca attacks. However, there is very little quantitative information on the frequency of these occurrences.

Our review of the literature reveals that there is no information which indicates that vessel strikes are an issue for the oceanic whitetip shark or giant manta rays in the *Action Area*.

3.5 Pollution and Marine Debris

Many different types of pollution can adversely affect threatened and endangered 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 threatened and endangered resources.

According to the U.S. Energy Information Administration (2022), American Samoa imports all of its fossil fuels for the territory's various energy needs including approximately 97% of its electric power generation, for the treatment of drinking and waste water, and for transportation. Total petroleum imports through Pago Pago Harbor are approximately 2,300 barrels a day, since 2011 (U.S. Energy Information Administration 2022).

Past oil spills that impacted the offshore marine environment or within the *Action Area* are listed in Table 12 as reported by the USCG (2013) and supplemented as noted. Where available, information on the impacts to natural resources is summarized.

Table 12. Important historical and recent spills in the *Action Area*. From the American Samoa Area Contingency Plan Section 9000-9 (March 2013) except where otherwise noted.

Date	Spill Name/Location	Oil Type and Volume	Natural Resource Impacts
10/7/1949	USS CHEHALIS (AOG-48) Fuel Dock, Pago Pago Harbor	100,000+ gallons	None noted
12/10/1991	Typhoon Val Pago Pago Harbor	36,000 gallons of oil and 600 lbs. of ammonia (NOAA 2022a)	Minimal wildlife impacts- reef associated; No ESA-listed species (Sifling et al. 2001)

Date	Spill Name/Location	Oil Type and Volume	Natural Resource Impacts
10/14/1993	F/V JIN SHIANG FA Rose Atoll	100,000 gallons diesel	Severe localized coral and clam kills down to depths of 15 ft. No ESA-listed species documented.
8/17/2010	*M/V Syota Maru* Pago Pago Harbor	Discharging oily bilge waste (DOJ 2014).	None noted.
2/7/2018	*F/V Chu Zai Fa No. 1* Leone Bay, American Samoa	No oil released (NOAA 2022b)	None noted

According to the American Samoa Area Contingency Plan, the daily working average for normal response events is between 25 to 100 gallons for any given release and do not require Honolulu Sector augmentation or involvement (USCG 2013). However, the cumulative number of minor spills is not noted.

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

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

PAHs are environmental contaminants that are naturally occurring and come from multiple anthropogenic sources, including oil spills. Some PAHs are carcinogenic, mutagenic (cause developmental abnormalities), genotoxic, immunotoxic, cause adverse respiratory effects, and larval mortality (U.S. Department of Health and Human Services 1995; Mastrangelo et al. 1996; Incardona et al. 2004; Bechmann et al. 2010; Balciog˘lu 2016; Zychowski and Godard-Codding 2017). PAHs can bioaccumulate in prey items leading to toxic effects in those species (Moore et al. 1989). Some PAHs provide serious adverse effects in the marine environment over 17 months after an oil spill event (Perez et al. 2008) and the EPA designated 16 PAHs as High Priority Pollutants because of their potential toxicity, prevalence, and persistence within the environment (40 CFR Part 423, Appendix A; Hussar et al. 2012). Therefore, PAHs are also considered Persistent Organic Pollutants, or POPs. PAHs appear to show greater partitioning and storage in tissues than other categories of POPs with some species at greater risk of long-term bioavailability although varying risks given life stage and sex (Munoz et al. 2021). Additionally, POPs are maternally transferrable in some species, like sea turtles, and affect neonates (Ehsanpour et al. 2014; De Andres et al. 2016; Munoz and Vermenien 2020 and references cited therein). Barraza et al. (2021) showed that location specific variables can influence POP loading in an individual and may vary between populations given high site fidelity, even amongst close geographic populations. However, although poorly researched, the literature also suggests that species higher in the trophic chain have some ability to metabolize *some* PAHs. Long-term fitness impacts may occur from chronic exposure that can ultimately cause more harm to populations (Camacho et al. 2012). Hutchinson and Simmonds (1992) suggested fibropapillomatosis may also be influenced by PAH exposure in sea turtles while long-term effects from chronic PAH exposure can result in cancers (Collins et al. 1998; Camacho et al. 2012). However, data linking PAHs and POPs to cancers in elasmobranchs is lacking.

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

In a productivity-susceptibility analysis and associated maximum entropy (MaxEnt) modeling approach, Romo-Curiel et al. (2022) assessed oil spill exposure to 24 large pelagic fish species in the Gulf of Mexico, given their productivity and susceptibility to exposure to determine potential for recovery. Both the oceanic whitetip and giant manta were species considered in their analysis. The oceanic whitetip shark had a medium vulnerability ranking and the giant manta ray had a high vulnerability ranking (Table 2 and Figure 2 in Romo-Curiel et al. 2022). The authors then used these rankings in conjunction with a hotspot analysis to determine an overall ranking given three geographic scenarios. Ultimately, the oceanic whitetip shark had a low ranking whereas the giant manta ray ranked medium across all scenarios in deep oceanic waters. We recognize, we do not have rankings for portion of the species' population(s) which occur in the Central or greater Pacific. However, we can consider these rankings applicable in our Region since it is focused on the biological factors of the species, and because the areas considered (i.e., the GOM versus the *Action Area*) are both deep pelagic waters, and the two species preferred depth ranges across ocean basins is the same (Miller and Klimovich 2017; Young et al. 2017).

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

Marine debris has become a widespread threat for a wide range of marine species that are increasingly exposed to it on a global scale. Plastic is the most abundant material type worldwide, accounting for more than 80% of all marine debris (Poeta et al. 2017). The most common impacts of marine debris are associated with ingestion or entanglement and both types of interactions can cause the injury or death of animals of many different species. Ingestion occurs when debris items are intentionally or accidentally eaten (e.g., through predation on already contaminated organisms or by filter feeding activity, in the case of large filter feeding marine organisms, such as whales) and enter in the digestive tract. Ingested debris can damage digestive systems and plastic ingestion can also facilitate the transfer of lipophilic chemicals (especially persistent organic pollutants) into the animal's bodies. An estimated 640,000 tons of fishing gear is lost, abandoned, or discarded at sea each year throughout the world's oceans (Macfadyen et al. 2009). These "ghost nets" drift in the ocean and can fish unattended for decades (ghost fishing), killing large numbers of marine animals through entanglement. Fisheries observers in the Hawaii longline fisheries (SSLL and DSLL) document marine debris that is encountered during fishing trips. From 2008-2015 they documented 1,426 debris encounters with marine debris. Over 75% of the debris encountered consisted of nets and tangled debris. Almost 85% of debris was collected and brought back to the docks; this amounted to over 211,838 pounds of debris (NOAA 2015).

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 et al. 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 POPs 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 toxins and POPs associated to microplastic ingestion (Germanov et al. 2018). While the ingestion of plastics is likely to negatively impact the health of the species, the levels of microplastics in manta ray feeding grounds, frequency of ingestion and the transfer of toxins are presently being studied to evaluate the impact on these species (Germanov 2015a, 2015b).

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

It is difficult to assess the precise impact pollution and marine debris have had on 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.

3.6 Anthropogenic Sound

Oceanic whitetip sharks and giant manta rays that occur in the *Action Area* are regularly exposed to multiple sources of anthropogenic sounds. Anthropogenic sounds that could affect ambient noise arise from the following general types of activities in and near the sea, any combination of which can contribute to the total sound at any one place and time. These sounds include commercial and recreational ships, aircraft, sonar, dredging, construction; oil gas, and mineral exploration and extraction in offshore areas; geophysical (seismic) surveys; sonars; explosions; and ocean research activities, all of which may generate anthropogenic sound in the *Action Area* to varying degrees (Richardson et al. 1995; NRC 2003). Oceanic whitetip sharks and giant manta rays may be adversely affected by anthropogenic sound in various ways. It can produce direct physical harm or may illicit behavioral responses including, but not limited to, cessation of feeding, resting, or social interactions, changes in habitat to avoid areas of higher sound levels, or changes in diving behavior (MMC 2007).

Oceanic whitetip sharks and giant manta rays have the potential to be impacted by increased levels of both background sound and high intensity, short-term sounds. Sources of anthropogenic sounds are becoming both more pervasive and more powerful, increasing both oceanic background sound levels and peak intensity levels (Hildebrand 2004). Exposure to vessel noise (including recreational boats, whale-watching boats, research vessels, and ships associated with oil and gas activities) for elasmobranchs, may result in short-term behavioral or physiological responses (e.g., avoidance, stress) but is not quantifiable in the *Action Area* at this time.

Elasmobranchs, like all fish, have an inner ear capable of detecting sound and a lateral line capable of detecting water motion caused by sound (Hastings and Popper 2005; Popper and Schilt 2008). Data for elasmobranch fishes suggest they are capable of detecting sounds from approximately 20 Hz to 1 kHz with the highest sensitivity to sounds at lower ranges (Myrberg

2001; Casper et al. 2003, 2012; Casper and Mann 2006, 2009; Ladich and Fay 2013). However, unlike most teleost fish, elasmobranchs do not have swim bladders (or any other air-filled cavity), and thus are unable to detect sound pressure and presumably only detect particle motion (Casper et al. 2012). As such, elasmobranchs are not considered particularly sensitive to sound (Casper et al. 2012). There have been no studies examining the direct effects of exposure to specific anthropogenic sound sources in any species of elasmobranchs (Casper et al. 2012).

The most likely response of ESA-listed elasmobranch exposed to sound stressors, if any, will be minor temporary changes in their behavior including increased swimming rate, avoidance of the sound source, or changes in orientation to the sound source, none of which rise to the level of harassment or harm. If these behavioral reactions were to occur, we would not expect them to result in fitness impacts such as reduced foraging or reproduction. We do not have information to describe the significance of anthropogenic sound on elasmobranchs in the *Action Area*.

3.7 Military Training and Testing Activities

As the *Action Area* contains portions of international waters, military activities may occur in the *Action Area* that are unknown to us. However, at this time, we do not have the ability to describe the risk posed to the species under consideration though we attempt to address United States resources in the Region.

The United States Navy (including the United States Marine Corps aboard naval vessels) may conduct training and testing activities, military readiness activities, or could potentially face active combat operations in the *Action Area*. The United States Pacific Fleet is the world's largest naval fleet with an estimated Area of Responsibility of 100 million square miles including the entire *Action Area*. The command oversees approximately 200 vessels, 1,200 aircraft, and 13,000 personnel (United States Department of the Navy 2019). Additionally, the United States Marines Indo-Pacific Command's Area of Responsibility includes the entire *Action Area*. To date, no Biological Opinions have been completed for operations within the *Action Area* as all United States Pacific Fleet and Marine Indo-Pacific Command training exercises are conducted in the Hawaii-Southern California Training and Testing Study Area or the Mariana Islands Training and Testing Area, which do not overlap with the *Action Area* and are covered by separate ESA Section 7 consultations (NMFS 2017c, 2018c).

During training, existing and established weapon systems and tactics may be used in realistic situations to simulate and prepare for combat. Activities include routine gunnery, missile, surface fire support, amphibious assault and landing, bombing, sinking, torpedo, tracking, and mine exercises. Testing activities are conducted for different purposes and include at-sea research, development, evaluation, and experimentation. Any training or readiness operations that may occur in the *Action Area* of this proposed action would be expected to complete separate ESA Section 7 consultation before operations commence. However, routine operations such as patrolling, search and rescue, maritime inspections, flight operations, refueling, humanitarian aid, disaster relief, etc. may occur on a regular basis.

Sector Honolulu of the United States Coast Guard provides services to a broad expanse of the Pacific including the *Action Area*. United States Coast Guard operations are ongoing and are expected to continue into the future and are similar in nature and scope to those described above for the United States Navy. However, Sector Honolulu's live fire drills, otherwise known as

GUNEX operations, are conducted in Hawaiian or Northern Mariana's waters, outside of the *Action Area*. Effects to ESA-listed species from Coast Guard munitions would likely only occur during active combat, drug interdictions, or other situations related to national security where ammunition would be fired. The United States Coast Guard is expected to complete Section 7 consultation for any training exercises that would occur in the *Action Area*, however they may conduct the same routine operations as the United States Navy and Marine Corps as noted above.

Responses of listed species struck by projectiles or from explosives may range from induced behavioral responses and avoidance measures, to direct impacts from munitions. An animal subjected to a direct hit or very near miss from a fired projectile would result in wounding or possibly death. Responses to an explosion would vary depending on proximity to the blast (Duronslet et al. 1986; Govoni et al. 2008; Viada et al. 2008; United States Navy 2017; Fetherston et al. 2019). Potential injuries from projectiles and explosions may include cuts, gashes, bruises, gunshot wounds, broken bones, hearing loss, rupture or hemorrhage of internal organs, amputations, or other broken body parts; any of which could result in an animal's death. Wounded animals may be less able to maneuver or travel, thereby increasing the animal's likelihood of being killed by predatory species (e.g., sharks). Impacts from harassment due to these types of activities include changes from foraging, resting, milling, and other behavioral states that require lower energy expenditures to traveling, avoidance, and behavioral states that require higher energy expenditures to flee an area (United States Navy 2017).

The United States Air Force and other military entities may conduct flight operations within the *Action Area*, however affects to ESA-listed species are not expected as aircraft would be at altitude where auditory affects to underwater animals would be minimal. Sonic booms would most likely provide the largest auditory stress, but would be highly unlikely to illicit effects from ESA-listed species as determined by prior consultations conducted by NMFS (ex: see NMFS 2019d, 2019e).

3.8 Synthesis of Baseline Impacts

The listed resources considered in this biological opinion have been and continue to be exposed to a wide variety of the past and present state, federal, and private actions in the *Action Area*, which includes 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. The baseline conditions affecting species include global climate change, fisheries and fisheries bycatch, vessel strikes, pollution from chemicals and marine debris, ocean noise from variety of sources, and military training and testing.

Multiple factors in the baseline, such as incidental bycatch in fisheries, are identified as threats affecting the status as threatened, and/or ability of species to recover.

- Oceanic Whitetip Shark- Overutilization is a listing factor for oceanic whitetip sharks because of its tendency to remain in the surface mixed layer of the water column (0-152 m depth) and in tropical latitudes where fishing pressure is often most concentrated for target species such as tuna, the species is frequently encountered and suffers high mortality rates in numerous fisheries throughout its global range. The oceanic whitetip is regularly caught incidentally with pelagic longlines, purse seines, handlines, troll and occasionally pelagic and even bottom trawls (Compagno 1984). In addition to mortality as a result of retention and finning in commercial fisheries, oceanic whitetip sharks experience varying levels of bycatch-related fishing mortality, including at-vessel and post-release mortality (81 FR 96304; 3/29/17).

- Giant Manta Ray - The most significant threat to the giant manta ray is overutilization for commercial purposes. Giant manta rays are both targeted and caught as bycatch in a number of global fisheries throughout their range, and are most susceptible to industrial purse-seine and artisanal gillnet fisheries. Efforts to address overutilization of the species through current regulatory measures are inadequate, as targeted fishing of the species still occurs despite prohibitions in a significant portion of the species' range. Also, measures to address bycatch of the species in industrial fisheries are rare (NMFS 2020) https://www.fisheries.noaa.gov/species/giant-manta-ray.

The stress regime created by the activities discussed in this *Environmental Baseline* continues to have a serious and adverse impact on oceanic whitetip sharks and giant manta rays, which we present below.

Giant manta rays face a high probability of extirpation as a result of environmental and demographic stochasticity. Due to their particular life-history characteristics (e.g., slow growth, late maturity, and low fecundity), giant manta rays have little potential to withstand high and sustained levels of fishing exploitation. The information available suggests that giant manta rays have a high probability of becoming extirpated in the Pacific Ocean unless they are protected from the combined threats of incidental capture in the industrial purse-seine fishery and intentional harvests in the artisanal gillnet fisheries that supply the international mobulid gill raker market. The number of individuals that continue to be incidentally captured and killed in fisheries in the *Action Area* contributes to the increased extinction risk of the species.

Of the other activities and their associated stressors, the propensity of vessel strikes to go unnoticed or unreported by vessel operators impedes an accurate assessment of the magnitude this threat poses to giant manta ray. However, giant manta ray occur in the pelagic waters within the *Action Area* where their density is sparse in comparison to nearshore aggregation sites where as a result of a higher density of rays, there is an increased risk of a vessel strike. Therefore, we do not expect vessel strikes to contribute to the increased extinction risk of the species.

Of the different types of pollution that can adversely affect giant manta rays, the ingestion and entanglement risk of marine debris may pose the most serious threat. Because giant manta rays must filter hundreds to thousands of cubic meters of water daily to obtain adequate nutrition (Paig-Tran et al. 2013), they can ingest microplastics directly from the water or indirectly through their contaminated planktonic prey (Setala et al. 2014). Microplastics can 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). If entangled in marine debris, 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. The number of individuals that continue to ingest and become entangled in marine debris in the *Action Area* contributes to the increased extinction risk of the species.

Giant manta ray may respond to noises by avoiding, halting their activities, experience reduced hearing by masking, or become attracted to source noises. Avoidance is most likely, and a common natural reaction and considered low risk. Nevertheless, giant manta rays are large agile animals and capable of swimming away safely from disturbances that would harm them, therefore, ocean noise in the *Action Area* is not expected to contribute to the increased extinction risk of the species

The oceanic whitetip shark face a high probability of becoming endangered as a result of both environmental and demographic stochasticity. Due to their life-history characteristics, oceanic whitetip sharks are more susceptible to the effects of high fishing exploitation. The information available suggests that oceanic whitetip sharks have high a probability of becoming extirpated in the Pacific Ocean unless they are protected from the combined threats of incidental capture and commercial utilization from worldwide fisheries. The number of individuals that continue to be incidentally captured and killed in fisheries in the *Action Area* contributes to the increased extinction risk of these species.

Of the other activities and their associated stressors, the propensity of vessel strikes to go unnoticed or unreported by vessel operators impedes an accurate assessment of the magnitude this threat poses to the oceanic whitetip shark. However, these sharks are large agile animals and capable of moving quickly if approached by a vessel. Therefore, we do not expect vessel strikes to contribute to the increased extinction risk of the species.

Of the different types of pollution that can adversely affect these shark species, the ingestion and entanglement risk of marine debris may pose the most serious threat. Ingestion of marine debris may lead to injury or possibly starvation, and entanglement in debris could directly or indirectly interfere with the shark's mobility, causing impairment in feeding, breeding, or migration. However, due to the foraging behavior of sharks and the limited reports on entanglement, marine debris in the *Action Area* is not expected to contribute to the increased extinction risk of these species.

Sharks may respond to noises by avoiding, halting their activities, experience reduced hearing by masking, or become attracted to source noises. Avoidance is most likely, and a common natural reaction and considered low risk. Nevertheless, sharks are large agile animals and capable of swimming away safely from disturbances that would harm them, therefore, ocean noise in the *Action Area* is not expected to contribute to the increased extinction risk of these 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.

In this section of our assessment, our risk analyses consider the consequences of the exposures and responses for the individuals and populations that are likely to be exposed to the fishery. We present the effects of the fishery on the threatened species, which must form the basis for our jeopardy determinations, in the *Integration and Synthesis* section, which estimates the risks the fishery poses to the threatened species by pulling together the information we presented in the *Status of Listed Resources*, *Environmental Baseline,* and *Effects of the Actions* sections of this opinion. That synthesis follows the *Cumulative Effects* section of this opinion, which assesses the probable effects of future State, local, and private actions that are reasonably certain to occur in the *Action Area*.

4.1 The American Samoa Longline Fishery

Since 2006, the Pacific Islands Region Observer Program has deployed observers on a sample of ASLL trips. However, initial observer coverage was low, less than 10%. In 2010 and 2011, annual observer coverage was increased to approximately 24 and 31% respectively. In September 2011, NMFS implemented Amendment 5 to the Pelagics FEP (76 FR 52888), which established specific requirements for fishing gear and deployment techniques to reduce incidental bycatch of green sea turtles. The objective of these requirements was to set gear at 100 m or deeper in the water column. Although the intent of Amendment 5 was to reduce the bycatch of green sea turtles, it could also affect the bycatch rates of giant manta ray and oceanic whitetip shark. Therefore, to account for the uncertainty of impact of Amendment 5, we rely on the anticipated number of interactions from 2012-2017, excluding data from 2010-2012, unless otherwise noted (McCracken 2019a).

4.2 Potential Stressors

Potential stressors associated with the proposed action include:

1. Interaction with including incidental 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; and
6. vessel emissions.

We determined that vessel noise, vessel collisions, introduction of discharges and other wastes, derelict gear, and vessel emissions are stressors that are not likely to adversely affect giant manta rays and oceanic whitetip sharks. Those stressors are covered later in this biological opinion (See *Appendix A*). As a result, in this section we focus primarily on the stressors created by active fishing, which results in hooking and entanglement of listed species and are thus likely to adversely affect them. We briefly discuss slipped or unobserved catch, and depredation of bait and catch.

4.3 Summarized Responses of ESA-listed Species to the Fishery

The most significant hazard the ASLL fishery presents to listed species results from hooking and entanglement by gear, which can injure or kill oceanic whitetip sharks and giant manta rays. If hooked or entangled, they may not immediately die from their wounds and can suffer impaired swimming or foraging abilities, altered migratory behavior, altered breeding or reproductive patterns, and latent mortality from their interactions.

Although survivability studies have been conducted on some listed species incidentally captured in longline fisheries, long-term effects are nearly impossible to monitor; therefore, a quantitative measure of the effect of oceanic whitetip shark and giant manta ray populations is very difficult. Even if oceanic whitetip sharks and giant manta rays 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 and giant manta rays 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; 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.

4.3.1 Entanglement in Longline Gear

Oceanic Whitetip 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 branchlines and mainline. An entanglement as such, could cause the shark to perish if it is unable to circulate water through its gills. The literature on sharks captured on longline gear is primarily focused on the effects of hooking, post release handling, and post hooking mortality, not entanglement in longline gear.

However, marine debris data compiled in NOAA's 2014 Marine Debris Program Report reveals several accounts of sharks entangled in natural fiber rope and monofilament (NOAA Marine Debris Program 2014). A shortfin mako shark entangled in natural fiber rope, resulted in scoliosis, abrasions and was undernourished (Wegner and Cartamil 2012) and the monofilament found encircling a blacknose shark caused its spine to be deformed (Schwartz 1984). In general, entanglement could directly or indirectly interfere with the shark's mobility, causing impairment in feeding, breeding, or migration.

Giant Manta Ray

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 branchline 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, some data is available regarding the reef manta. Reef manta are reasonable surrogate species as, prior to 2009, the two manta species were categorized as one. Surveys of the reef manta from 2005-2009 at an aggregation site off Maui, Hawaii, revealed that 10% of the population had an amputated or non-functional cephalic fin (Deakos et al. 2011). Most of these injuries were attributed to entanglement in fishing line (most likely from recreational or nearshore fisheries) since the straight edge cut of all amputated cephalic fins resemble the severing effects of monofilament (Deakos et al. 2011). In fact, eight individuals had physical evidence of entanglement with fishing line; two individuals had hooks in the cephalic fin, two had monofilament wrapped around the cephalic fin and another two had scars from previous line entanglements, and two individuals had line that had begun to cut part way through the cephalic fin (Deakos et al. 2011). Deakos et al. (2011) observed that individuals in this population with an amputated cephalic fin appeared healthy, however, considering the function of the cephalic fin to guide food into the manta's mouth, feeding efficiency is most likely reduced, and the absence of this fin may negatively affect size, growth rate and reproductive success. Lastly, Deakos et al. (2011) report that videos show two reef manta rays in Hawaii, which were entangled in mooring lines, perish and become immediately consumed by sharks. Although mooring lines are not used in this fishery, the material is similar to polypropylene float line, entangles these animals, and shows that predators are quick to take advantage of an entangled animal.

4.3.2 Hooking

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, incidental capture on a longline is a stressful experience that can last an average 10-12 hours in the ASLL fishery, and may last longer. During capture, the amount of water flow over the gills is limited and biochemical recovery can take up to 2 to 7 days, and even longer for injured sharks (Campana et al. 2009). In addition, sharks are vulnerable to predation while being captured due to their restricted mobility, and after their release due to exhaustion and injury. Furthermore, handling procedures can cause additional damage (e.g., cutting the jaw, tail, gaffing, etc.), stress, or death.

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

Giant Manta Ray

The giant manta ray primarily feeds on planktonic organisms such as euphausiids, copepods, mysids, decapod larvae and shrimp, but some studies have noted their consumption of small and moderate sized fishes (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 branchline. 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.

4.3.3 Trailing Gear (Line)

Oceanic Whitetip Sharks

Excessive trailing gear could directly or indirectly interfere with the 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 WCPFC 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.

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 branchline 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 (Scott et al. 2022). 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.

4.4 Species-Specific Analyses

The narratives that follow present our exposure and response analyses for the oceanic whitetip sharks and giant manta rays 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 ASLL fishery in the future based on the number and rate of individuals incidentally captured in the fishery since 2010 and 2012. Unlike the SSLL fishery, which has 100% observer coverage, the ASLL 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.

For our response analyses, we try to assess how animals respond after being incidentally captured in ASLL gear. For the most part these analyses are informed by published observer reports of 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 as a result of being incidentally 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 deep-set longline fishing gear affects the 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 narratives that follow this introduction present and discuss intervals: the exposure estimates present mean values with 95th percentiles (all from McCracken 2019a) while response and risk estimates present mean values with lower and upper 95% confidence intervals. It is important to note that all values in a confidence interval are reasonably compatible with the data although the point estimate (in our case, the mean value) is the value most compatible with the data.

For example, Table 14 presents annual estimates of the number of oceanic whitetip sharks that are likely to be incidentally captured in the ASLL fishery (annual mean = 620; 95th percentile = 1,100). The estimates in Table 14 should be interpreted as "the data suggest that the number of oceanic whitetip sharks incidentally captured in any given year should average about 620 and rarely will be higher than 1,100; over any five consecutive year period, we expect the average to be no more than 696.4 oceanic whitetip sharks and no more than 3,482 total incidentally captured over five years

The upper limits of 95% confidence intervals are important because they are compatible with the data (although less so than the mean) and represent greater risks for endangered and threatened species. Although these upper limits will always be higher than the average values, they may still be exceeded in some years. Our assessments do not ignore the lower limits of 95% confidence intervals, but do not focus on them because they represent lesser risk to endangered and threatened species. For risk analyses over longer time intervals (40 years used in this assessment), we rely on mean estimates because long time series are most likely to include values both above and below the mean.

4.4.1 Oceanic Whitetip Shark

Exposure Analysis

Between 2006 and 2019, 1,412 confirmed oceanic whitetip sharks were observed caught in the ASLL fishery (Table 13; NMFS Observer Program unpublished data). Between 2006 and 2009, observer coverage was below 10% and not considered representative of the fleet's effort. Therefore, incidental captures that occurred over this time frame were not used to estimate future anticipated numbers of incidental captures and instead we focus on the data from 2010 to 2019 when observer coverage average was about 22%, providing a more reliable dataset to evaluate the fishery's impacts. From 2010 to 2019, 1,211 oceanic whitetip sharks were observed captured by the ASLL fishery. McCracken (2019, 2020) used a Bayesian inference model to estimate total captures based on observed captures and fishery effort. She estimated that there were a total of 6,318 oceanic whitetip sharks incidentally captured by the ASLL fishery from 2010 to 2019. NMFS' observers are instructed to document interactions with sharks by species when possible, however, when a shark breaks the line or is able to come free of the hook before being pulled alongside the vessel, the observer may not be able to distinguish whether the shark is an oceanic whitetip shark or another species. In this situation, observers are instructed to use the unidentified shark classification. To estimate how many observed unidentified sharks are oceanic whitetip sharks, we used a ratio of unidentified sharks to the combined total of confirmed oceanic whitetip sharks and 18 other shark species commonly incidentally captured in the ASLL fishery to estimate the potential of oceanic whitetip sharks that may have been present in the unknown shark category. In addition, we used the Wilson Score Method to calculate confidence intervals, which uses asymptotic variance without continuity correction (Newcombe 1998).

The resulting proportion of observed unidentified sharks incidentally captured between 2010 and 2019 that were likely oceanic whitetip sharks was 9% (95% CI: 8.5% to 9.5%). We use the upper 95th percentile value of 9.5% of unidentified sharks are likely to be oceanic whitetip sharks to estimate the annual number of unidentified sharks that were likely oceanic whitetip sharks (Table 13).

Table 13. Oceanic whitetip shark interaction data displaying condition at end of interaction to determine percentage released alive in the ASLL fishery. Observer coverage from 2006-2009 (light gray shaded area) was below 10%, and not considered representative of the fleet's effort, and therefore not used by McCracken (2019, 2020) to estimate the mean and 95th percentile of anticipated captures.

Year	Number of Observed Interactions	Estimated (McCracken 2019a, 2020)	Released dead	Released Alive	Estimated Unidentified Sharks	UnID Sharks Likely Oceanic Whitetip	Total Estimated Oceanic Whitetip Sharks
2006	51	-	11	40			
2007	59	-	18	40			
2008	46	-	16	30			
2009	45	-	13	32			
2010	146	1,176	42	104	76	7	1184
2011	101	319	39	61	60	6	325
2012	72	470	25	47	51	5	475
2013	97	407	18	79	26	2	410
2014	103	464	38	65	21	2	466
2015	186	827	62	124	50	5	832
2016	169	899	63	106	56	5	794
2017	65	458	24	41	35	3	488
2018	129	513	46	83	133	13	526
2019	143	870	23*	52*	96	9	880
Totals	1,211***	6,243	380**	762**	604	57	6,380

*Only includes data up to the 2nd quarter. Also excludes two sharks that were retained, one each in 2007 and 2011.

**Total released dead/alive from 2010 to the 2nd quarter of 2019.

***This total does not include data from the years 2006 to 2009 (light grey) when the data were not representative of the entire fleet. Total observed interactions including those years is 1,412.

Demographic Patterns of Exposure

While we have data for total annual incidental captures from 2006 to 2019, only data up to the 2nd quarter of 2019 have been assessed for release condition, length and sex, therefore we focus on this segment of the data for these factors. In total, 1,344 oceanic whitetip sharks were observed incidentally captured from 2006 to the 2nd quarter of 2019.

Sharks are not always boarded for a number of reasons in the ASLL fishery, including poor weather, the shark's size, condition at landing, efficiency of maintaining fishery operations, safer and less stressful for the animal, and for the crew's safety. Therefore, observers estimate lengths based on their best professional judgement and with known measurements around the rail of the vessel. Of the 1,344 observed interactions of oceanic whitetip sharks between 2006 and 2019 (2nd quarter), length estimates were provided for 412 individuals. The average estimated length was 122 cm (4.0 ft., range 1 ft. to 8 ft.).

When feasible, observers collect actual length measurements. Currently the protocol is to measure every third fish, regardless of species (NMFS 2017a). The actual number of oceanic whitetip sharks boarded and measured during this time frame was 82 individuals and included 22 measurements that were out of protocol (not the 3rd fish). These measurements were opportunistically collected by the observers. The average lengths were as follows; pre-caudal 84.12 cm (n = 60); fork length of 96.06 cm (n = 60); outside-of-protocol pre-caudal lengths 98.05 cm (n = 22); and out-of-protocol fork length of 109.72 cm (n = 22). Average of all pre-caudal lengths resulted in 87.85 cm (n = 82). The total average overall fork length is 99.73 cm (n = 82). The largest recorded specimen had a fork length of 165 cm and a pre-caudal length of 149 cm. When comparing this data to Joung et al. (2016) the data show the average shark that was measured in this fishery is approximately 2 years old with a maximum age of 7 years. However, these data should be regarded with some caution. Length data only exist for 6.1% (82/1344*100) of the observed specimens, the growth curves measure total length and the measurements collected in the ASLL fishery are fork length measurements, and since all sharks are not landed and measured primarily due to safety reasons, the sharks that are measured are either dead or smaller individuals. In fact, 22% (18/82*100) of the measured sharks were alive and average size of these sharks is 78 cm.

In the North Pacific, females mature at about 168-196 cm TL, and males at 175-189 cm TL, which corresponds to an age range of 4 years (Seki et al. 1998) to 8.5-8.8 years (Joung et al. 2016) for females and 5 years (Seki et al. 1998) to 6.8-8.9 years (Joung et al. 2016) for males. Thus, we can infer that all but three sharks measured were likely juveniles.

Of the 1,344 confirmed oceanic whitetip interactions, 156 oceanic whitetip sharks were identified according to sex, with 49% (77) identified as females and 51% (79) male oceanic whitetip sharks. The remaining (n = 1,188) were not identified (88%). While sharks can be visually sexed with ease by looking at the ventral surface (i.e., underside) for claspers, signifying a male shark, as previously discussed, many variables determine whether this data can be collected on an individual shark. Each interaction is unique and the observer cannot always maintain a visual line of sight on the animal through the entire interaction. Poor weather and sea state, the shark's size, condition at landing, whether the crew cuts the line or snaps from pressure, animal and crew safety can all influence an observers ability to identify claspers on a shark.

Spatial and Temporal Patterns of Exposure

All interaction data with oceanic whitetip sharks from the ASLL fishery were reviewed from 2006 through the second quarter of 2019 except for total annual interactions which were assessed through 2019. Comprehensively, incidental capture numbers are highest in March through and

May, with slight increase in November and December (Figure 16, orange bars). If interaction rates are considered by dividing the number of observed incidental captures by the number of observed sets each month, a similar pattern emerges, however March and April have much higher interaction rates of 0.060 and 0.067 sharks per set respectively compared to the next highest months of May and December which each have 0.044 sharks per set. All other months have fewer than 0.032 sharks per set (Figure 16, blue bars).

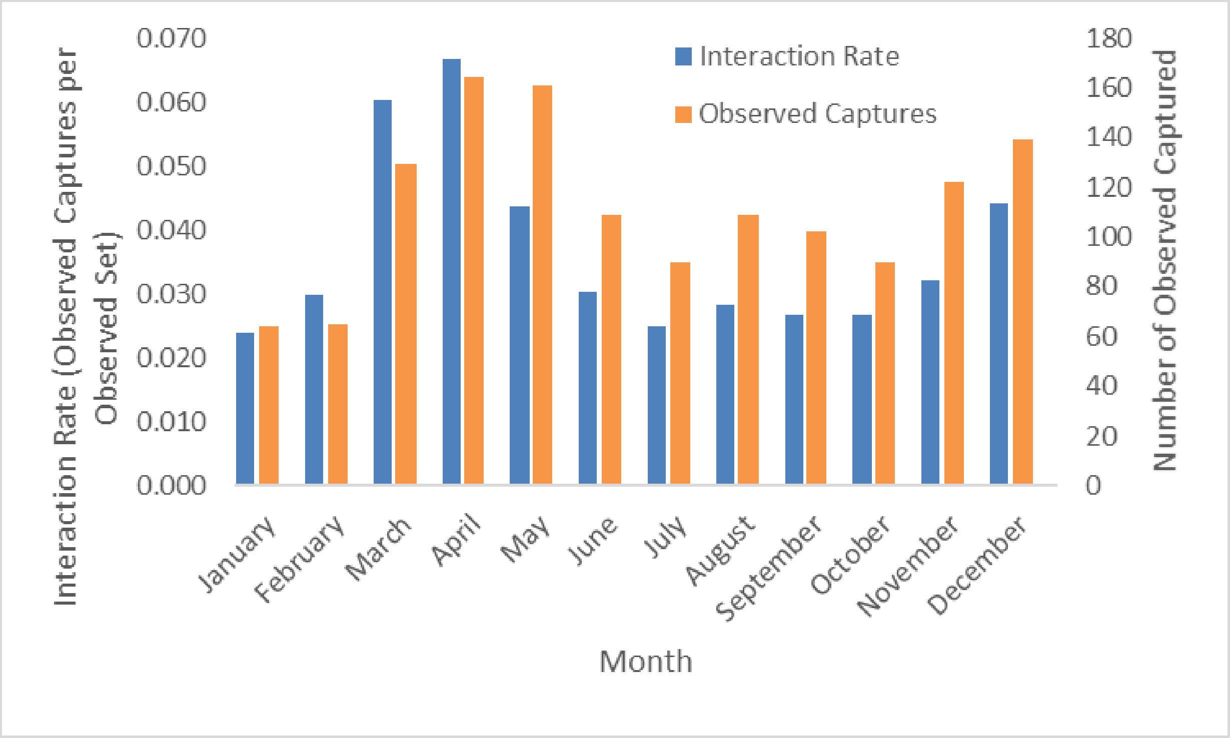


Figure 16. Total number of observed oceanic whitetip shark interactions (*n*= 1,344; orange bars) and interaction rates (observed shark captures divided by observed sets; blue bars) by month in the ASLL fishery, 2006 through 2019 (2nd quarter).

The highest estimated annual incidental captures of oceanic whitetip sharks occurred in 2010 (1,176 estimated incidental captures) with additional high years occurring in 2015 (827 estimated captures), 2016 (788 estimated incidental captures), and 2019 (870 estimated incidental captures; Figure 17). Overall, there is no trend in observed or estimated incidental captures per year.

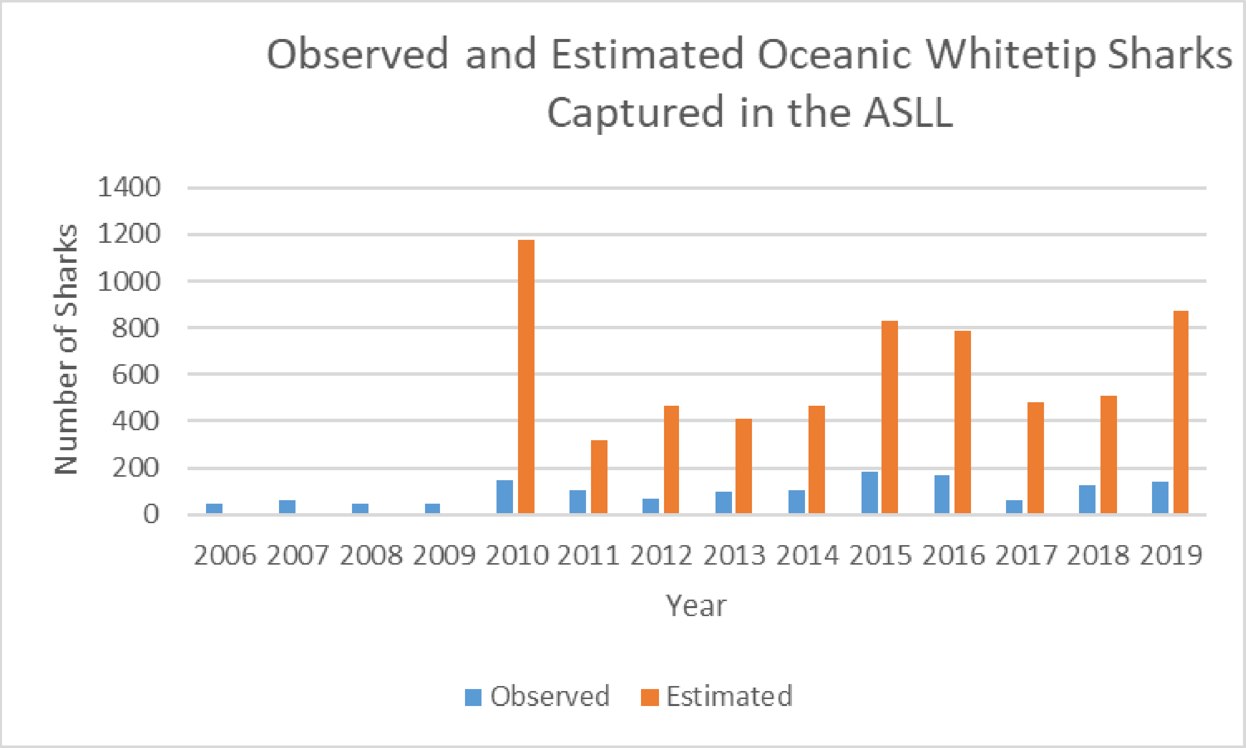


Figure 17. Observed (blue bars) and estimated (orange bars) oceanic whitetip shark interactions in the ASLL fishery, 2006-2019. Estimated numbers are from McCracken (2019a, 2020).

Using haul back locations, we mapped all sets where incidental interactions occurred using ArcGIS' (Figure 18) 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. The majority of the interactions occur in the north quadrant of the EEZ. Comparing the data to comprehensive longline effort, we can see the majority of the fishing effort is also in the northern quadrant although some small differences are apparent (see Figure 19). The number of interactions are slightly higher in density in the southwest corner of the EEZ; however reflect where fishing effort is occurring. Additionally, the northwest corner of the EEZ has a higher density of interactions when compared to a low fishing effort. Lastly, when looking just north of the Rose Atoll exclusion area, the density appears to be higher than the effort in this area. Overall, the interactions reflect where fishing effort is occurring with areas of higher density of interactions in small portions of the EEZ.

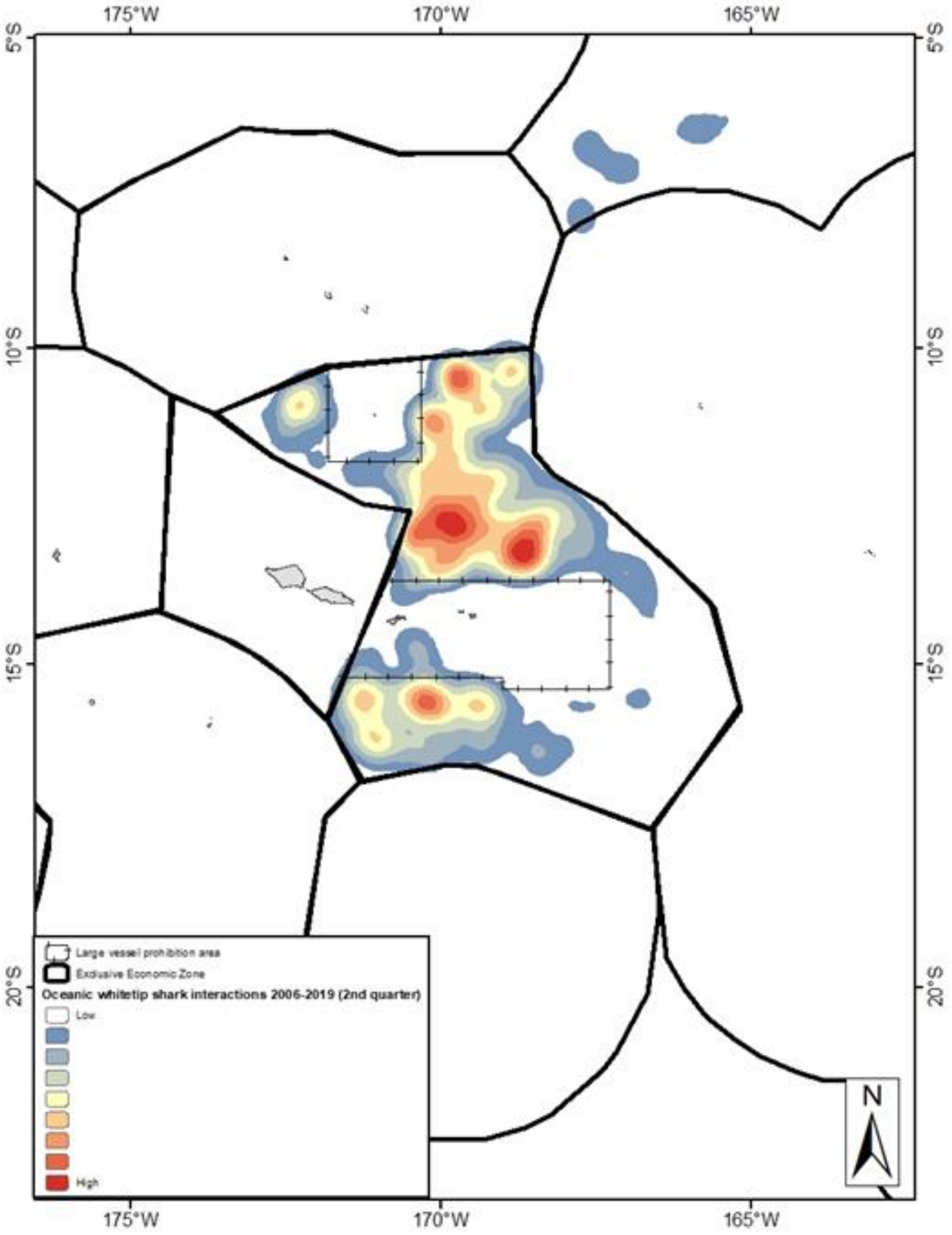


Figure 18. Geographical representation of aggregate oceanic whitetip shark interactions with the ASLL fishery between 2006 and 2019 (2nd quarter).

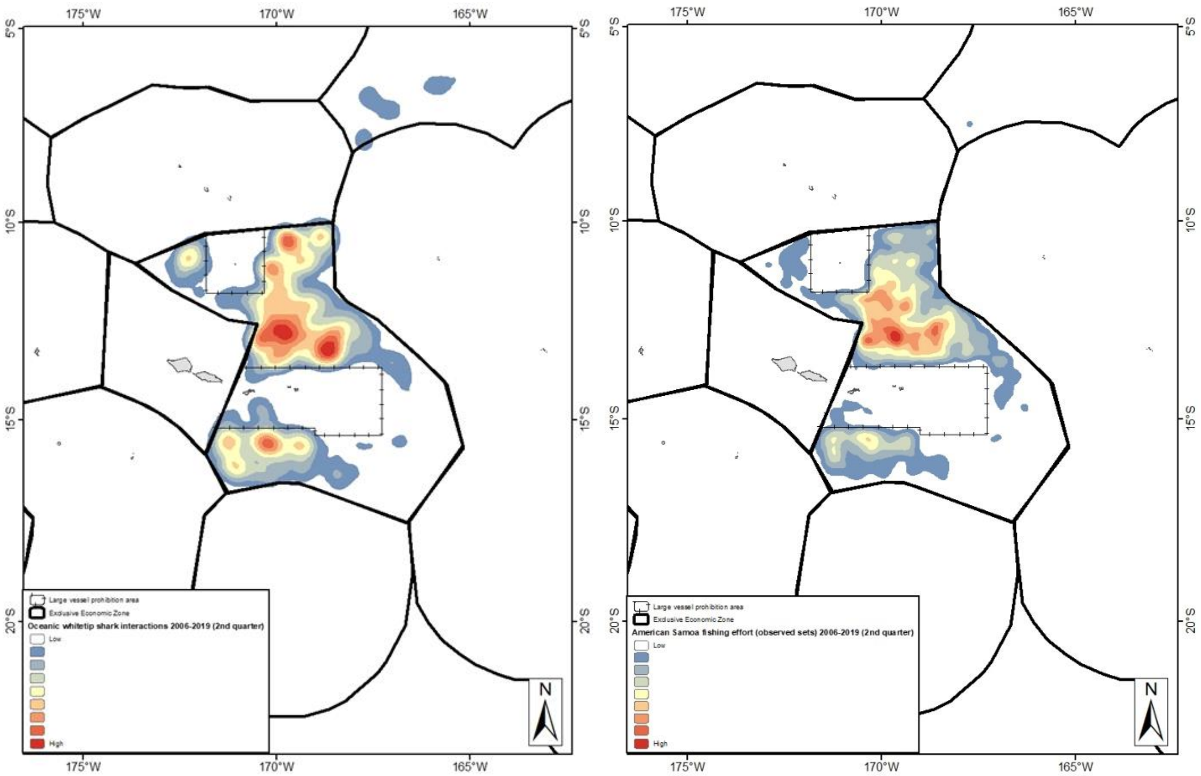


Figure 19. Side-by-side comparison of oceanic whitetip shark interactions (left; *n* =1,344 observed) versus fishing effort (right) from 2006 to 2019 (2nd quarter).

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interaction levels (McCracken 2019a; Table 14) using Bayesian statistical inference techniques as described in the *Approach to the Assessment* section of this biological opinion. We also report the anticipated mean number of annual mortalities associated with the 5-year running average and the cumulative number of anticipated mortalities over 5 years. There is no 95th percentile metric associated with the running average.

NMFS predicts that the ASLL fishery will incidentally capture a total of 6,964 oceanic whitetip sharks over the next 10 years. Over a 40 year time horizon, if captures remain consistent with historical numbers, we would anticipate a cumulative total of 27,856 incidental captures.

Table 14. Anticipated mean, 95th percentile (from McCracken 2019a; 2012-2017), maximum 5-year running average and maximum cumulative 5-year (2012-2019) exposures of oceanic whitetip sharks with fishing gear in the ASLL fishery. Numbers in parentheses for the maximum 5-year running average and running sum are inclusive of unidentified sharks assigned to oceanic whitetip sharks. Note that McCracken (2019a) did not provide mean and 95th percentile values for unidentified sharks and therefore we cannot calculate the prorated values for those metrics.

Number of years	Mean	95th percentile
1	620	1,110
Maximum 5-Yr Running Average	696.4 (704)	N/A
Maximum 5-Yr Running Sum	3,482 (3,520)	N/A

Response Analysis

Oceanic whitetip sharks in the ASLL 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.

At-Vessel Mortality

From 2010 to the 2nd quarter of 2019, 380 oceanic whitetip sharks were released dead, 762 were released alive, and 2 were retained. Using only the data for released sharks (i.e. excluding the one that was retained), the average at-vessel mortality is 33.3% (380/(380+762); 95% CI: 30.6% to 36.1%). Supporting data and the disposition of each interaction can be found in Table 13 (NMFS Observer Program unpublished data). We focus our analysis on the mean at-vessel mortality rate of 33.3%.

Post-Release Mortality

To-date, 35 oceanic whitetip sharks were tagged in the ASLL, of which two tags didn't report and two tags had to be removed from the analysis due to handling concerns. Of the remaining 31 tags, 4 died and 27 survived for a mean post-interaction mortality rate of 12.9% (95% CI: 5.1% to 28.9%; inferred from data in Hutchinson et al. 2021 and using the Wilson Score method to construct confidence intervals). Most of the mortalities occurred within 4 days of release, with one delayed mortality occurring at 22 days post release. The mortalities are reasonably attributable to the shark's interaction with the fishery with a high degree of confidence (Hutchinson et al. 2021). An additional 27 oceanic whitetip sharks were tagged in the DSLL with 25 viable tags and 4 mortalities (Hutchinson et al. 2021). To assess the tagged sharks for biases for uninjured sharks, Hutchinson et al. (2021) present release condition for encountered oceanic whitetip sharks (tagged and not tagged) but do not segregate the information between the ASLL and the DSLL, so we present the data for both fisheries combined. Of the 56 tags used to estimate post-interaction mortality rates in the ASLL and DSLL, 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 15; 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; therefore, the distribution of conditions for tagged oceanic whitetip sharks is representative of the conditions for all incidentally captured oceanic whitetip sharks in the ASLL and DSLL, with a slight bias for A and I conditions. We do note that of the 56 tags, 14 were MiniPAT tags designed for collecting long-term data (8 in the ASLL and 6 in the DSLL), and all of these were placed on AG condition code sharks to maximize the probability of longer-term data reporting (Hutchinson et al. 2021). Therefore, data from that subset of tags may not be representative of all oceanic whitetip sharks interacting with the fishery.

Table 15. Capture and release conditions for oceanic whitetip sharks tagged (reporting tags retained in the study only) in the DSLL and ASLL fisheries (Hutchinson et al. 2021). A= Alive; AG= Alive in Good Condition; I = Injured.

Capture Condition	Release Condition	Number	Percent
A	A	8	14.3
AG	A	1	1.8
AG	AG	41	73.2
AG	I	3	5.35
I	I	3	5.35
	Totals:	56	100

Hutchinson et al. (2021) conducted Bayesian survival analyses on the tag data for both fisheries. Overall, they found that condition at release, branchline leader material, and the amount of trailing gear left at release had the largest impact on post-release mortality. They present post-interaction survival rates at 1, 30, 60, 180, and 360 days post-interaction. We focus on the 30-day survival rates as being the most indicative of mortalities resulting from interaction with the fishery. Hutchinson et al. (2021) report 30 day survival rates for oceanic whitetip sharks interacting with the ASLL to be 0.72 (90% Credible Interval: 0.18-0.97; see Table 7 in Hutchinson et al. 2021), suggesting a mortality rate of 0.28 (0.03-0.82). We use this mean mortality value of 0.28 in our analysis.

The data from Hutchinson et al. (2021) are species and fishery-specific, and we therefore consider them to be the best available scientific data to inform post-interaction mortality rates for oceanic whitetip sharks incidentally captured in the ASLL. We also conducted a systematic literature review to place the mortality rates from Hutchinson et al. (2021) in context with results of other studies. 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). As noted previously, as the fishery uses monofilament leaders, we anticipate additional sharks may bite off the line and escape before being observed, however we have no information the frequency of this occurring for this fishery. It is also likely that any sharks that escape the line have a high survival rate and do not substantially contribute to mortalities caused by this fishery.

Total Mortality

Our calculated at-vessel mortality rate differs from that calculated by Hutchinson et al. (2021) for the ASLL slightly due to the different years of data used; therefore, we do not use the total mortality rates of Hutchinson et al. (2021) but defer to the raw observer data used for this analysis. To estimate total mortality rates (TM), we combine at-vessel (AVM) and post-interaction (PIM) mortality rates as follows:

$$TM = AVM + PIM(1 - AVM)$$

Using this equation with AVM = 33.3% and PIM = 28%, the total mortality rate for oceanic whitetip sharks interacting with the ASLL is 52%.

Table 16. The estimated number of oceanic whitetip sharks that are likely to die as a result of being incidentally captured in the ASLL per year, assuming a total mortality rate of 52%. We also report the anticipated mean number of annual mortalities associated with the 5-yr running average and the cumulative number of anticipated mortalities over 5 years. Numbers in parentheses for the maximum 5-yr running average and running sum represent the number of expected mortalities inclusive of unidentified sharks that are likely oceanic whitetip sharks.

Rate	**Number of Mortalities**
Annual Mean	322
Annual 95th Percentile	577
5-Year Running Average	362 (366)
5-Year Cumulative Sum	1,809 (1,829)

When considering the mean number of annual captures (*n*= 620; see Table 14), NMFS predicts that 322 oceanic whitetip sharks may die from their incidental capture in one year (Table 16). If annual incidental captures reach the 95th percentile, which could happen any given year but would not be likely to happen over a long time series, then we would expect 1,110 oceanic whitetip sharks to be incidentally captured in the ASLL fishery (see Table 14) resulting in the death of between 577 oceanic whitetip sharks.

Risk Analysis

In this section, we examine the probable consequences of incidentally capturing nearly 28,160 oceanic whitetip sharks in the ASLL fishery over the next 40 years (*n*= 704*40), and killing nearly 14,640 individuals during the same time period (366*40). As noted previously, most of our analysis focuses on incidental captures, mortality, and injury that can be associated with latent mortality.

Based on a mean number of interactions of 620 sharks, we estimate that about 322 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 1,110 individual interactions. Note that these numbers do not include unidentified sharks that were likely to be oceanic whitetip sharks. Based on the maximum 5-year running average, and including unidentified sharks that were likely to have been oceanic whitetip sharks, we estimate that about 704 oceanic whitetip sharks will be incidentally captured each year (*n*= 3,520 individuals every five years) in the ASLL fishery. From these animals, we estimate that 366 individuals will die as a result of the interaction (*n* = 1,829 mortalities in 5 years).

The best available genetic information for this species suggests that one population occurs in the Pacific Ocean. Although distinctions are often made between the East Pacific and the West Pacific, this distinction may be one of convenience based on management and is potentially biologically arbitrary. The only formal stock assessment for the Pacific only represents part of the population that likely occurs within the Pacific Ocean–the West and Central Pacific portion of the population’s range (aka. the West and Central Pacific stock). To assess the risk that the ASLL 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 ASLL 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 ASLL 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 ASLL 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 ASLL fishery interacts with about 0.08% of the Pacific population, and 0.04% of the population dies as a result of their interaction with the ASLL fishery in the initial years based on mean estimated interactions (Table 17, Table 18). If the population declines at a rate of 0.13% per year, by 2062 the population will have declined by 5% and the fishery will interact annually with about 0.10% of the population with 0.05% 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 0.08% of the population with 0.04% being killed.

Accumulating captures over the initial 10 years suggests that up to 0.45% of the population would be killed over 10 years if the population is increasing; 0.48% 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., 30°N and 35°S) 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 ASLL fishery interacts with approximately 0.05% 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 0.06% of the population with 0.03% 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 0.05% of the population with 0.02% being killed.

Accumulating incidental captures over the initial 10 years suggests that up to 0.27% of the population would be killed over 10 years if the population is increasing; 0.29% if the population is declining (Table 17, Table 18). Our estimates assume that the estimated population trends in the WCPO is indicative of the trend for the entire Pacific population.

Table 17. Estimated effect of ASLL 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 decrease).

	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	0.08	0.05	0.04	0.03
Max. 5-Year Running Average	2022	772,196	1,286,995	0.09	0.05	0.05	0.03
Max. 5-Year Running Average	2032	762,222	1,270,372	0.09	0.06	0.05	0.03
10-year based on 5-yr running average	2032	762,222	1,270,372	0.92	0.55	0.48	0.29
Max. 5-Year Running Average	2042	752,377	1,253,965	0.09	0.06	0.05	0.03
10-year Mean Projected 20 years	2042	752,377	1,253,965	0.94	0.56	0.49	0.29
Max. 5-Year Running Average	2052	742,660	1,237,769	0.09	0.06	0.05	0.03
10-year Mean projected 30 years	2052	742,660	1,237,769	0.95	0.57	0.49	0.30
Max. 5-Year Running Average	2062	733,067	1,221,782	0.10	0.06	0.05	0.03
10-year Mean projected 40 years	2062	733,067	1,221,782	0.96	0.58	0.50	0.30
95th Percentile	2062	733,067	1,221,782	0.15	0.09	0.08	0.05

Table 18. Estimated effect of ASLL 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 20% reduction in fishery mortalities (0.36% per year 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	0.08	0.05	0.04	0.03
Max. 5-Year Running Average	2022	783,632	1,306,052	0.09	0.05	0.05	0.03
Max. 5-Year Running Average	2032	812,356	1,353,925	0.09	0.05	0.05	0.03
10-year based on 5-yr running average	2032	812,356	1,353,925	0.87	0.52	0.45	0.27
Max. 5-Year Running Average	2042	842,134	1,403,557	0.08	0.05	0.04	0.03
10-year Mean Projected 20 years	2042	842,134	1,403,557	0.84	0.50	0.43	0.26
Max. 5-Year Running Average	2052	873,002	1,455,005	0.08	0.05	0.04	0.03
10-year Mean projected 30 years	2052	873,002	1,455,005	0.81	0.48	0.42	0.25
Max. 5-Year Running Average	2062	905,002	1,508,338	0.08	0.05	0.04	0.02
10-year Mean projected 40 years	2062	905,002	1,508,338	0.78	0.47	0.40	0.24
95th Percentile	2062	905,002	1,508,338	0.12	0.07	0.06	0.04

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 (2) that we believe likely paints a more realistic picture of the total effect of the ASLL 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 0.5% over 10 years.

Our assessment illustrates that the ASLL fishery interacts with (incidentally captures) a very small portion of the Pacific population of oceanic whitetip sharks. At projected abundance levels (1,292,023 individuals), incidentally capturing an average of 704 and killing 366 oceanic whitetip sharks from the Pacific Ocean population per year constitutes death of less than 0.03% of the population even if the population is decreasing at a rate of 0.13% per year. Even when we treat the WCPO stock estimate (775,000 individuals) as if it was a reasonable minimum estimate for the Pacific population, and assume that the population is declining at 0.13% per year, the ASLL fishery only kills 0.05% of the oceanic whitetip shark population annually. We consider the implications of this effect on OWT, as a species, in the *Integration and Synthesis* section of this opinion.

4.4.2 Giant Manta Ray

Exposure Analysis

Considering the filter-feeding nature of mobulid rays, the relatively low number of interactions in longline fisheries may represent a very low selectivity of this particular type of fishing gear (Mas et al. 2015). 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). In addition, giant manta rays are large animals, thus they are seldom landed and brought onboard the vessel due to their size. Therefore, they tend to be released with at least the hook attached and often with trailing gear.

Interactions between ray spp., including giant manta rays, in the ASLL fishery (McCracken 2019a; NMFS unpublished data) are listed in Table 19 below. Between 2006-2009, the American Samoa observer program was developing and resolving safety issues concerning placing observers on vessels and coverage was below 10%. Therefore, interactions prior to 2010 with protected species are not used to estimate the mean and 95th percentile values of future anticipated captures (McCracken 2019a). In addition, although the BE analyzed data through 2018, anticipated captures were requested earlier that year, therefore, the necessary data fields to analyze incidental captures of protected species through 2018 were not yet available. Therefore, the analyses of McCracken (2019a) cover the years of 2012 to 2017 (i.e., mean and 95th percentiles of annual capture numbers). McCracken (2020) used observer data from 2018 and 2019 to update yearly point estimates for those years, but did not update the mean and 95th percentile values. For the analysis in this opinion, we use the data from the full time series of

2012 to 2019 to calculate the maximum 5-yr running average/sum and we note the mean and 95th percentile values represent 2012 to 2017 data.

NMFS' observers are instructed to document interactions with rays by species when possible, but observations can include giant manta rays, mobula (devil rays), and unidentified rays. When the animal breaks the line or is able to come free of the hook before being pulled alongside the vessel, the observer may not be able to distinguish whether the ray species is a manta or mobula ray. In this situation, observers are instructed to use the classification *Mobulidae* spp. Likewise, if the observer is unable to identify the ray species as a manta or mobula, they are instructed to use the unidentified ray classification. In addition, observers are not currently instructed to distinguish between the coastal or reef manta, *Manta alfredi* and the giant manta ray, *M. birostris*. Therefore, the estimate of giant manta ray interactions may include the reef manta ray.

Table 19. Observed and estimated interactions with giant manta rays, mobulidae spp. and unidentified rays in the ASLL longline fishery from 2010-2019 (McCracken 2019a, 2020).

Year	**Manta Ray**		***Mobulidae* spp.**		**unidentified rays**	
-	**Obs**	**Est**	**Obs**	**Est**	**Obs**	**Est**
2006	-	-	-	-	-	-
2007	-	-	1	-	-	-
2008	-	-	-	-	1	-
2009	1	-	-	-	-	-
2010	3	11	1	12	0	0
2011	3	11	1	4	6	16
2012	3	29	0	0	0	0
2013	2	8	0	0	1	9
2014	1	2	1	3	0	5
2015	0	3	0	0	3	13
2016	0	0	2	7	0	4
2017	0	0	0	6	1	4
2018	0	0	2	8	-	-

Year	**Manta Ray**		***Mobulidae* spp.**		**unidentified rays**	
2019	0	0	4	24	-	-
Grand Total	13	64	12	64	12	56

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) estimated to be incidentally captured from 2010 to 2019 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 20). 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 estimated to be incidentally captured from 2012 to 2019 and used the Wilson Score Method to estimate the 95% confidence interval around the proportion of giant manta ray (Table 21). We multiplied that upper 95% CI by the number of *Mobulidae* spp. and unidentified rays that McCracken (2019a, 2020) estimated were incidentally captured each year from 2010 to 2019 to estimate the proportions of each unidentified category that were likely to have been giant manta ray. As noted previously, because there is no observer code for reef manta ray, some individuals identified as giant manta ray are likely to have been reef manta ray. This would inflate both the estimates of individuals identified as giant manta ray, as well as the proportion of unidentified individuals assigned to giant manta rays, however, at this time we do not have data to ascertain what the misidentification rate might be. In total, we estimate from 2010-2019, the ASLL fishery had 86 interactions with giant manta rays (64 estimated giant manta ray plus 22 from the *Mobulidae* spp. and 0 from the unidentified ray categories).

Table 20. The proportion of estimated incidental captures identified as giant manta ray and *Mobula* (devil rays) from 2010 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	64	164	228	0.28	0.72	0.23	0.34
Mobula (Devil Rays)	164	64	228	0.72	0.28	0.66	0.77

Table 21. The proportion of estimated incidental captures identified as giant manta ray, *Mobula* (devil rays), and pelagic stingrays from 2016 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	0	0	55,902	0.00	1.00	0.0000	0.0001
Mobula (Devil Rays)	73	0.0013	55,902	0.00	1.00	0.0010	0.0016
Pelagic Stingrays	55,829	0.998	55,902	1.00	0.00	0.9984	0.9990

Demographic Patterns of Exposure

The sex of manta rays that this fishery interacts with is unknown because the animals typically cannot be boarded, nor identified by sex during an interaction. We also cannot make inferences about the age of captured manta ray because observers do not collect size data."

Spatial and Temporal Patterns of Exposure

We plotted the observed confirmed giant manta ray, *Mobulidae* spp. and unidentified ray interactions that have occurred in the ASLL fishery between 2006 and 2019 in ArcGIS. We note the American Samoa observer program was developing and resolving safety issues concerning placing observers on vessels; coverage was below 10%. Therefore, interactions prior to 2010 with protected species are not used to estimate the anticipated number of interactions (McCracken 2019a) however, these interactions (and those that occurred after 2017) are included in the spatial and temporal exposure analysis.

Spatial remoteness between the data do not allow for viable density mapping products to be produced. The majority of all ray interactions were in the EEZ. However, five interactions were in the high seas northeast of the EEZ and in the northeast section of the Cook Islands EEZ. No spatial or temporal patterns are apparent from the data due to the small number of data points. The average hook number for these interactions was on hook number 14 out of an average of 30 hooks per float (NMFS unpublished data). Therefore, giant manta rays are being hooked at approximately 215 meters on the deepest hooks on the catenary curve, consistent with the hypothesis that they are feeding on the deep scattering layer (Dewar pers. comm. 2019).

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interaction levels (McCracken 2019a; Table 22) using Bayesian statistical inference techniques as described in the *Approach to the Assessment* section of this biological opinion. We also report the maximum 5-year running average and the cumulative number of anticipated incidental captures over 5 years. There is no 95th percentile metric associated with the running average and sum. We present both numbers based on observed and identified incidental captures and numbers that include unidentified individuals that are likely giant manta ray.

NMFS predicts that the ASLL fishery will incidentally capture a total of 114 giant manta rays over the next 10 years. Over a 40 year time horizon, if captures remain consistent with historical numbers, we would anticipate a cumulative total of 456 incidental captures.

Table 22. Anticipated mean, 95th percentile (from McCracken 2019a; 2012-2017), maximum 5-year running average and maximum cumulative 5-year (2012-2019) exposures of giant manta ray applying the summary proportions for *Mobulidae* spp. and unidentified rays estimated exposure in terms of anticipated interactions with fishing gear in the ASLL fishery. Numbers in parentheses are inclusive of unidentified rays assigned to giant manta rays.

Number of years	**Mean**	**95th percentile**
1	8.7 (11)	28 (33)
5-Yr Running Average	8.4 (11.4)	N/A
5-Yr Cumulative based on the 5-yr Running Average	42 (57)	N/A

Response Analysis

None of the 13 giant manta rays observed captured in the ASLL fishery was dead at capture resulting in a mean at vessel mortality rate of 0%. 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 13 giant manta rays observed interacting with the ASLL fishery, none were reported as dead at-vessel and we have no information on the condition at release other than they were released alive. 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 ASLL gear.

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 ASLL, 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. We also note that in the DSLL Supplemental biological opinion (NMFS 2022b) a second method was considered for estimating post-interaction mortality, assigning 100% mortality to entangled giant manta ray. We do not have any information for giant manta ray released from the ASLL fishery regarding whether they were entangled and we therefore cannot apply this method to estimate post-interaction mortality of giant manta ray in the ASLL.

Between 2006 and 2019 (2nd quarter; note we have total capture numbers for all of 2019 but details on individual captures only through 2nd quarter of 2019), there were a total of 37 observed interactions with giant manta rays, *Mobulidae* spp., and unidentified rays. The corresponding at vessel mortality rate for giant manta ray interactions in this fishery is 0 (95% CI: 0.00, 0.04), however, incorporating the Ryder et al. (2006) post release criteria, the post release mortality rate is 0.29 (0.17, 0.45). We used this mortality rate to estimate the total number of future mortalities from the adjusted mean and upper future capture estimates for giant manta rays. NMFS predicts that the ASLL fishery would interact with a mean of 11 and up to 33 giant manta rays each year. As a result, the number of giant manta rays that would likely die from their incidental capture would be between 4 and 10 over any one year. We also report the anticipated mean number of annual mortalities associated with the 5-yr running average. We also note there is no 95th percentile metric associated with the running average.

Table 23. Cumulative number of giant manta ray, *Mobulidae* spp., and unidentified rays captured in the ASLL fishery from 2006 through 2019 (2nd quarter) by their injury and release condition. Numbers in parentheses are the corresponding post-interaction mortality rates from Ryder et al. (2006) for leatherback sea turtles. The 95% confidence interval for the fishery mortality rate was calculated using the Wilson Score Method without continuity correction (Newcombe 1998).

Injury Category	**Line < 1/2 Disc Width**	**Line ≥ 1/2 Disc Width**	**No Gear**	**Grand Total**
External	1 (0.15)	34 (0.30)	1 (0.10)	36
Insertion Not Visible	-	-	-	0
Insertion Visible	-	-	-	0
Jaw	-	-	-	0
Total mortalities	0.15	**10.2**	**0.10**	**10.45**
Fishery mortality rate (95% CI)		**-**	-	0.29 (0.17, 0.45)

Table 24. Anticipated mortalities of giant manta ray interacting with fishing gear in the ASLL based on predicted exposure values for the anticipated mean, 95th percentile (from McCracken 2019a; 2012-2017), maximum 5-year running average and maximum cumulative 5-year (2012-2019) using the mean mortality rate of 0.29. Numbers in parentheses are the mortalities including unidentified individuals that are likely giant manta ray.

Period of ATL (yrs.)	**Mortality estimate for the mean**	**Adjusted upper estimate**
1	3 (3)	8 (10)
5-Yr Running Average	2.4 (3.3)	N/A
5-Yr Cumulative based on the 5-yr Running Average	12 (17)	N/A

Risk Analysis

We predict that the ASLL fishery will interact with a mean of 11 (95th percentile: 33) giant manta rays each year inclusive of unidentified individuals that are likely to be giant manta ray. The corresponding at vessel mortality rate for giant manta ray interactions in this fishery is 0 (95% CI: 0.00, 0.04). Using the leatherback sea turtle post-interaction mortality coefficients (Ryder et al. 2006), the number of giant manta rays that would likely die from their capture will average 3 (95th percentile 10) in any one year (Table 24). Over 10 years, using the maximum 5-year running sum of 57 captures, we anticipate 114 captures with 33 mortalities. Over 40 years, if captures remain the same, we expect a total of 45 captures with 132 mortalities.

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: they are separate populations. 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 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 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 populations exposed to the ASLL fishery, and the size of those populations is unknown; however, these are important attributes necessary for understanding the effect of fishery-related captures and mortalities on giant manta rays and their risk of extinction. As mentioned previously, the *Action Area* of the ASLL fishery overlaps with the southeastern *Action Area* of the United States WCPO purse seine fishery. Therefore, to approximate the number of giant manta ray populations that may be affected by the ASLL fishery, we incorporated a similar analysis to the United States WCPO purse seine fishery biological opinion (NMFS 2021) and DSLL supplement (NMFS 2022b) with some modifications to adapt to the specifics of the ASLL fishery as appropriate.

Known giant manta ray subpopulations are found in oceanic waters near productive coastlines, continental shelves, offshore pinnacles, seamounts or oceanic islands. Hence, if a cluster of giant manta ray captures was observed not associated with known populations reported in the literature, we assumed the presence of a subpopulation and centered it on a feature such as an island or seamount. Although directed studies that identify population substructure are preferred, this was the best use of available proxies for delineating independent demographic units (aka subpopulations) of giant manta rays. While this introduces uncertainties in our definition of subpopulations, given (1) what is known from the literature about limited movements of giant manta ray within home ranges (e.g., Graham et al. 2012; Stewart et al. 2016a; Beale et al. 2017) and (2) the fact that the observer data indicates giant manta ray occur in the area, we consider this the best approach to ensure we are capturing likely subpopulations. Overall we identified seven potential subpopulations that may be impacted by the ASLL fishery (Table 25). Six of these were identified in the analysis conducted for the United States WCPO purse seine fishery biological opinion (NMFS 2021), the seventh, American Samoa, was identified based on the distribution of observed giant manta ray incidentally captured by the ASLL fishery.

As noted in the *Status of Listed Species* section, several studies have tracked individual giant manta rays and provided information on the spatial extent of giant manta ray subpopulations.

The analysis in the United States WCPO purse seine fishery biological opinion (NMFS 2021) uses 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). The resulting values ranged from 457 km in Bahia de Banderas, Mexico to 590 km in Isla de la Plata Ecuador (Figure 20). Therefore, 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, the radius was increased to 500 km in order to have a higher level of confidence to encompass all fishery captures that are affecting 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, NMFS (2020) assumed that up to 15% of individuals belonging to a specific population would be between 500 and 1500 km away from the center point of the aggregation.

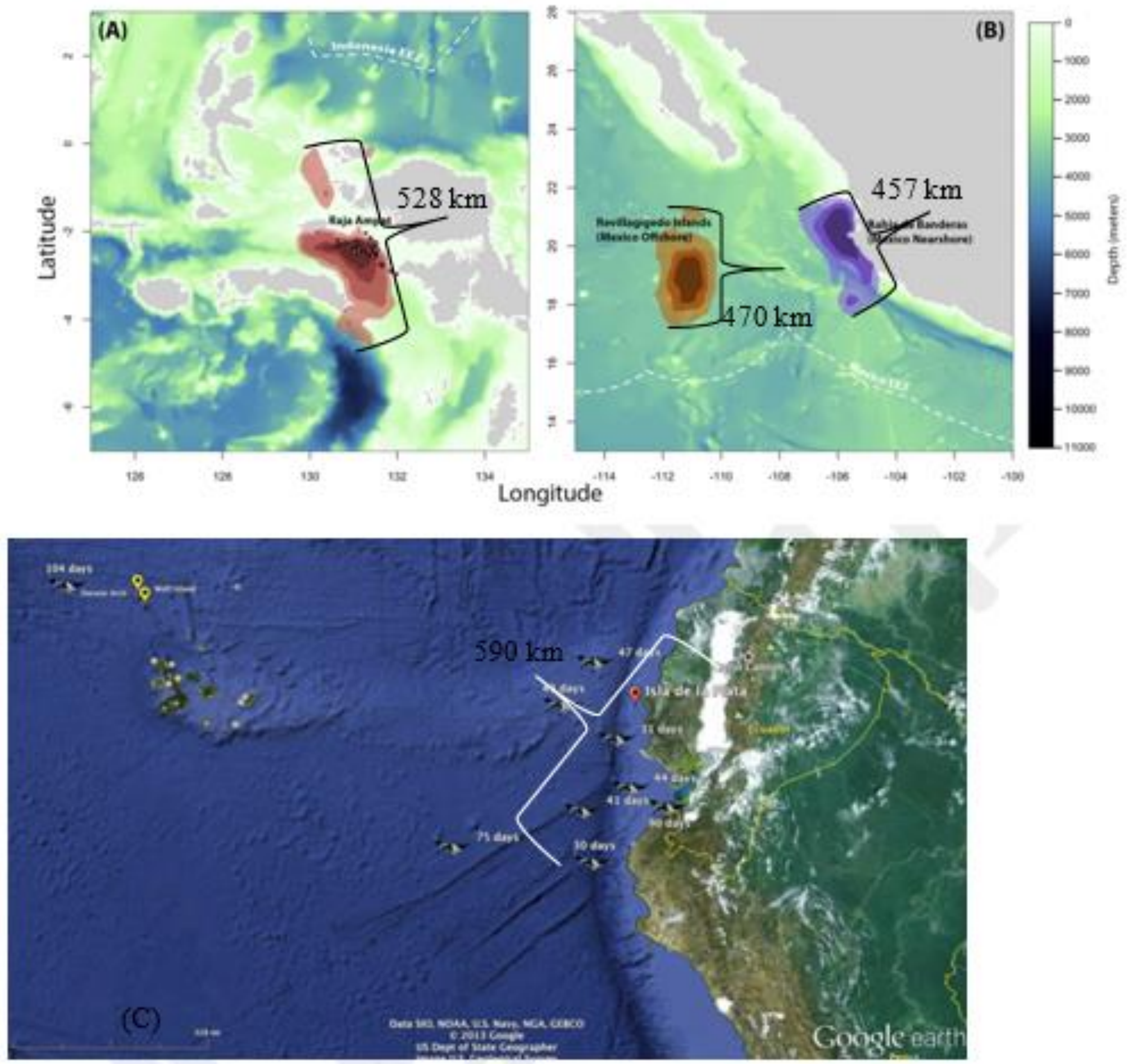


Figure 20. Extents of giant manta ray subpopulations based on tracking tagged individuals. Black brackets (white brackets in C) highlight the longest dimension of the area of occupancy with the length noted in kilometers (km). Locations: A) Raja Ampat, Indonesia (Stewart et al. 2016a); B) Revillagigedo Islands (left) and Bahia de Banderas (right), Mexico (Stewart et al. 2016a); C) Isla de la Plata, Ecuador (Hearn et al. 2014).

Assuming a cluster of giant manta ray captures observed was not associated with known subpopulations reported in the literature, incorporating the analysis used in our biological opinion on the United States WCPO purse seine fishery (NMFS 2021), we assumed the presence of a subpopulation and centered it on a feature such as an island or seamount. Therefore, we included a population in American Samoa based on the 15 observed incidental captures of potential giant manta rays mostly within the American Samoa EEZ, and centered this subpopulation on the island of Tau. Integrating the subpopulations identified in NMFS (2021), we identified a total of seven giant manta ray subpopulations that may be impacted by the ASLL fishery (Table 25; Figure 21). For the Tuvalu and Kiribati-Line Islands subpopulations in Table 25, no observed captures occurred within 500 km of the center point of these two subpopulations, and 1 (Tuvalu) or 2 (Kiribati-Line Islands) occurred within between the 500 and 1,500 km radius buffers, however when the 15% was applied to this region, we estimate no impacts for these subpopulations. It is possible that the fishery may have a small impact on these two subpopulations but we do not consider them further in this analysis and focus only on the other 5 subpopulations listed in Table 25. The locations of these 5 subpopulations are depicted in Figure 21 where the American Samoa subpopulation is shown in dark orange. All 15 observed giant manta ray incidentally captured occurred within this 1,500 km boundary, with 13 of the 15 occurring within the 500 km boundary. The paler orange 500 km circles and 1,500 km open circle boundaries represent the Cook Islands (to the south) and Kiribati-Phoenix Islands West (to the north) subpopulations. One of the observed giant manta ray captures occurred in the 500 km boundary in each of these subpopulations. The purple 500 km circles and 1,500 km open circle boundaries represent the Fiji (to the south) and Kiribati-Phoenix Islands East (to the north) subpopulations. No observed giant manta ray captures occurred within the 500 km boundary for these subpopulations, only within the 1,500 km boundary.

The anticipated annual exposures based on the 5-yr running average and the annual 95th percentile for the five subpopulations sum to more than the total anticipated annual values for each metric. This is because observed giant manta ray captures may be assigned to more than one subpopulations given the overlapping nature of the 500 and 1,500 km radii (Table 25, Figure 21).

Table 25. Giant manta ray subpopulations potentially impacted by the ASLL 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 (NMFS 2021). For the anticipated exposures we report the maximum 5-yr running average along with the annual 95th percentile from McCracken (2019a).

Giant Manta Ray Subpopulation	Center Point Latitude	Center Point Longitude	Observed Giant Manta Ray Captures	Anticipated Annual Exposures based on the 5-yr running Average (95th percentile)	Source
Tuvalu	-6.084	177.367	0	0 (0)	Inferred from United States WCPO purse seine capture locations, centered on the island of Tuvalu
Kiribati Phoenix Islands East	-4.713	-174.376	2	1 (4)	Inferred from United States WCPO purse seine capture locations, centered on seamount locations from Macmillan-Lawler and Harris (2016)
Kiribati Phoenix Islands West	-3.032	-167.468	3	2 (7)	Inferred from United States WCPO purse seine capture locations, centered on seamount locations from Macmillan-Lawler and Harris (2016)
Fiji	-14.115	179.021	2	1 (4)	O’Malley et al. 2013
Kiribati, Line Islands	1.897469	-157.424	0	0 (0)	O’Malley et al. 2013, centered on the island of Kiritimati (Christmas Island)
Cook Islands	-13.026	-163.608	3	2 (17)	O’Malley et al. 2013
American Samoa*	-14.232	-169.463	15	11 (33)	Inferred from American Samoa capture locations, centered on the island of Tau

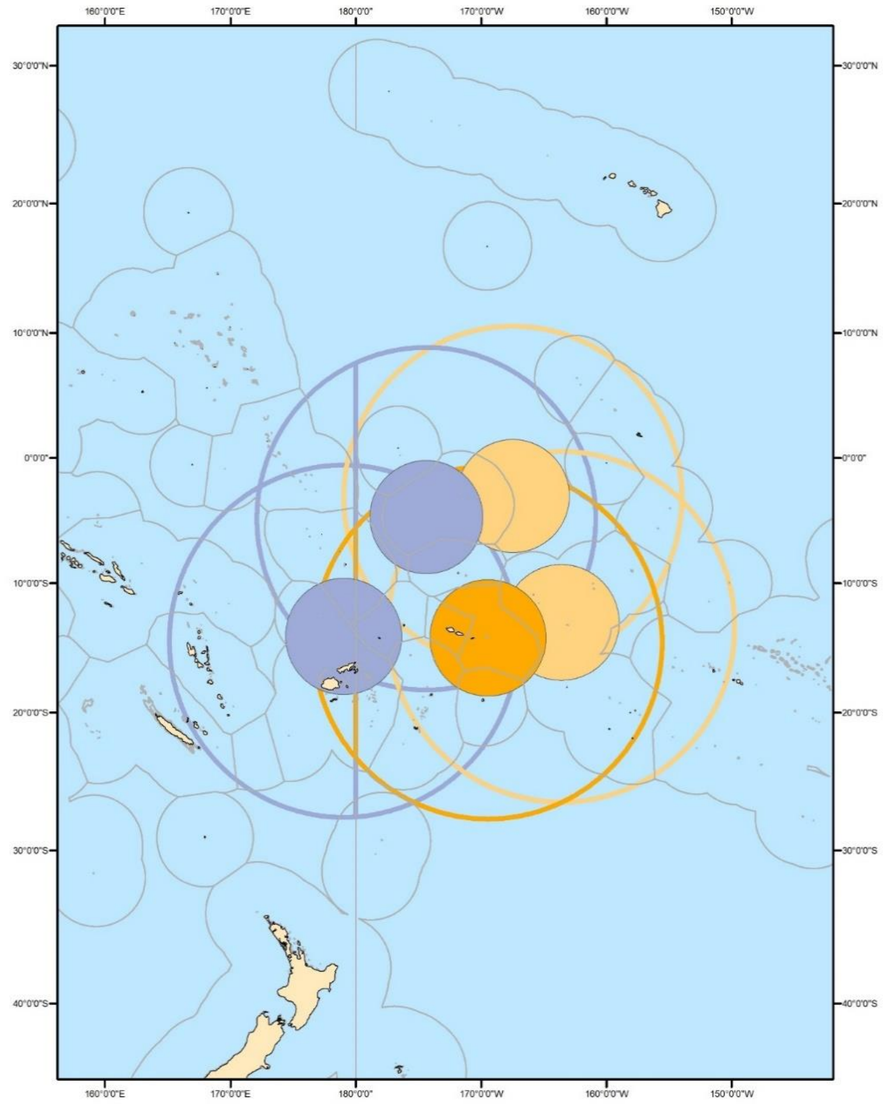


Figure 21. Map of fishing area for the ASLL fishery showing the five purported giant manta ray subpopulations potentially impacted by the ASLL fishery. Dark orange circle represents the 500 km boundary of the purported American Samoa subpopulation, the dark orange open circle represents the 1,500 km boundary around this subpopulation. The paler orange 500 km circles and 1,500 km boundaries represent the Cook Islands (to the south) and Kiribati-Phoenix Islands West (to the north) subpopulations. The purple 500 km circles and 1,500 km boundaries represent the Fiji (to the south) and Kiribati-Phoenix Islands East (to the north) subpopulations.

To overcome the absence of information on the number of manta ray subpopulations that are likely to be exposed to the ASLL fishery, the size of those subpopulations, their connectivity, and identification uncertainty, we developed several scenarios to assess the probable effects of the ASLL 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 (2021) 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 (2021) 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 (2021) 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[3]. Therefore, for this analysis, we follow the same methodology we are used in the US WCPO purse seine biological opinion (NMFS 2021) and focus on r_{max} values of 0.019 and 0.042.

Both 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 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 et al. 2017; Beale et al. 2019). As noted in the *Status of Listed Species*, 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 each scenario, population abundance was projected over 10 years, 20 years, 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 population based either on the maximum 5-yr running average (Scenario 1) or the annual upper 95th percentile (Scenario 2, Table 24) multiplied by the mean estimated post-interaction mortality rate of 0.29. 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

---

[3] 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 r_{max} for giant manta ray.

a worst-case analysis for the approximate maximum number of interactions we anticipate will occur in any given year.

We anticipate that misidentification is an issue with the ASLL 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 (or even other mobulids). However, we have no information as to the magnitude of any potential misidentification rate for this fishery. Until data can be obtained regarding this issue, we assume all observed individuals recorded as giant manta ray 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: five discrete populations; 29% post-release mortality rate; maximum 5-yr running average for anticipated future annual interactions; 10, 20, and 40 years.
>
> Scenario 2: five discrete populations; 29% post-release mortality rate; 95th percentile of anticipated future annual interactions; 10, 20, and 40 years.

Table 26. Parameters and values used for all scenarios.

Parameter	**Values**
Initial Population Abundance	60, 100, 200, 400, 600, 800, 1000, 1500, 1875, 2464
Fishery Captures per Year, per Population	2 to 33
Post Release Mortality Rate	0.29
Intrinsic Population Growth Rate	0.019, 0.042

In Scenario 1, for the American Samoa subpopulation, we found that if all anticipated captures from the fishery (based on the 5-yr running average) are attributed to this subpopulations, populations smaller than 100 individuals would decrease, while larger populations would continue to increase across all timeframes for $r_{max} = 0.019$ (Table 26). For the other four subpopulations potentially impacted by the fishery, all continue to increase at all initial population abundances (Table 26). If $r_{max} = 0.042$, all combinations of subpopulations and initial abundances had increasing population trajectories with the exception of 60 individuals for American Samoa.

At the 95th percentile (Scenario 2), the American Samoa subpopulation would decrease if the initial abundance is less than 400 ($r_{max} = 0.019$) or 200 ($r_{max} = 0.042$) individuals (Table 26) otherwise it would increase. The Cook Islands and Kiribati – Phoenix Islands West subpopulations would decline if initial abundance was less than 100 and $r_{max} = 0.019$, at larger abundances we found increasing population trajectories and all initial abundances had increasing population trajectories if $r_{max} = 0.042$ (Table 26). The remaining subpopulations had increasing population trajectories at all initial abundances and values of r_{max} (Table 26).

The data available are not sufficient data to answer the question: "how many giant manta ray populations are likely to be exposed to the longline fisheries and what is their estimated abundance?" However, the data available suggest: that giant manta ray populations or subpopulations 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.

Table 27. Results of scenarios 1 and 2 for the American Samoa subpopulation where the 5-year running average estimated number of interactions is 11 per year, or 33 per year under scenario 2. Results are given for two values of the maximum intrinsic population growth rate (r_{max}; 0.019 and 0.042). Numbers represent the percent difference between initial abundance and abundance the final year of the simulation (10, 20, or 40 years); red numbers represent population declines (population abundance in year 10 less than population abundance in year 0) and black numbers represent population increases.

Initial Abundance	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
American Samoa; Scenario 1; 10 years										
R_{max} = 0.019	0.34	0.12	0.05	0.13	0.15	0.17	0.18	0.19	0.19	0.20
R_{max} = 0.042	0.09	0.16	0.34	0.43	0.46	0.48	0.49	0.50	0.50	0.51
American Samoa; Scenario 1; 20 years										
R_{max} = 0.019	0.74	0.26	0.10	0.28	0.34	0.37	0.39	0.41	0.42	0.43
R_{max} = 0.042	0.22	0.40	0.86	1.09	1.16	1.20	1.22	1.25	1.27	1.28
American Samoa; Scenario 1; 40 years										
R_{max} = 0.019	1.00	0.64	0.25	0.69	0.84	0.92	0.96	1.02	1.04	1.07
R_{max} = 0.042	1.00	0.72	0.72	1.31	2.84	3.60	3.86	3.98	4.06	4.16
American Samoa; Scenario 2; 10 years										
R_{max} = 0.019	1.00	0.88	0.34	0.06	0.03	0.07	0.10	0.14	0.15	0.16
R_{max} = 0.042	1.00	0.69	0.09	0.22	0.32	0.37	0.40	0.44	0.46	0.47
American Samoa; Scenario 2; 20 years										
R_{max} = 0.019	1.00	1.00	0.74	0.14	0.06	0.16	0.22	0.30	0.33	0.36
R_{max} = 0.042	1.00	1.00	0.22	0.55	0.80	0.93	1.01	1.11	1.15	1.19
American Samoa; Scenario 2; 40 years										

Initial Abundance	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
R_{max} = 0.019	1.00	1.00	1.00	0.35	0.15	0.40	0.54	0.74	0.82	0.90
R_{max} = 0.042	1.00	1.00	0.72	1.82	2.67	3.09	3.35	3.69	3.82	3.95

Table 28. Results of scenarios 1 and 2 for the Cook Islands and Kiribati Phoenix Islands West subpopulations where the 5 yr running average estimated number of interactions is 2 per year and 95th percentile of 7 under scenario 2 for both subpopulations. Results are given for two values of the maximum intrinsic population growth rate (r_{max}; 0.019 and 0.042). Numbers represent the percent difference between initial abundance and abundance the final year of the simulation (10, 20 or 40 years); red numbers represent population declines (population abundance in year 10 less than population abundance in year 0) and black numbers represent population increases.

Initial Abundance	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**
Cook Islands and Kiribati Phoenix Islands West; Scenario 1; 10 years									
R_{max} = 0.019	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20
R_{max} = 0.042	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52
Cook Islands and Kiribati Phoenix Islands West; Scenario 1; 20 years									
R_{max} = 0.019	20	0.06	0.22	0.34	0.40	0.42	0.43	0.44	0.45
R_{max} = 0.042	0.80	1.01	1.16	1.24	1.27	1.28	1.29	1.30	1.30
Cook Islands and Kiribati Phoenix Islands West; Scenario 1; 40 years									
R_{max} = 0.019	0.15	0.54	0.84	0.99	1.04	1.06	1.08	1.10	1.11
R_{max} = 0.042	2.67	3.35	3.86	4.11	4.20	4.24	4.26	4.30	4.31
Cook Islands and Kiribati Phoenix Islands West; Scenario 2; 10 years									
R_{max} = 0.019	0.15	0.01	0.10	0.15	0.17	0.18	0.19	0.19	0.20
R_{max} = 0.042	0.12	0.28	0.40	0.46	0.48	0.49	0.50	0.51	0.51
Cook Islands and Kiribati Phoenix Islands West; Scenario 2; 20 years									
R_{max} = 0.019	0.34	0.02	0.22	0.34	0.38	0.40	0.41	0.43	0.44
R_{max} = 0.042	0.29	0.70	1.01	1.16	1.21	1.24	1.25	1.28	1.28
Cook Islands and Kiribati Phoenix Islands West; Scenario 2; 40 years									
R_{max} = 0.019	0.84	0.05	0.54	0.84	0.94	0.99	1.02	1.06	1.07
R_{max} = 0.042	2.67	3.35	3.86	4.11	4.20	4.24	4.26	4.30	4.31

Table 29. Results of Scenarios 1 and 2 for the Kiribati Phoenix Islands East and Fiji subpopulations where the 5 yr running average estimated number of interactions is 1 per year and 95th percentile of 4 under scenario 2 for both subpopulations. Results are given for two values of the maximum intrinsic population growth rate (r_{max}; 0.019 and 0.042). Numbers represent the percent difference between initial abundance and abundance the final year of the simulation (10, 20 or 40 years); red numbers represent population declines (population abundance in year 10 less than population abundance in year 0) and black numbers represent population increases.

Initial Abundance	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**
Kiribati Phoenix Islands East and Fiji; Scenario 1; 10 years									
R_{max} = 0.019	0.21	0.21	0.21	0.21	0.21	0.21	0.21	0.21	0.21
R_{max} = 0.042	0.52	0.52	0.52	0.52	0.52	0.52	0.52	0.52	0.52
Kiribati Phoenix Islands East and Fiji; Scenario 1; 20 years									
R_{max} = 0.019	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46
R_{max} = 0.042	1.32	1.32	1.32	1.32	1.32	1.32	1.32	1.32	1.32
Kiribati Phoenix Islands East and Fiji; Scenario 1; 40 years									
R_{max} = 0.019	1.14	1.14	1.14	1.14	1.14	1.14	1.14	1.14	1.14
R_{max} = 0.042	4.37	4.37	4.37	4.37	4.37	4.37	4.37	4.37	4.37
Kiribati Phoenix Islands East and Fiji; Scenario 2; 10 years									
R_{max} = 0.019	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20
R_{max} = 0.042	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52
Kiribati Phoenix Islands East and Fiji; Scenario 2; 20 years									
R_{max} = 0.019	0.06	0.22	0.34	0.40	0.42	0.43	0.44	0.45	0.45
R_{max} = 0.042	0.80	1.01	1.16	1.24	1.27	1.28	1.29	1.30	1.30
Kiribati Phoenix Islands East and Fiji; Scenario 2; 40 years									
R_{max} = 0.019	0.15	0.54	0.84	0.99	1.04	1.06	1.08	1.10	1.11
R_{max} = 0.042	2.67	3.35	3.86	4.11	4.20	4.24	4.26	4.30	4.31

4.5 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.

For our cumulative effects analysis for this consultation we looked to state and regional maps and land use plans, and other similar sources of information that indicate plans for the harbor areas where the fishery vessels are moored, or other changes to state fisheries and state waters, to the extent practicable. For activities farther to the center of the *Action Area* (pelagic areas) based on our prior experience, we expect most of the private or non-federal activities would amount to foreign fisheries, which is considered under the *Environmental Baseline* section of this opinion and are expected to continue into the future.

In our review of the literature, we found a number of state managed vessel-based fisheries which exist in American Samoa that fish pelagic waters (up to 25 miles offshore) potentially overlapping with a portion of the longline fishery, and have done so since the 1970s. We note, while there is a shoreline subsistence fishery in American Samoa, we would not expect overlap to occur with the longline fishery.

Craig et al. (2013) discusses three artisanal fisheries, the pelagic troll fishery, the bottom handline fishery, and a pelagic tournament fishery. Additionally, a small boat (alia) longline fleet has operated in American Samoa since the 1990s (Kleiber and Leong 2018). However, we highlight that overlap only occurs with these various fisheries for vessels transiting activities through geographic space rather than fishing activities as the longline fleet is required by regulation to fish farther offshore. As previously mentioned, the LVPA extends seaward around the various islands restricting vessels from this fishery (monohull longline vessels >50 ft. in length) from fishing for pelagic management unit species (Kleiber and Leong 2018; NMFS 2019a). Although vessels exempted from the LVPA may potentially fish in waters where these fisheries occur, due to the current exclusion zone and lack of known interactions in this area because of the exclusion, we cannot forecast where interactions will occur once longline vessels begin fishing this area. It is likely the longline vessels will fish at a deep bathymetric profile, but we are uncertain where these other fisheries fish. Furthermore, we note that the federally managed bottomfish fishery (discussed in the *Environmental Baseline*) should not be confused with the recreational bottomfish *handline* fishery.

Reported catch composition from these various artisanal and recreational fisheries are only target catch species and do not categorize any potential interactions with ESA-listed species that we are aware of. Therefore, we cannot determine what level of effects these fisheries may have on the species considered herein. However, these activities are ongoing and are expected to continue into the future. With the exemptions to the LVPA, vessels from this fishery fish in areas where these current fisheries operate providing competition for catch.

Various nearshore (mostly recreational) fisheries such as shallow bottomfishing, reef trolling, spearfishing, whipping/casting, trapping, and netting also occur (Loomis et al. 2019). Again, vessels from the ASLL fleet would only overlap these various fisheries when transiting to and from shore. Like the pelagic fisheries, we cannot categorize the effects to ESA-listed species considered herein but these activities are ongoing and are expected to continue into the 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 action area 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 authorization of the ASLL, which we expect would continue in perpetuity since longlining has occurred since the early 1990s. 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 the ASLL 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 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.

5.1 Oceanic Whitetip Shark

On average the ASLL fishery is expected to incidentally capture a mean of 620 (95th percentile: 1,110) threatened oceanic whitetip sharks each year, in some years, numbers higher or lower are likely to be incidentally captured. Based on a total mortality rate of 0.52, we expect 322 of the oceanic whitetip sharks incidentally captured are likely to die as a result of their incidental capture each year. Over a 40 year analytical time frame we would anticipate a cumulative total of about 27,856 incidental captures and 14,473 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 (Rice and Harley 2012; Futerman 2018). The potential impacts from climate change on oceanic whitetip shark habitat are highly uncertain, but given their broad distribution in various habitat types, these species may be able 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 ASLL 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 incidental capture in fisheries. 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 ASLL fishery under its current operation (Rice et al. 2021).

The action results in the capture of an average of 620 oceanic whitetip sharks and the resulting death of an average of 322 sharks per year. The maximum 5-year running average is 704 sharks with 366 mortalities. At projected abundance levels (1,292,023 individuals), this constitutes death of 0.03% of the population per year. Even when we treat the WCPO stock estimate (775,000 individuals) as if it was a reasonable minimum estimate for the Pacific population, the ASLL fishery only removes 0.05% of the current population annually.

We do not have sufficient data to accurately estimate the current population trend. Therefore, we conducted our assessment based on a range from -0.13% to +0.36%, but that range includes removing animals at the historic mortality rate from the ASLL as currently managed. This means without the proposed action, the population trends would range from -0.08% to 0.41% per year. Furthermore, the best available information (Bigelow et al. 2022) indicates the current population trend is positive (including removing animals at the historic mortality rate). Therefore, we are reasonably certain the population trend will continue to be positive with the continued authorization of the American Samoa longline fishery as currently managed. Because oceanic whitetip shark population abundance will continue to grow with the proposed action, we are reasonably certain it will not cause material changes having biological consequences to the species' numbers, reproduction, or distribution. In accordance with Section 1.3.1 (*Jeopardy Analyses*) above, NMFS does not anticipate the ASLL fishery will reduce appreciably the likelihood of the survival or recovery of oceanic whitetip sharks in the wild by reducing their reproduction, numbers, or distribution.

5.2 Giant Manta Ray

As described in the *Status of Listed Species,* the giant manta ray is listed as threatened throughout its range and there were no abundance estimates available for the species when it was listed because the species is only sporadically observed. More recent data has provided abundance estimates for some well-studied subpopulations (Table 4); however, estimates of the global population size as well as population sizes within the *Action Area* remain unavailable. The species appears to have a population substructure that is composed of independent demographic units or subpopulations, with small distinct home ranges. There are a few available estimates of the size of subpopulations, but many available counts are opportunistic and likely do not represent robust population estimates. In general, viable giant manta ray subpopulations are likely to be larger than available count data suggest, and likely contain more than 1,000 individuals (Joshua Stewart, Manta Trust, pers. comm. to A. Garrett 2021). These population sizes are in keeping with the current understanding of effective population sizes necessary for the genetic diversity needed to maintain evolutionary fitness in naturally outbreeding diploid species (N_e>1,000; Frankham et al. 2014). 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 a reduced possibility of successful recolonization (Marshall et al. 2018).

Overall, in many regions, the numbers of giant manta ray appear be declining, 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).

The most significant and certain threat to the giant manta ray is overutilization for commercial purposes and non-targeted bycatch and fishery interactions. Targeted fisheries for mantas have existed for decades and historically, the giant manta ray was exploited for meat, cartilage, and skin (Heinrichs et al. 2011; Lewis et al. 2015). However, driven by the international trade in gill plates, fisheries targeting mantas have expanded and pose a serious threat to the giant manta ray (CITES 2013). In addition, giant manta rays are frequently caught as bycatch in a number of commercial and artisanal fisheries worldwide, particularly commercial longline, trawl, purse-seine and gillnet fisheries off Europe, western Africa, the Atlantic coast of the United States, Australia, the Indian Ocean, and within the Pacific.

As described in the *Environmental Baseline*, giant manta ray are currently affected by several stressors within the *Action Area,* including climate change, fisheries, vessel strikes, and marine debris; however both direct harvest and bycatch in fisheries are the dominant stressor in the baseline that affects the species. Effects from the ASLL and other United States fisheries have resulted in interactions with the giant manta ray in the *Action Area*. The United States WCPO purse seine fishery is estimated to capture a maximum 5-year average of 47 per year and up to 50 in any one year. Between 2012 and 2015, there were 348 observed interactions with giant manta rays in the international WCPO longline fishery (Tremblay-Boyer and Brouwer 2016) which, based on the approximate 2.4% observer coverage of this fishery, suggests approximately 14,500 giant manta rays were captured over that time period. United States fisheries that operate out of the West Coast regions are not known to capture giant manta ray.

Large-scale impacts that affect ocean temperatures, currents, and potentially food chain dynamics, may pose a threat to this species. However, given the migratory behavior of the giant manta ray and its tolerance to both tropical and temperate waters, these animals may be able to shift their range or distribution to remain in an environment conducive to their physiological and ecological needs, which may make them more resilient to these effects.

As described in the *Effects of the Action* we predict that the ASLL fishery will interact with a mean of 11 and *up to* 33 giant manta rays each year. The corresponding at vessel mortality rate for giant manta ray interactions in this fishery is 0 (95% CI: 0.00, 0.04). Using the leatherback sea turtle mortality coefficients (0.29, Ryder et al. 2006), the number of giant manta rays that would likely die from their capture would be up to 10 in any one year (Table 24).

If individual giant manta rays do not immediately die from their encounter, they may suffer impaired swimming or foraging abilities, altered migratory behavior, altered breeding or reproductive patterns, and latent mortality from their interactions. Depending on the length of time an animal is out of water, and how they are handled, will likely affect the individual's chance of survival.

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 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 ASLL fishery. This allowed us to identify five potential subpopulations that may be impacted by this fishery (Figure 21). To assess the potential effect of the 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 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 *Risk* section, we are reasonably certain our subpopulation sizes are larger than 1,000 individuals. We did so because, as previously described in the *Status of the Listed Resources*, the current understanding of effective population sizes necessary for the genetic diversity needed to maintain evolutionary fitness in isolated populations such as ours is greater than 1,000 (Frankham et al. 2014). 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.

In Scenario 1, for the American Samoa subpopulation under the proposed action, we found that if all anticipated captures from the fishery (based on the 5-yr running average) are attributed to this subpopulation, populations smaller than 100 individuals would decrease, while larger populations would continue to increase across all timeframes for $r_{max} = 0.019$ (Table 26). For the other four subpopulations potentially impacted by the fishery, all continue to increase at all initial population abundances (Table 26). If $r_{max} = 0.042$, all combinations of subpopulations and initial abundances had increasing population trajectories with the exception of 60 individuals for American Samoa.

At the 95th percentile (Scenario 2), the American Samoa subpopulation would decrease if the initial abundance is less than 400 ($r_{max} = 0.019$) or 200 ($r_{max} = 0.042$) individuals (Table 26), otherwise it would increase. The Cook Islands and Kiribati – Phoenix Islands West subpopulations would decline if initial abundance was less than 100 and $r_{max} = 0.019$, at larger abundances we found increasing population trajectories and all initial abundances had increasing population trajectories if $r_{max} = 0.042$ (Table 26). The remaining subpopulations had increasing population trajectories at all initial abundances and values of r_{max} (Table 26).

In total, we looked at 200 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 five subpopulations assuming a minimum initial size of 200, we do not find that the proposed action causes material changes having biological consequences to the species' numbers, reproduction, or distribution. In accordance with Section 1.3.1 (*Jeopardy Analyses*) above, NMFS does not anticipate the ASLL 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 necessarily concerning 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.

6 CONCLUSION

After reviewing their current status, 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 of the ASLL 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 ASLL fishery in Table 30.

.

Table 30. The anticipated mean annual captures and mortalities from the ASLL fishery. These are the estimated number of incidental captures, expanded from observed number of captures. For giant manta ray we include values prorated for unidentified individuals in parentheses, we could not calculate those values for the mean and 95th percentile values for oceanic whitetip sharks because McCracken (2019a) did not provide mean and 95th percentile values for unidentified sharks.

Species	**Annual**	
	Mean Number Captured	**Mean Number Killed**
Oceanic whitetip shark [a]	620	322
Giant manta ray	9 (11)	3 (3)

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 Section 7.5 "*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 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 (Table 31).

Table 31. Reinitiation triggers for the ASLL fishery. These are the estimated number of captures, expanded from observed number of incidental captures and inclusive of prorated numbers based on unidentified captures. Exceeding the maximum 5-year running sum over any five consecutive years is a reinitiation trigger.

Species	**Maximum 5-Year Running Sum**
Oceanic whitetip shark [a]	3,520
Giant manta ray	57

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 PRD 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 ASLL fishery, as described in the proposed action, on threatened oceanic whitetip sharks and giant manta rays and to monitor the level and nature of any incidental takes. These measures are non-discretionary.

1. NMFS shall require that oceanic whitetip sharks and giant manta rays 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 post-release survivorship.

2. NMFS shall ensure that the ASLL 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 shall undertake and comply with the following terms and conditions to implement the reasonable and prudent measures identified in Section 7.3 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 oceanic whitetip sharks and giant manta rays for each interaction by species, gear, and set information, as well as the presence or absence of tags on these species.
    b. 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.
    c. NMFS' SFD shall provide an annual report to the NMFS' PRD that details the results of its monitoring of bycatch during each fishing season. These reports shall be submitted in writing within one month after data is finalized and will summarize all statistical information based on the previous fishing year (January 1 through December 31).

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 SFD should continue to research modifications to fishing gear (e.g., hook size, hook shape, hook offset, hook appendage, bait type, line type, depth configuration, float configuration, deterrents, decoys, etc.) and ESA-listed species handling methods (dehookers, lifting methods, etc.) to reduce ESA-listed species bycatch and mortality in commercial longline fisheries.
2. NMFS SFD should develop and implement a tagging program to examine the genetic profile of giant manta rays captured in the ASLL fishery to better inform management and recovery goals for these species. This should explore site use and residency patterns of giant manta rays released alive in the fishery and to examine post release mortality metrics pertinent to this specific fishery. This data would help to clarify what ray species is being captured (*M. alfredi* vs. *M. birostris*), and where these species are being captured in the fishery in order to avoid these areas by reducing potential interactions in the future.
3. NMFS should encourage RFMOs to require reporting of oceanic whitetip catches and discards, and for Parties to increase reporting of oceanic whitetip catch and disposition to improve data quality and quantify the impact of fishing on the species.
4. NMFS should enhance bilateral cooperation and engagement with key countries that have large international longline fleets to promote conservation and recovery of oceanic whitetip sharks and giant manta rays considered herein.

In order to keep NMFS' PRD informed of actions minimizing or avoiding adverse effects or benefitting listed species or their habitats, SFD should notify the PRD in writing upon initiating any of these conservation recommendations in their final action.

7.5 Reinitiation of Consultation Notice

This concludes formal consultation on the operation of the ASLL fishery as currently managed. 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

Acebes, J. M., and M. Tull. 2016. The History and Characteristics of the Mobulid Ray Fishery in the Bohol Sea, Philippines. PLoS One. 11(8):e0161444.

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

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

Adams, J., M. Sweezey, and P. V. Hodson. 2014. Oil and oil dispersant do not cause synergistic toxicity to fish embryos. Environmental Toxicology and Chemistry. 33(1):107-114.

Agamy, E. 2013a. Impact of laboratory exposure to light Arabian crude oil, dispersed oil and dispersant on the gills of the juvenile brown spotted grouper (*Epinephelus chlorostigma*): a histopathological study. Marine Environment Research. 86:46-55.

Agamy, E. 2013b. Sub chronic exposure to crude oil, dispersed oil and dispersant induces histopathological alterations in the gills of the juvenile rabbit fish (*Siganus canaliculatus*). Ecotoxicology and Environmental Safety. 92:180-190.

Alava, M. N. R., E. R. Z. Dolumbalo, A. A. Yaptinchay, and R. B. Trono. 2002. Fishery and trade of whale sharks and manta rays in the Bohol Sea, Philippines. Elasmobranch biodiversity, conservation and management: Proceedings of the international seminar and workshop Sabah, Malaysia; 1997. p. 132-148.

Allen, C. D., G. E. Lemons, T. Eguchi, R. A. LeRoux, C. C. Fahy, P. H. Dutton, S. H. Peckham, and J. A. Seminoff. 2013. Stable isotope analysis reveals migratory origin of loggerhead turtles in the Southern California Bight. Marine Ecology Progress Series. 472:275-285.

Anderson, R. C., A. M. Shiham, A. M. Kitchen-Wheeler, and G. Stevens. 2011. Extent and economic value of manta ray watching in Maldives. Tourism in Marine Environments. 7(1):15-27. 58.

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

Balcıoglu, E. B. 2016. Potential effects of polycyclic aromatic hydrocarbons (PAHs) in marine foods on human health: a critical review. Toxin Reviews. 35(3-4):98-105.

Barnes, R. H. 2005. Indigenous Use and Management of Whales and Other Marine Resources in East Flores and Lembata, Indonesia. Senri Ethnological Studies. 67:77-85.

Barraza, A. D., L. M. Komoroske, C. D. Allen, T. Eguchi, R. Gossett, E. Holland, D. D. Lawson, R. A. LeRoux, V. Lorenzi, and J. Seminoff. 2020. Persistent organic pollutants in green sea turtles (*Chelonia mydas*) inhabiting two urbanized Southern California habitats. 153:110979.

Baum, J., E. Medina, J. A. Musick, and M. Smale. 2015. *Carcharhinus longimanus,* Oceanic whitetip shark. The IUCN Red List of Threatened Species 2015: e.T39374A85699641.

(Downloaded on 30 June 2018) http://dx.doi.org/10.2305/IUCN.UK.2015.RLTS.T39374A85699641.en.

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

Beale, C.S., J.D. Stewart, E. Setyawan, A.B. Sianipar, and M.V. Erdmann. 2019. Population dynamics of oceanic manta rays (*Mobula birostris*) in the Raja Ampat Archipelago, West Papua, Indonesia, and the impacts of the El Niño–Southern Oscillation on their movement ecology. Wiley, Diversity and Distributions. 00:1-16.

Beerkircher, L. R., E. Cortes, and M. Shivji. 2008. Case study: Elasmobranch bycatch in the pelagic longline fishery off the southeastern United States, 1992–1997. In: Camhi, M., E. K. Pikitch, E. A. Babcock, editors. Sharks of the Open Ocean: Biology, Fisheries Conservation. p. 242-246. Blackwell Publishing Ltd.

Bechmann, R. K., B. K. Larsen, I. C. Taban, L. I. Hellgren, P. Møller, and S. Sanni. 2010. Chronic exposure of adults and embryos of *Pandalus borealis* to oil causes PAH accumulation, initiation of biomarker responses and an increase in larval mortality. Marine Pollution Bulletin. 60(11):2087-2098.

Bembenek-Bailey, S. A., J. N. Niemuth, P. D. McClellan-Green, M. H. Godfrey, C. A. Harms, H. Gracz, and M. K. Stoskopf. 2019. NMR Metabolomic Analysis of Skeletal Muscle, Heart, and Liver of Hatchling Loggerhead Sea Turtles (*Caretta caretta*) Experimentally Exposed to Crude Oil and/or Corexit. Metabolites. 9(2).

Bigelow, K., J. Rice, and F. Carvalho. 2022. Future Stock Projections of Oceanic Whitetip Sharks in the Western and Central Pacific Ocean (Update on Project 101). Scientific Committeee Eighteenth Regular Session. WCPFC-SC18-2022/EB-WP-02. 19 p.

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.

Bonfil, R., S. Clarke, H. Nakano, M. D. Camhi, E. K. Pikitch, and E. A. Babcock. 2008. The biology and ecology of the oceanic whitetip shark, *Carcharhinus longimanus*. Sharks of the open ocean: Biology, Fisheries and Conservation. 128-139.

Brainard, R. E., C. Birkeland, C. M. Eakin, P. McElhany, M. W. Miller, M. Patterson, and G. A. Piniak. 2011. Status review report of 82 candidate coral species petitioned under the U.S. Endangered Species Act. Pacific Islands Fisheries Science Center. 579 p.

Braun, C. D., G. B. Skomal, S. R. Thorrold, and M. L. Berumen. 2015. Movements of the reef manta ray (*Manta alfredi*) in the Red Sea using satellite and acoustic telemetry. Marine Biology. 162(12):2351-2362.

Brodziak, J., W. A. Walsh, and R. Hilborn. 2013. Model selection and multimodel inference for standardizing catch rates of bycatch species: a case study of oceanic whitetip shark in the Hawaii-based longline fishery. Canadian Journal of Fisheries and Aquatic Sciences. 70(12):1723-1740.

Burgess, K. B. 2017. Feeding ecology and habitat use of the giant manta ray *Manta birostris* at a key aggregation site off mainland Ecuador. The University of Queensland, Queensland. p. 174.

Camacho, M., L. D. Boada, J. Oros, P. Calabuig, M. Zumbado, and O. P. Luzardo. 2012. Comparative study of polycyclic aromatic hydrocarbons (PAHs) in plasma of Eastern Atlantic juvenile and adult nesting loggerhead sea turtles (*Caretta caretta*). Marine Pollution Bulletin. 64(9):1974-1980.

Camargo, S. M., R. Coelho, D. Chapman, L. Howey-Jordan, E. J. Brooks, D. Fernando, N. J. Mendes, F. H. Hazin, C. Oliveira, M. N. Santos et al. 2016. Structure and Genetic Variability of the Oceanic Whitetip Shark, *Carcharhinus longimanus*, Determined Using Mitochondrial DNA. PLoS One. 11(5):e0155623.

Campana, S. E., W. Joyce, and M. J. Manning. 2009. Bycatch and discard mortality in commercially caught blue sharks *Prionace glauca* assessed using archival satellite pop-up tags. Marine Ecology Progress Series. 387:241-253.

Carpenter, K. E., and V. H. Niem. 2001. FAO species identification guide for fishery purposes. The living marine resources of the Western Central Pacific Volume. 5. Bony fishes part 3 (Menidae to Pomacentridae).

Carey, J. and D. Roach. 2020. Biodemography: An Introduction to Concepts and Methods. Princeton University Press; Illustrated edition.

Casper, B. M., M. B. Halvorsen, and A. N. Popper. 2012. Are sharks even bothered by a noisy environment? The Effects of Noise on Aquatic Life. p. 93-97. Springer.

Casper, B. M., P. S. Lobel, and H. Y. Yan. 2003. The hearing sensitivity of the little skate, *Raja erinacea*: A comparison of two methods. Environmental Biology of Fishes. 68(4):371-379.

Casper, B. M., and D. A. Mann. 2006. Evoked potential audiograms of the nurse shark (*Ginglymostoma cirratum*) and the yellow stingray (*Urobatis jamaicensis*).Environmental Biology of Fishes. 76(1):101-108.

Casper, B. M., and D. A. Mann. 2009. Field hearing measurements of the Atlantic sharpnose shark *Rhizoprionodon terraenovae*. Journal of fish biology.75(10):2768-2776.

Chanrachkij, I., and A. Loog-on. 2003. Preliminary report on ghost fishing phenomena by drifting FADs in eastern Indian Ocean. Southeast Asian Fisheries Development Center Training Department. 22 p.

Chin, A., P. M. Kyne, T. I. Walker, and R. B. McAuley. 2010. An integrated risk assessment for climate change: analyzing 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, T. B. 2010. Abundance, home range, and movement patterns of manta rays (*Manta alfredi*, *M. birostris*) in Hawaii. University of Hawaii at Manoa.

Clarke, S., S. Harley, S. Hoyle, and J. Rice. 2011a. An indicator-based analysis of key shark species based on data held by SPC-OFP. WCPFC-SC7-2011/EB-WP-01. 88 p.

Clarke, S., L. Manarangi-Trott, and T. A. 2016. Data available to the Commission to address the implementation and effectiveness of CMM 2010-07 regarding shark finning.

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

Clarke, S. C., S. J. Harley, S. D. Hoyle, and J. S. Rice. 2012. Population trends in Pacific Oceanic sharks and the utility of regulations on shark finning. Conservation Biology. 27(1):197-209.

Clarke, S. C., M. K. McAllister, E. J. Milner-Gulland, G. P. Kirkwood, C. G. Michielsens, D. J. Agnew, E. K. Pikitch, H. Nakano, and M. S. Shivji. 2006. Global estimates of shark catches using trade records from commercial markets. Ecology Letters. 9(10):1115-1126.

CMS. 2014. Proposal for the inclusion of the reef manta ray (*Manta alfredi*) in CMS Appendix I and II. 18th Meeting of the Scientific Council, UNEP/CMS/ScC18/Doc.7.2.9. 17 p.

Coelho, R., F. H. V. Hazin, M. Rego, M. Tambourgi, P. Oliveira, P. Travassos, F. Carvalho, and G. Burgess. 2009. Notes on the reproduction of the oceanic whitetip shark, *Carcharhinus longimanus,* in the southwestern Equatorial Atlantic Ocean. Collective Volume of Scientific Papers ICCAT. 64(5):1734-1740.

Collins, J. F., J. P. Brown, G. V. Alexeeff, and A. G. Salmon. 1998. Potency equivalency factors for some polycyclic aromatic hydrocarbons and polycyclic aromatic hydrocarbon derivatives. Regulatory Toxicology Pharmacology. 28(1):45-54.

Compagno, L. J. V. 1984. FAO species catalogue Vol. 4, part 2 sharks of the world: An annotated and illustrated catalogue of shark species known to date. Food and Agriculture Organization of the United Nations.

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

Cortes, E., C. A. Brown, and L. Beerkircher. 2007. Relative abundance of pelagic sharks in the western North Atlantic Ocean, including the Gulf of Mexico and Caribbean Sea. Gulf and Caribbean Research. 19(2):37-52.

Couturier, L. I., F. R. Jaine, and T. Kashiwagi. 2015. First photographic records of the giant manta ray *Manta birostris* off eastern Australia. PeerJ. 3:e742.

Couturier, L. I. E., A. D. Marshall, F. R. A. Jaine, T. Kashiwagi, S. J. Pierce, K. A. Townsend, S. J. Weeks, M. B. Bennett, and A. J. Richardson. 2012. Biology, ecology and conservation of the Mobulidae. Journal of Fish Biology. 80(5):1075- 1119.

Craig, P., B. Ponwith, F. Aitaoto, and D. Hamm. 2013. The Commercial, Subsistence, and Recreational Fisheries of American Samoa. Marine Fisheries Review. 22(2):109-113.

Croll, D. A., H. Dewar, N. K. Dulvy, D. Fernando, M. P. Francis, F. Galvan-Magana, M. Hall, S. Heinrichs, A. D. Marshall, D. McCauley 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.

D. Hawaii. 2008. Ocean Mammal Institute v. Gates, 546 F. Supp. 2d 960. District Court, D. Hawaii. 13 p.

Danion, M., S. Le Floch, F. Lamour, J. Guyomarch, C. J. E. Quentel, and E. Safety. 2011. Bioconcentration and immunotoxicity of an experimental oil spill in European sea bass (*Dicentrarchus labrax L*.). 74(8):2167-2174.

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

De Andres, E., B. Gomara, D. Gonzalez-Paredes, J. Ruiz-Martin, and A. Marco. 2016. Persistent organic pollutant levels in eggs of leatherback turtles (*Dermochelys coriacea*) point to a decrease in hatching success. Chemosphere. 146:354-361.

Dewar, H. 2002. Preliminary report: Manta harvest in Lamakera. Oceanside, California. Report from the Pfleger Institute of Environmental Research and the Nature Conservancy. 3 p.

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

Domingo, A., M. Pons, S. Jimenez, P. Miller, C. Barcelo, and Y. Swimmer. 2012. Circle Hook Performance in the Uruguayan Pelagic Longline Fishery. Bulletin of Marine Science. 88(3):499-511. Doney, S. C., M. Ruckelshaus, J. E. Duffy, J. P. Barry, F. Chan, C. A. English, H. M. Galindo, J. M. Grebmeier, A. B. Hollowed, N. Knowlton et al. 2012. Climate change impacts on marine ecosystems. Annual Review of Marine Science. 4:11-37.

Duffy, C. A. J., and D. Abbott. 2003. Sightings of mobulid rays from northern New Zealand, with confirmation of the occurrence of *Manta birostris* in New Zealand waters. New Zealand Journal of Marine and Freshwater Research. 37(4):715-721.

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

Duronslet, M. J., C. Caillouet Jr, S. Manzella, W. K. W. Indelicato, C. T. Fontaine, B. D. Revera, T. Williams, and D. Boss. 1986. The effects of an underwater explosion on the sea turtles *Lepidochelys kempi* and *Caretta* with observations of effects on other marine organisms. NMFS Biological Laboratory, Galveston, Texas. p. 17.

Erdmann, M. 2014. New MMAF-CI-SEAA manta tagging program launched by Mark Erdmann.

Ehsanpour, M., M. Afkhami, R. Khoshnood, K. J. J. B. o. e. c. Reich, and toxicology. 2014. Determination and maternal transfer of heavy metals (Cd, Cu, Zn, Pb and Hg) in the Hawksbill sea turtle (*Eretmochelys imbricata*) from a nesting colony of Qeshm Island, Iran. 92(6):667-673.

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. 169 p.

Fernando, D., and G. Stevens. 2011. A study of Sri Lanka's manta and mobula ray fishery. The Manta Trust.

Fetherston, T., S. Turner, G. Mitchell, and E. Guzas. 2019. Marine Mammal Lung Dynamics when Exposed to Underwater Explosion Impulse. The Anatomical Record. 302(5):718-734.

Filmalter, J. D., M. Capello, J. L. Deneubourg, P. D. Cowley, and L. Dagorn. 2013. Looking behind the curtain: quantifying massive shark mortality in fish aggregating devices. Frontiers in Ecology and the Environment. 11(6):291-296.

Ford, A. 2009. Modeling the Environment. Second Edition. Island Press. p. 380.

Fossi, M. C., M. Baini, C. Panti, M. Galli, B. Jimenez, J. Munoz-Arnanz, L. Marsili, M. G. Finoia, and D. Ramirez-Macias. 2017. Are whale sharks exposed to persistent organic pollutants and plastic pollution in the Gulf of California (Mexico)? First ecotoxicological investigation using skin biopsies. Comparative Biochemistry and Physiology C-Toxicology & Pharmacology. 199:48-58.

Fossi, M. C., D. Coppola, M. Baini, M. Giannetti, C. Guerranti, L. Marsili, C. Panti, E. de Sabata, and S. Clo. 2014. Large filter feeding marine organisms as indicators of microplastic in the pelagic environment: the case studies of the Mediterranean basking shark (*Cetorhinus maximus*) and fin whale (*Balaenoptera physalus*). Marine Environmental Research. 100:17-24.

Fossi, M. C., C. Panti, C. Guerranti, D. Coppola, M. Giannetti, L. Marsili, 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.

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.

Gennip, S. J. V., E. E. Popova, A. Yool, G. T. Pecl, A. J. Hobday, and C. J. B. Sorte. 2017. Going with the flow: the role of ocean circulation in global marine ecosystems under a changing climate. Global Change Biology. 23(7):2602-2617.

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

Germanov, E. S. 2015b. Microplastics & Megafauna. Available at: https://elitzagermanov.wordpress.com/

Germanov, E. S., and A. D. Marshall. 2014. Running the gauntlet: regional movement patterns of *Manta alfredi* through a complex of parks and fisheries. PLoS One. 9(10):e110071.

Germanov, E. S., A. D. Marshall, L. Bejder, M. C. Fossi, and N. R. Loneragan. 2018. Microplastics: No small problem for filter-feeding megafauna. Trends in Ecology & Evolution. 33(4):227-232.Gilman, E., D. Kobayashi, T. Swenarton, N. Brothers, P. Dalzell, and I. Kinan-Kelly. 2007. Reducing sea turtle interactions in the Hawaii-based longline swordfish fishery. Biological Conservation. 139(1-2):19-28.

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

Govoni, J. J., M. A. West, L. R. Settle, R. T. Lynch, and M. D. Greene. 2008. Effects of underwater explosions on larval fish: Implications for a coastal engineering project. Journal of Coastal Research. 24(2b):228-233.

Graham, R. T., M. J. Witt, D. W. Castellanos, F. Remolina, S. Maxwell, B. J. Godley, and L. A. Hawkes. 2012. Satellite tracking of manta rays highlights challenges to their conservation. PLoS One. 7(5):e36834.

Guinder, V., and J. C. Molinero. 2013. Climate Change Effects on Marine Phytoplankton. Boca Raton: CRC Press. p. 68-90.

Hall, M., and M. Roman. 2013. Bycatch and non-tuna catch in the tropical tuna purse seine fisheries of the world. Rome, Italy. No. 978-92-5-107241-7. 262 p.

Hare, S. R., S. J. Harley, and W. J. Hampton. 2015. Verifying FAD-association in purse seine catches on the basis of catch sampling. Fisheries Research. 172:361-372.

Hasarangi, D. G. N., R. Maldeniya, and S. S. K. Haputhantri. 2012. A Review on shark fishery resources in Sri Lanka. IOTC–2012–WPEB08–15 Rev_1. 15 p.

Hastings, M. C., and A. N. Popper. 2005. Effects of sound on fish. California Department of Transportation.

Hazen, E. L., S. J. Jorgensen, R. R. Rykaczewski, S. J. Bograd, D. G. Foley, I. D. Jonsen, S. A. Shaffer, J. P. Dunne, D. P. Costa, L. B. Crowder et al. 2012. Predicted habitat shifts of Pacific top predators in a changing climate. Nature Climate Change. 3(3):234-238.

Hazin, F. H., H. G. Hazin, and P. Travassos. 2007. CPUE and catch trends of shark species caught by Brazilian longliners in the Southwestern Atlantic Ocean. Collective Volume of Scientific Papers ICCAT. 60(2):636-647.

Hearn, A. R., D. Acuna, J. T. Ketchum, C. Penaherrera, J. Green, A. Marshall, M. Guerrero, and G. Shillinger. 2014. Elasmobranchs of the Galapagos marine reserve. p. 23-59. The Galapagos Marine Reserve. Springer.

Heinrichs, S., M. O'Malley, H. Medd, and P. Hilton. 2011. Manta Ray of Hope: Global Threat to Manta and Mobula Rays. Manta Ray of Hope Project.

Hildebrand, J. A. 2004. Impacts of anthropogenic sound on cetaceans. Unpublished paper submitted to the International Whaling Commission Scientific Committee SC/56E. 13 p.

Hinojosa-Alvarez S, Walter RP, Diaz-Jaimes P, Galván-Magaña F, Paig-Tran EM. 2016. A potential third manta ray species near the Yucatán Peninsula? Evidence for a recently diverged and novel genetic *Manta* group from the Gulf of Mexico. PeerJ 4:e2586

Hoenig, J. M., and S. H. Gruber. 1990. Life-History Patterns in the Elasmobranchs: Implications for Fisheries Management. In: Pratt Jr., H. L., S. H. Gruber, T. Taniuch, editors. Elasmobranchs as Living Resources: Advances in the Biology, Ecology, Systematics, and the Status of the Fisheries. NOAA Technical Report NMFS 90. 528 p.

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

Howey-Jordan, L. A., E. J. Brooks, D. L. Abercrombie, L. K. Jordan, A. Brooks, S. Williams, E. Gospodarczyk, and D. D. Chapman. 2013. Complex movements, philopatry and expanded depth range of a severely threatened pelagic shark, the oceanic whitetip (*Carcharhinus longimanus*) in the western North Atlantic. PLoS One. 8(2):e56588.

Howey, L. A., E. R. Tolentino, Y. P. Papastamatiou, E. J. Brooks, D. L. Abercrombie, Y.Y. Watanabe, S. Williams, A. Brooks, D. D. Chapman, and L. K. B. Jordan. 2016. Into the deep: the functionality of mesopelagic excursions by an oceanic apex predator. Ecology and Evolution. 6(15):5290-5304.

Hussar, E., S. Richards, Z.-Q. Lin, R. P. Dixon, K. A. J. W. Johnson, Air,, and S. Pollution. 2012. Human health risk assessment of 16 priority polycyclic aromatic hydrocarbons in soils of Chattanooga, Tennessee, USA. 223(9):5535-5548.

Hutchinson, M. R., D. G. Itano, J. A. Muir, and K. M. Holland. 2015. Post-release survival of juvenile silky sharks captured in a tropical tuna purse seine fishery. Marine Ecology Progress Series. 521:143-154.

Hutchinson, J., and M. Simmonds. 1992. Escalation of threats on marine turtles. Oryx. 26(2):95-102.

Hutchinson, M., F. Poisson, and Y. Swimmer. 2017. Developing best handling practice guidelines to safely release mantas and mobulids captured in commercial fisheries. 13th Regular Session of the Scientific Committee. 9–17 August 2017, Rarotonga, Cook Islands. Information papers. WCPFC-SC13-2017/SA-IP-08.

Hutchinson, M., and K. Bigelow. 2019. Quantifying Post Release Mortality Rates of Sharks Incidentally Captured in Pacific Tuna Longline Fisheries and Identifying Handling Practices to Improve Survivorship. Scientific Committee Fifteenth Regular Session. Pohnpei, Federated States of Micronesia. WCPFC-SC15-2019/EB-WP-04 (Rev.01). 26 p.

Hutchinson, M., Z. Siders, J. Stahl, and K. Bigelow. 2021. Quantitative estimates of post-release survival rates of sharks captured in Pacific tuna longline fisheries reveal handling and discard practices that improve survivorship. PIFSC Data Report DR-21-001. 61 p.

Incardona, J. P., T. K. Collier, and N. L. Scholz. 2004. Defects in cardiac function precede morphological abnormalities in fish embryos exposed to polycyclic aromatic hydrocarbons. Toxicology Applied Pharmacology. 196(2):191-205.

IOTC (Indian Ocean Tuna Commission). 2011. Report of the Fourteenth Session of the IOTC Scientific Committee. IOTC–2011–SC14–R[E]. 259 p.

IOTC. 2015. Status of the Indian Ocean oceanic whitetip shark (OCS: *Carcharhinus longimanus*). IOTC–2015–SC18–ES18[E]. 7 p.

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. 151 p.

IPCC. 2018. Summary for Policymakers. In: Masson-Delmotte, V., P. Zhai, H.-O. Portner, D. Roberts, J. Skea, P. R. Shukla, A. Pirani, W. Moufouma-Okia, C. Pean, R. Pidcock 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 p.

Jaine, F.R., Couturier, L.I., Weeks, S.J., Townsend, K.A., Bennett, M.B., Fiora, K., and A. J. Richardson. (2014) When giants turn up: sighting trends, environmental influences and habitat use of the manta ray *Manta alfredi* at a coral reef. PLoS One 7: e46170

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

Jones, T. T., S. Martin, T. Eguchi, B. Langseth, J. Baker, and A. Yau. 2018. Review of draft response to PRD's request for information to support ESA section 7 consultation on the effects of Hawaii-based longline fisheries on ESA listed species. NMFS Pacific Islands Fisheries Science Center, Honolulu, HI. 35 p.

Joung, S. J., N. F. Chen, H. H. Hsu, and K. M. Liu. 2016. Estimates of life history parameters of the oceanic whitetip shark, *Carcharhinus longimanus*, in the Western North Pacific Ocean. Marine Biology Research. 12(7):758-768.

Karl, T. R., J. M. Melillo, T. C. Peterson, and S. J. Hassol. 2009. Global climate change impacts in the United States. Cambridge University Press.

Kashiwagi, T., T. Ito, and F. Sato. 2010. Occurrences 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.

Kleiber, D., and K. Leong. 2018. Cultural Fishing in American Samoa. Pacific Islands Fisheries Science Center, PIFSC Administrative Report H-18-03. 21 p.

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*, 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.

Ladich, F., and R. R. Fay. 2013. Auditory evoked potential audiometry in fish. Reviews in Fish Biology and Fisheries. 23(3):317-364.

Last, P. R., and J. D. Stevens. 2009. Sharks and rays of Australia. CSIRO Publishing. p.656.

Lawson, J. M., S. V. Fordham, M. P. O'Malley, L. N. Davidson, R. H. Walls, M. R. Heupel, G. Stevens, D. Fernando, A. Budziak, and C. A. Simpfendorfer. 2017. Sympathy for the devil: a conservation strategy for devil and manta rays. PeerJ. 5:e3027.

Lawson, T. 2011. Estimation of Catch Rates and Catches of Key Shark Species in Tuna Fisheries of the Western and Central Pacific Ocean Using Observer Data. Information Paper EB IP–02. Seventh Regular Session of the Scientific Committee of the WCPFC. Pohnpei, FSM. 9th–17th August. 52 p.

Lessa, R., F. M. Santana, and R. Paglerani. 1999a. Age, growth and stock structure of the oceanic whitetip shark, *Carcharhinus longimanus*, from the southwestern equatorial Atlantic. Fisheries Research. 42(1-2):21-30.

Lessa, R., R. Paglerani, and F. Santana. 1999b. Biology and morphometry of the oceanic whitetip shark, *Carcharhinus longimanus* (Carcharhinidae), off North-Eastern Brazil. Cybium: international journal of ichthyology. 23(4):353-368.

Lewis, S. A., N. Setiasih, F. Fahmi, D. Dharmadi, M. P. O'Malley, S. J. Campbell, M. Yusuf, and A. B. Sianipar. 2015. Assessing Indonesian manta and devil ray populations through historical landings and fishing community interviews. PeerJ PrePrints.

Liu, K.-M., and W.-P. Tsai. 2011. Catch and life history parameters of pelagic sharks in the Northwestern Pacific. Keelung, Chinese Taipei, ISC Shark Working Group Workshop. 12 p.

Lomolino, M. V., and R. Channel. 1998. Range collapse, re-introductions, and biogeographic guidelines for conservation. Conservation Biology. 12(2):481-484.

Lomolino, M. V., and R. Channell. 1995. Splendid isolation: patterns of geographic range collapse in endangered mammals. Journal of Mammalogy. 76(2):335-347.

Loomis, D. K., M. E. Allen, and C. Hawkins. 2019. American Samoa Fishing Community Perceptions of the Marine Protected Area Siting Process and its Implications. Western Pacific Regional Fishery Management Council Pacific Island Fisheries Research Program. 72 p.

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

Lutcavage, M. E., P. L. Lutz, G. D. Bossart, and D. M. Hudson. 1995. Physiologic and clinicopathologic effects of crude oil on loggerhead sea turtles. Archives of Environmental Contamination Toxicology Applied Pharmacology. 28(4):417-422.

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

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

Madigan, D. J., E. J. Brooks, M. E. Bond, J. Gelsleichter, L. A. Howey, D. L. Abercrombie, A. Brooks, and D. D. Chapman. 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. [Accessed April 3, 2019] Available at: https://www.facebook.com/mantamatcher/

Mantua, N. J., S. R. Hare, Y. Zhang, J. M. Wallace, and R. C. Francis. 1997. A Pacific Interdecadal Climate Oscillation with Impacts on Salmon Production. Bulletin of the American Meteorological Society. 78(6):1069-1080.

Marshall, A., M. B. Bennett, G. Kodja, S. Hinojosa-Alvarez, F. Galvan-Magana, M. Harding, G. Stevens, and T. Kashiwagi. 2011. *Mobula birostris*. http://www.iucnredlist.org/details/198921/0

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

Marshall, A., T. Kashiwagi, M. B. Bennett, M. Deakos, G. Stevens, F. McGregor, T. Clark, H. Ishihara, and K. Sato. 2018. *Mobula alfredi* (amended version of 2011 assessment). The IUCN Red List of Threatened Species 2018: e.T195459A126665723.

Marshall, A. D., and M. B. Bennett. 2010. Reproductive ecology of the reef manta ray *Manta alfredi* in southern Mozambique. Journal of fish biology. 77(1):169-190.

Marshall, A. D., L. J. V. Compagno, and M. B. Bennett. 2009. Redescription of the genus Manta with resurrection of *Manta alfredi* (Krefft, 1868)(Chondrichthyes; Myliobatoidei; Mobulidae). Zootaxa. 2301:1-28.

Martin, S. L., S. M. Stohs, and J. E. Moore. 2015. Bayesian inference and assessment for rare-event bycatch in marine fisheries: a drift gillnet fishery case study. Ecological Applications. 25(2):416-429.

Mastrangelo, G., E. Fadda, and V. Marzia. 1996. Polycyclic aromatic hydrocarbons and cancer in man. Environmental Health Perspectives. 104(11):1166-1170.

McCarty, J. P. 2001. Ecological consequences of recent climate change. Conservation Biology. 15(2):320-331.

McCauley, R. D., J. Fewtrell, A. J. Duncan, C. Jenner, M. N. Jenner, J. D. Penrose, R. I.T. Prince, A. Adhitya, J. Murdoch, and K. McCabe. 2000. Marine seismic surveys—a study of environmental implications. The APPEA Journal. 40(1):692- 708.

McCracken, M. L. 2019a. 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. 2019b. 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. 26 p.

McCracken, M. L. 2020. Data Report Estimation of Bycatch with Sea Turtles, Seabirds, Bony Fish, Sharks, and Rays in the American Samoa Permitted Longline Fishery for Years 2016-2019. PIFSC Data Report DR-20-021. Pacific Island Fisheries Science Center. Honolulu, HI. 1 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.

McCracken, M. L. and B. Cooper. 2022. Data Report Estimation of Bycatch with Sea Turtles, Seabirds, Bony Fish, Sharks, and Rays in the 2020 Permitted American Samoa Permitted Longline Fishery. PIFSC Data Report DR-22-001. Pacific Island Fisheries Science Center. Honolulu, HI. 2 p.

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

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

Miller, M. H., and C. Klimovich. 2017. Endangered Species Act Status Review Report: Giant Manta Ray (*Manta birostris*) and Reef Manta Ray (*Manta alfredi*). Final report to National Marine Fisheries Service, Office of Protected Resources, Silver Spring, MD. September 2017. 128 p.

Mitchelmore, C. L., C. A. Bishop, and T. K. Collier. 2017. Toxicological estimation of mortality of oceanic sea turtles oiled during the Deepwater Horizon oil spill. Endangered Species Research. 33:39-50.

MMC (Marine Mammal Commission). 2007. Marine Mammals and Noise: A Sound Approach to Research and Management. A Report to Congress from the Marine Mammal Commission. 370 p.

Moazzam, M. 2018. Unprecedented decline in the catches of mobulids: an important component of tuna gillnet fisheries of the Northern Arabian Sea. 1472–7. Molony, B. 2005. Estimates of the mortality of non-target species with an initial focus on seabirds, turtles and sharks. 1st Meeting of the Scientific Committee of the Western and Central Pacific Fisheries Commission. p. 8-19.

Munoz, C. C., A. J. Hendriks, A. M. J. Ragas, and P. Vermeiren. 2021. Internal and Maternal Distribution of Persistent Organic Pollutants in Sea Turtle Tissues: A Meta-Analysis. Environ Sci Technol. 55(14):10012-10024.

Munoz, C. C., and P. Vermeiren. 2020. Maternal Transfer of Persistent Organic Pollutants to Sea Turtle Eggs: A Meta-Analysis Addressing Knowledge and Data Gaps Toward an Improved Synthesis of Research Outputs. Environmental Toxicology Chemistry. 39(1):9-29.

Moore, M., D. Livingstone, and J. Widdows. 1989. Hydrocarbons in marine mollusks: biological effects and ecological consequences. CRC Press Inc.: Boca Raton, FL.

Mourier, J. 2012. Manta rays in the Marquesas Islands: first records of *Manta birostris* in French Polynesia and most easterly location of *Manta alfredi* in the Pacific Ocean, with notes on their distribution. Journal of fish biology. 81(6):2053-2058.

Moyes, C. D., N. Fragoso, M. K. Musyl, and R. W. Brill. 2006. Predicting postrelease survival in large pelagic fish. Transactions of the American Fisheries Society. 135(5):1389-1397.

Murray, K. T. 2011. Interactions between sea turtles and dredge gear in the U.S. sea scallop (*Placopecten magellanicus*) fishery, 2001–2008. Fisheries Research. 107(1-3):137-146.

Murua, J., G. Moreno, and V. Restrepo. 2017. Progress on the adoption of non-entangling drifting fish aggregating devices in tuna purse seine fleets. Collective Volume of Scientific Papers ICCAT. 73:958-973. Musyl, M. K., R. W. Brill, D. S. Curran, N. M. Fragoso, L. M. McNaughton, A. Nielsen,B. S. Kikkawa, and C. D. Moyes. 2011. Postrelease survival, vertical and horizontal movements, and thermal habitats of five species of pelagic sharks in the central Pacific Ocean. Fishery Bulletin. 109(4):341-368.

Myrberg, A. A. 2001. The acoustical biology of elasmobranchs. The behavior and sensory biology of elasmobranch fishes: an anthology in memory of Donald Richard Nelson. p. 31-46. Springer.

Nakano, H., M. Okazaki, and H. Okamota. 1997. Analysis of catch depth by species for tuna longline fishery based on catch by branch lines. Bulletin of the Natural Resources Institute, Far Seas Fishery. (34):43-62.

Newcombe, R. G. 1998. Two sided confidence intervals for the single proportion: comparison of seven methods. Statistics in medicine. 17(8):857-872.

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

NMFS. 2004b. Management measures to implement new technologies for the western Pacific pelagic longline fisheries. A regulatory amendment to the fishery management plan for the pelagic fisheries of the western Pacific region including a final supplemental environmental impact statement. 310 p.

NMFS. 2009. Endangered Species Act Section 7 Consultation Biological Opinion and Incidental Take Statement: Continued authorization of pelagic troll and handline fisheries, as managed under the Fishery Management Plan for Pelagic Fisheries of the Western Pacific Region. NMFS, Pacific Islands Region, Protected Resources Division. 31 p.

NMFS. 2010. Biological opinion on measures to reduce interactions between Green Sea Turtles and the American Samoa-based Longline Fishery-Implementation of an Amendment to the Fishery Ecosystem Plan for Pelagic Fisheries of the Western pacific Region on ESA-listed marine species. p. 91.

NMFS. 2015a. Biological and Conference Opinion on the continued operation of the American Samoa longline fishery. Honolulu, HI. 190 p.

NMFS. 2015b. Letter of Concurrence. Continued operation of the coral reef, bottom fish, crustacean, and precious coral fisherys under the Fisheries Ecosystem Plan for American Samoa. Pacific Island Regional Office. Protected Resources Division.4p.

NMFS. 2016. Revised guidance for treatment of climate change in NMFS Endangered Species Act decisions. U.S. Department of Commerce. p. 1-8.

NMFS. 2017a. American Samoa Longline Observer Program Field Manual. American Samoa Field Office, Pago Pago, AS. 155 p.

NMFS. 2017b. Biological Evaluation: Potential Impacts of the WCPO Purse Seine Fishery. National Marine Fisheries Service, Pacific Islands Region, Division of Protected Resources. Honolulu, HI. 212 p.

NMFS. 2017c. Biological Opinion on (1) U.S. Military Mariana Islands Training and Testing Activities; and (2) the NMFS's Promulgation of Regulations and Issuance of a Letter of Authorization Pursuant to the MMPA. Office of Protected Resources. Silver Spring, MD. 569 p.

NMFS. 2018a. Biological Evaluation: Potential Effects of the Hawaii Deep-set Longline Fishery on Endangered Species Act Listed Species and their Designated Critical Habitat. Honolulu, HI. 74 p.

NMFS. 2018b. Biological Opinion on U.S. Navy Hawaii-Southern California Training and Testing and the National Marine Fisheries Service's Promulgation of Regulations Pursuant to the Marine Mammal Protection Act for the Navy to "Take" Marine Mammals Incidental to Hawaii-Southern California Training and Testing. Office of Protected Resources. Silver Spring, MD. 683 p.

NMFS. 2019a. Biological Evaluation. Potential Effects of the American Samoa Pelagic Longline Fishery on Marine Species Listed under the Endangered Species Act. Honolulu, HI. 92 p.

NMFS. 2019b. Biological Opinion. Continued Authorization of the Hawaii Shallow-set Longline Fishery. Pacific Islands Regional Office. 506 p.

NMFS. 2019c. Purse Seine Analysis methods. Pacific Islands Regional Office. International Fisheries Division. 17 p.

NMFS. 2019d. Formal Consultation under the Environmental Standards for United States Army Kwajalein Atoll Activities in the Republic of the Marshall Islands Biological Opinion And Formal Consultation under Section 7 of the Endangered Species Act; Single Flight Experiment-2 (FE-2). Pacific Island Regional Office. 84 p.

NMFS. 2019e. Formal Consultation under the Environmental Standards for United States Army Kwajalein Atoll Activities in the Republic of the Marshall Islands Biological Opinion And Informal Consultation under Section 7 of the Endangered Species Act; Air-launched Rapid Response Weapon Flight Tests. Pacific Island Regional Office. 74 p.

NMFS. 2021. 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. 2022a. Endangered Species Act (ESA) Section 7(a)(2) Biological Opinion. Reinitiation of Endangered Species Act Section 7 consultation on the bottomfish fisheries of American Samoa, Guam, the Northern Mariana Islands, and the Main Hawaiian Islands as managed under the American Samoa, Mariana Archipelago, and Hawaii Archipelago Fishery Ecosystem Plans. NMFS Pacific Island Regional Office. Honolulu, Hawaii. 80 p.

NMFS. 2022b. Endangered Species Act Section 7(a)(2) Consultation Oceanic Whitetip Shark and Giant Manta Ray Supplemental Biological Opinion. Supplement to the Authorization of the Hawaii Deep-Set Longline Fishery; Effects to Oceanic Whitetip Sharks and Giant Manta Rays. NMFS Pacific Island Regional Office. Honolulu, Hawaii. 144 p.

NOAA (National Oceanic and Atmospheric Administration). 2022a. Vessels and Corals: A Tale in American Samoa. NOAA’s Office of Response and Restoration. [Accessed on October 13, 2022]. https://response.restoration.noaa.gov/vessels-and-corals-tale-american-samoa

NOAA. 2022b. Incident Responses for February 2018. NOAA’s Office of Response and Restoration. [Accessed on October 13, 2022]. https://blog.response.restoration.noaa.gov/incident-responses-february-2018

NOAA. 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 E. V. Hillyer. 1989. Mobulid rays off eastern Venezuela (Chondrichthyes, Mobulidae). Copeia.607-614.

NRC (National Research Council). 2003. Ocean Noise and Marine Mammals.Washington (DC): The National Academies Press.

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

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

Omar-Ali, A., C. Hohn, P. J. Allen, J. L. Rodriguez, and L. Petrie-Hanson. 2015. Tissue PAH, blood cell and tissue changes following exposure to water accommodated fractions of crude oil in alligator gar, *Atractosteus spatula*. Marine Environmental Research. 108:33-44.

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

Peatman, T., V. Allain, S. Caillot, P. Williams, and N. Smith. 2017. Summary of purse seine fishery bycatch at a regional scale, 2003-2016. Scientific Committee Thirteenth Regular Session Rarotonga, Cook Islands 9-17 August 2017. 74 p.

Peatman, T., L. Bell, V. Allain, P. Caillot, S. Williams, I. Tuiloma, A. Panizza, L. Tremblay-Boyer, S. Fukofuka, and N. Smith. 2018. Summary of longline fishery bycatch at a regional scale, 2003-2017 Rev 2 (22 July 2018). Busan, Republic of Korea 8-16 August 2018. 61 p.

Peatman, T., and S. Nicol. 2021. Updated Purse Seine Bycatch Estimates in the WCPO. Scientific Commitee Seventeenth Regular Session. Online 11 – 19 August 2021. WCPFC-SC17-2021/ST-IP-06. 41 p.

Perez, C., A. Velando, I. Munilla, M. López-Alonso, and D. J. E. s. Oro. 2008. Monitoring polycyclic aromatic hydrocarbon pollution in the marine environment after the Prestige oil spill by means of seabird blood analysis. Environmental Science Technology. 42(3):707-713.

Pilling, G. M., N. Smith, G. Moreno, C. van der Geest, V. Restrepo, and J. Hampton. 2017. Review of research into drifting FAD designs to reduce species of special interest bycatch entanglement and bigeye/yellowfin interactions. Scientific Committee Thirteenth Regular Session. Rarotonga, Cook Islands. WCPFC-SC13-2017/EB-WP-02. 16 p.

Pilling, G. M., N. Smith, G. Moreno, C. van der Geest, V. Restrepo, and J. Hampton. 2018. Review of research into drifting FAD designs to reduce species of special interest bycatch entanglement and bigeye/yellowfin interactions. Scientific Committee Thirteenth Regular Session. Rarotonga, Cook Islands. WCPFC-SC13-2017/EB-WP-02. 16 p.

Poeta, G., E. Staffieri, A. T. R. Acosta, and C. Battisti. 2017. Ecological effects of anthropogenic litter on marine mammals: A global review with a “black-list” of impacted taxa. Hystrix the Italian Journal of Mammalogy. 28(2):253–264.

Popper, A. N., and C. R. Schilt. 2008. Hearing and acoustic behavior: basic and applied considerations. p. 17-48. Fish Bioacoustics. Springer.

Portnoy, D. S., J. R. McDowell, E. J. Heist, J. A. Musick, and J. E. Graves. 2010. World phytogeography and male-mediated gene flow in the sandbar shark, *Carcharhinus plumbeus*. Molecular Ecology. 19(10):1994-2010.

Rice, J., and S. Harley. 2012. Stock assessment of silky 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., L. Tremblay-Boyer, R. Scott, S. Hare, and A. Tidd. 2015. Analysis of stock status and related indicators for key shark species of the Western Central Pacific Fisheries Commission. Paper presented at: 11th Regular Session of the Scientific Committee of the WCPFC. Pohnpei, Federated States of Micronesia.

Rice, 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.

Richardson, W. J., C. R. Greene Jr, C. I. Malme, and D. H. Thomson. 1995. Marine mammals and noise. Academic Press.

Rochman, C. M., T. Kurobe, I. Flores, and S. J. Teh. 2014. Early warning signs of endocrine disruption in adult fish from the ingestion of polyethylene with and without sorbed chemical pollutants from the marine environment. Science of the Total Environment. 493:656-661.

Rohner, C. A., S. J. Pierce, A. D. Marshall, S. J. Weeks, M. B. Bennett, and A. J. Richardson. 2013. Trends in sightings and environmental influences on a coastal aggregation of manta rays and whale sharks. Marine Ecology Progress Series.482:153-168.

Romo-Curiel, A. E., Z. Ramirez-Mendoza, A. Fajardo-Yamamoto, M. R. Ramirez-Leon, M. C. Garcia-Aguilar, S. Z. Herzka, P. Perez-Brunius, L. E. Saldana-Ruiz, J. Sheinbaum, K. Kotzakoulakis et al. 2022. Assessing the exposure risk of large pelagic fish to oil spills scenarios in the deep waters of the Gulf of Mexico. Marine Pollution Bulletin. 176:113434.

Ruck, C. L. 2016. Global genetic connectivity and diversity in a shark of high conservation concern, the oceanic whitetip, *Carcharhinus longimanus*. M. Sc. thesis. Nova Southeastern University. 64 p.

Ryder, C. E., T. A. Conant, and B. A. Schroeder. 2006. Report of the Workshop on Marine Turtle Longline Post-Interaction Mortality. NOAA Tech. Memo. NMFS- F/OPR-29. 36 p.

Sales, G., B. B. Giffoni, F. N. Fiedler, V. G. Azevedo, J. E. Kotas, Y. Swimmer, and L. Bugoni. 2010. Circle hook effectiveness for the mitigation of sea turtle bycatch and capture of target species in a Brazilian pelagic longline fishery. Aquatic Conservation-Marine and Freshwater Ecosystems. 20(4):428-436.

Sanchez, S. 2016. Improved Surveillance to Protect Ecuador’s Manta Populations. https://wildaid.org/improved-surveillance-to-protect-ecuadors-manta-populations/.

Santana, F. M., P. J. Duarte-Neto, and R. P. Lessa. 2004. *Carcharhinus longimanus*. In: Lessa, R. P., M. F. Nobrega, J. L. Bezerra Jr., editors. Dinsmica de Populacoes e Avaliacao de Estoques dos Recursos Pesqueiros da Regiao Nordeste. Vol II. Universidade Federal Rural de Pernambuco Deoartanebti de Pesca. Laboratorio de Dinâmica de Populacoes Marinhas - DIMAR.

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

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

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

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

Simmonds, M. P., and W. J. Eliott. 2009. Climate change and cetaceans: concerns and recent developments. Journal of the Marine Biological Association of the United Kingdom. 89(01).

Smallegange, I. M., I. B. C. van der Ouderaa, and Y. Tibirica. 2016. Effects of yearling, juvenile and adult survival on reef manta ray (*Manta alfredi*) demography. PeerJ. 4:e2370.

Smith, S. E., D. W. Au, and C. Show. 1998. Intrinsic rebound potentials of 26 species of Pacific sharks. Marine and Freshwater Research. 49(7):663-678.

Smith, S. E., D. W. Au, and C. Show. 2008. Intrinsic rates of increase in pelagic elasmobranchs. p. 288-297. In: Camhi, M. D., E. Pikitch, and E. A. Babcock, editors. Sharks of the Open Ocean: Biology, Fisheries Conservation. Blackwell Publishing.

Stevens, G., D. Fernando, and G. N. Di Sciara. 2018. Guide to the Manta and Devil Rays of the World. Princeton University Press. Stevens, J. D., and J. M. Lyle. 1989. Biology of Three Hammerhead Sharks (*Eusphyra blochii, Sphyrna mokarran* and *S. lewini*) from Northern Australia. Australian Journal of Marine and Freshwater Research. 40:129-146.

Stewart, J. D., C. S. Beale, D. Fernando, A. B. Sianipar, R. S. Burton, B. X. Semmens, and O. Aburto-Oropeza. 2016a. Spatial ecology and conservation of *Manta birostris* in the Indo-Pacific. Biological Conservation. 200:178-183.

Stewart, J. D., E. M. Hoyos-Padilla, K. R. Kumli, and R. D. Rubin. 2016b. Deep-water feeding and behavioral plasticity in *Manta birostris* revealed by archival tags and submersible observations. Zoology. 119(5):406-413.

Stewart, J. D., M. Nuttall, E. L. Hickerson, and M. A. Johnston. 2018. Important juvenile manta ray habitat at Flower Garden Banks National Marine Sanctuary in the northwestern Gulf of Mexico. Marine Biology. 165(7).

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

Swimmer, Y., and E. Gilman. 2012. Report of the Sea Turtle Longline Fishery Post- release Mortality Workshop, November 15−16, 2011. NOAA Tech. Memo. NOAA-TM-NMFS-PIFSC-34. 31 p.

Swingle, W. M., S. G. Barco, T. D. Pitchford, W. A. Mclellan, and D. A. Pabst. 1993. Appearance of Juvenile Humpback Whales Feeding in the Nearshore Waters of Virginia. Marine Mammal Science. 9(3):309-315.

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

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

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

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

Tremblay-Boyer, L., and S. Brouwer. 2016. Review of available information on non-key shark species including mobulids and fisheries interactions. Western and Central Pacific Fisheries Commission Scientific Committee, editor. Twelfth Regular Session. Bali, Indonesia, August 3-11.

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.

United States Coast Guard. 2013. American Samoa Area Contingency Plan (ANNEX A of the Hawaii Area Contingency Plan). 444 p.

U.S. Department of Health and Human Services. 1995. Toxicological profile for polycyclic aromatic hydrocarbons (PAHs). US Agency for Toxic Substances Disease Registry, Atlanta, GA. 487 p.

U. S. Department of the Navy. 2019. About Us. Commander, U.S. Pacific Fleet. [Accessed September 12, 2019]. https://www.cpf.navy.mil/about/#aor

U.S. Energy Information Administration. 2022. American Samoa Territory Energy Profile. [Accessed October 13, 2022]. https://www.eia.gov/state/print.php?sid=AQ.

U.S. Environmental Protection Agency (EPA). 2022. EPA Biological Evaluation for the Hawaii Dispersant Preauthorization Plan (July 2019). 256 p.

U.S. EPA. 2010. Climate Change Indicators in the United States.80.

USFWS and NMFS. 1998. Endangered Species Consultation Handbook. Procedures for inducting Consultation and Conference Activities under Section 7 of the Endangered Species Act. 315 p.

USGCRP. 2017. Climate Science Special Report (CSSR): Fourth National Climate Assessment, Volume I. Wuebbles, D. J., D. W. Fahey, K. A. Hibbard, D. J. Dokken, B. C. Stewart, T. K. Maycock, editors. US Global Change Research Program. Washington, DC. p. 470.

Van Houtan, K. S. 2011. Assessing the impact of fishery actions to marine turtle populations in the North Pacific using classical and climate-based models. NMFS. 25 p.

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

Verdote, D. M., and A. Ponzo. 2014. First record of *Manta alfredi* in the Bohol Sea, Philippines. Program and Poster Abstracts of Shark International. Durban, South Africa.

Viada, S. T., R. M. Hammer, R. Racca, D. Hannay, M. J. Thompson, B. J. Balcom, and N. W. Phillips. 2008. Review of potential impacts to sea turtles from underwater explosive removal of offshore structures. Environmental impact assessment review. 28(4-5):267-285.

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

WCPFC. 2017. Best handling practices for the safe release of mantas & mobulids. WCPFC Summary Report Attachment P. Manila, Philippines 3–7 December 2017.

WCPFC. 2021. Science and Scientific Data Functions. Available at: https://www.wcpfc.int/node/29966 [Accessed last on: July 12, 2021].

Wegner, N. C., and D. P. Cartamil. 2012. Effects of prolonged entanglement in discarded fishing gear with substantive biofouling on the health and behavior of an adult shortfin mako shark, *Isurus oxyrinchus*. Marine Pollution Bulletin. 64(2):391-394.

White, E. R., M. C. Myers, J. M. Flemming, and J. K. Baum. 2015. Shifting elasmobranch community assemblage at Cocos Island—an isolated marine protected area. Conservation Biology. 29(4):1186-1197.

White, W. T., C. Bartron, and I. C. Potter. 2008. Catch composition and reproductive biology of Sphyrna lewini (Griffith & Smith) (*Carcharhiniformes, Sphyrnidae*) in Indonesian waters. Journal of fish biology. 72(7):1675-1689.

Wiley, D. N., R. A. Asmutis, T. D. Pitchford, and D. P. Gannon. 1995. Stranding and Mortality of Humpback Whales, Megaptera-Novaeangliae, in the Mid-Atlantic and Southeast United-States, 1985-1992. Fishery Bulletin. 93(1):196-205.

Williams, S. E., L. P. Shoo, J. L. Isaac, A. A. Hoffmann, and G. Langham. 2008. Towards an integrated framework for assessing the vulnerability of species to climate change. PLoS Biol. 6(12):2621-2626.

Worm, B., H. K. Lotze, I. Jubinville, C. Wilcox, and J. Jambeck. 2017. Plastic as a Persistent Marine Pollutant. Annual Review of Environment and Resources, Vol 42. 42(1):1-26.

WPRFMC. 2016. Report of the Rare Events Bycatch Workshop Series. Honolulu, HI. 45 p.

WPRFMC. 2018. 2017 Annual Stock Assessment and Fishery Evaluation Report Pacific Island Pelagic Fishery Ecosystem Plan. In: Kingma, E., A. Ishizaki, T. Remington, S. Spalding, editors. Western Pacific Regional Fishery Management Council. Honolulu, Hawaii 96813 USA.

WPRFMC. 2019. 2018 Annual Stock Assessment and Fishery Evaluation Report Pacific Island Pelagic Fishery Ecosystem Plan. In: Kingma, E., A. Ishizaki, T. Remington, S. Spalding, editors. Western Pacific Regional Fishery Management Council. Honolulu, Hawaii 96813 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. 372 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 for the Pacific Pelagic Fisheries Fishery Ecosystem Plan 2021. T Remington, M Fitchett, A Ishizaki, J DeMello (Eds.). Honolulu: Western Pacific Regional Fishery Management Council. Honolulu, Hawaii 96813 USA.390 pp. + Appendices. USA.

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

Zychowski, G. V., and C. A. Godard-Codding. 2017. Reptilian exposure to polycyclic aromatic hydrocarbons and associated effects. Environmental Toxicology and Chemistry. 36(1):25-35.

9 APPENDIX A: GENERAL EXPOSURE PROFILE

9.1 Stressors Not Likely to Adversely Affect Listed Resources

9.1.1 Vessel Noise

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 ASLL fishing vessels. However, NMFS expects that vessel noises would have an insignificant effect on listed species because they would not be expected to result in measurable responses (should never reach the scale where harm or harassment occurs).

Given the size of the ASLL fishery (the small number of vessels in the fishery and the wide area they cover), the fact that the sound field produced by the vessels in the fishery is relatively small and would move with the vessel, the animals would be moving as well, vessel transit vectors would be predictable, sudden or loud noises would be unlikely or infrequent, we would expect that any exposure to noises generated by this fishery would be short-term and transient, and generally ignored. Numerous studies demonstrate that marine animals are unlikely to change their behavior when confronted with stimuli with these attributes, and we would also expect masking would be highly unlikely to occur, if not improbable. 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 will elicit behavioral reactions to the level of harm or harassment on oceanic whitetip sharks and giant manta rays. Thus, NMFS is reasonably certain this stressor will only have insignificant effects on these two species.

9.1.2 Collision with vessels

The proposed action would expose oceanic whitetip sharks and giant manta rays to the risk of collision with vessels. ASLL 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. Oceanic whitetip sharks and giant manta rays 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 ASLL 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.

ASLL fishery vessels also burn fuel and emit carbon into the atmosphere during fishing operations and transiting. The majority of ASLL vessels have diesel engines with an average of 460 horsepower (Ito et al. 1998). 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 do not have an accurate estimate of the carbon footprint of the ASLL fishery, we are reasonably certain the contribution to global greenhouse gases to be relatively inconsequential based on the low number of participants in the fishery.

9.1.4 Vessel Groundings

Environments where 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.

Additionally, 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 4.90×10^{-3} chance a vessel may potentially ground and trap an individual is extremely unlikely, and therefore discountable.