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

Action Agency:	National Marine Fisheries Service, Pacific Islands Region, Sustainable Fisheries Division
Federal Action:	Authorization of the American Samoa Longline Fishery
Consultation Conducted by:	National Marine Fisheries Service, Pacific Islands Region, Protected Resources Division
NMFS File No. (ECO):	PIRO-2019-00634
PIRO Reference No.:	I-PI-18-1727-AG
Approved By:	MALLOY.SARAH.JOAN.1262526743 Digitally signed by MALLOY.SARAH.JOAN.1262526743 Date: 2023.05.15 08:02:06 -10'00' Sarah Malloy Acting Regional Administrator, Pacific Islands Region
Date Issued:	5/15/23

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1 INTRODUCTION

Section 7(a)(2) of the Endangered Species Act of 1973, as amended (ESA; 16 U.S.C. 1536(a)(2)) requires each federal agency to 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 endangered and threatened species, and critical habitat that has been designated for those species.

On July 5, 2022, in *Animal Legal Defense Fund v. Haaland, et al.*, 4:19-cv-06812-JST, the U.S. District Court for the Northern District of California issued an order vacating the 2019 regulations that were revised or added to 50 CFR part 402 in 2019 ("2019 Regulations," see 84

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[1] Take" is defined by the ESA as harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct. NMFS defines "harass" as to "create the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavioral patterns which include, but are not limited to, breeding, feeding, or sheltering" (Application and Interpretation of the Term "Harass" Pursuant to the Endangered Species Act: NMFS Guidance Memo May 2, 2016). NMFS defines "harm" as "an act which actually kills or injures fish or wildlife." 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.

FR 44976, August 27, 2019) without making a finding on the merits. On September 21, 2022, the U.S. Court of Appeals for the Ninth Circuit granted a temporary stay of the district court's July 5 order. On November 14, 2022, the Northern District of California issued an order granting the government's request for voluntary remand without vacating the 2019 regulations. The District Court issued a slightly amended order two days later on November 16, 2022. As a result, the 2019 regulations remain in effect, and we are applying the 2019 regulations here. For purposes of this consultation and in an abundance of caution, we considered whether the substantive analysis and conclusions articulated in the biological opinion and incidental take statement would be any different under the pre-2019 regulations. We have determined that our analysis and conclusions would not be any different.

This biological opinion has been prepared in accordance with the requirements of section 7 of the ESA, the implementing regulations (50 CFR 402), agency policy, and guidance and is based on information contained in SFD's biological evaluation (NMFS 2019a), NMFS and FWS recovery plans and status reviews for sea turtles (NMFS and FWS 1998a, 1998b, 1998c, 1998d, 2007a, 2007b, 2007c, 2007d, 2013a, 2013b, 2014; Seminoff et al. 2015), and elasmobranchs (Miller and Klimovich 2016; Young et al. 2017) and the Western Pacific Fishery Management Council (Council) annual stock assessment and fishery evaluation reports (SAFE reports) for Pacific Island pelagic FEP for 2017 (WPRFMC 2018, 2019, 2020, 2021, 2022), 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 (DPSs) 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 DPS 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 DPSs, 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 March 12, 2019, NMFS SFD requested reinitiation of formal consultation. Consistent with 50 CFR 402.16, SFD reinitiated consultation because the ASLL met three of the four possible reinitiation triggers. The ASLL exceeded the amount of incidental take for 6 DPSs of sea turtles, and several new species were listed that may be affected by the action. Specifically, the following conditions met the reinitiation trigger:

- From July 2015 through June 2019, NMFS observers reported three observed interactions with hawksbill sea turtles, one in 2016, and two in 2018. Three interactions with hawksbills result in an expanded fleet-wide total of approximately 21 interactions. As a result, 21 interactions have exceeded the 3-year ITS of 6 interactions authorized under the 2015 BiOp.
- NMFS observers have also reported eight interactions with olive ridley sea turtles, one in 2015, three in 2016, and two in both 2017 and 2018. Eight interactions with olive ridley sea turtles result in an expanded fleet-wide total of approximately 51 interactions. As a result, 51 interactions have exceeded the 3-year ITS of 33 interactions authorized under the 2015 BiOp.
- From July 2015 through June 2019 the NMFS Observer Program reported 14 observed fishery interactions with green sea turtles. Between 2016-2018 there were four interactions per year and two interactions through the second quarter of 2019. These 14 interactions result in an expanded fleet-wide total of approximately 72 interactions. The estimated total interactions for each of the DPSs are prorated based on the estimated proportions indicated in the 2015 BiOp. Because NMFS cannot determine the DPS a green sea turtle belongs to at the time of an interaction, we must allocate a portion of each interaction to each of the five DPSs of green sea turtles whose pelagic distribution overlaps with the American Samoa fishery from genetic data collected from green sea

turtles captured in the fishery[2]. As a result of this proration, four of the five green DPSs described in the 2015 BiOp, namely the East Indian West Pacific, Southwest Pacific, Central South Pacific, and East Pacific, have exceed the 3-year ITS authorized under the 2015 BiOp.

- In January 2018, NMFS listed two new species, the giant manta ray and the oceanic whitetip shark, as threatened under the ESA (83 FR 2196 and 83 FR 4153, respectively). The ASLL fishery interacts with these two species.
- On September 28, 2018, NMFS issued a final rule to list the chambered nautilus as threatened under the ESA (83 FR 48976). There is currently no information to suggest that the ASLL fishery has interacted with the chambered nautilus, however, it occurs within the Action Area.

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.

1.2 Description of the Proposed Action

NMFS SFD proposes to authorize the pelagic ASLL fishery as currently managed under the existing regulatory framework of the Pelagic FEP and other applicable laws including the Endangered Species Act (16 U.S.C. 1531 et seq.) and Marine Mammal Protection Act (16 U.S.C. 1361 et seq.). As specified above in the Consultation History, new species have been listed as threatened or endangered under the ESA and pursuant to criteria (2), (3), and (4) of the regulations at 50 CFR 402.16, reinitiation of consultation is therefore required for the authorization of the United States ASLL fishery as currently managed. 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, pomfrets, 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 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

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[2] The NMFS Southwest Fisheries Science Center conducted a direct count genetic analysis from green turtle samples collected by observers in the fishery. Portions of each interaction with a green sea turtle are allocated to a species by multiplying the genetic percentage of each species (from the direct count analysis), by the number of interactions [2](Observed interactions) x (Species' Genetic Fraction). For example 60(.50) = 30 interactions with the Central South Pacific green sea turtle).

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 two 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.

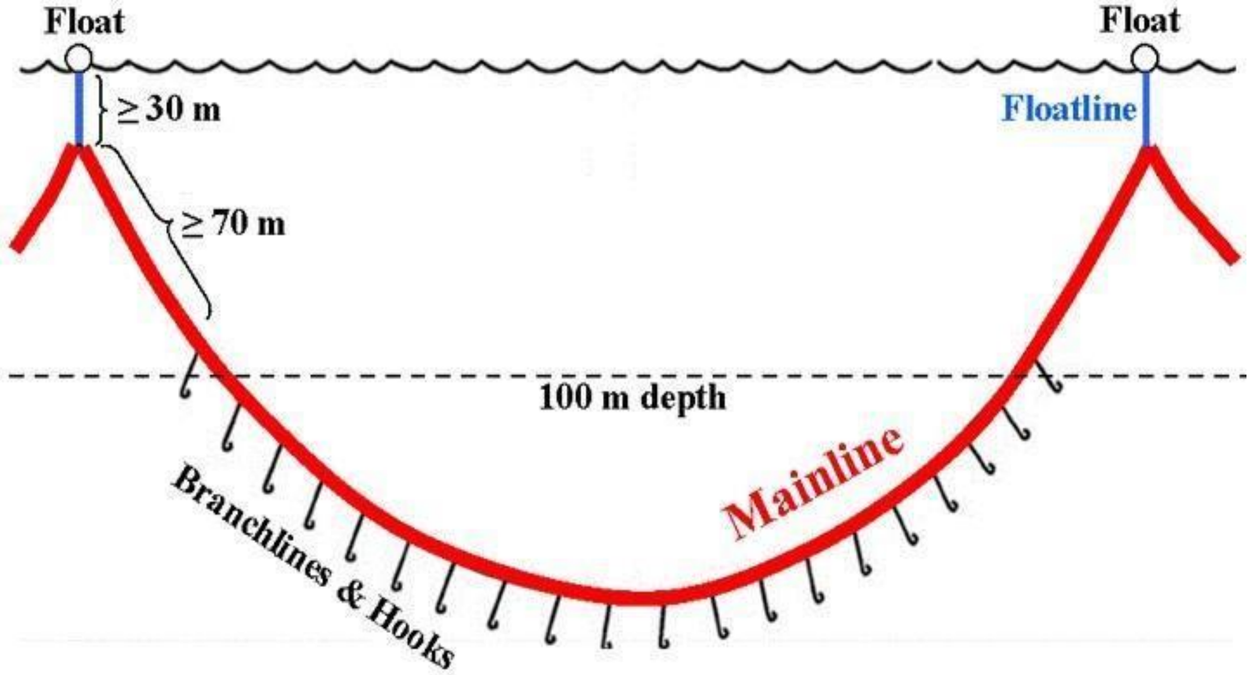


Figure 1. Gear configuration in the ASLL fishery (NMFS 2019a).

The Pelagic FEP and its implementing regulations at 50 CFR 665 contain a number of requirements to prevent and mitigate the effects of longline fisheries 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.
- Western and Central Pacific Fisheries Commission (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.

Sea Turtle Interaction Mitigation:

- Regulations require all hooks must be placed at least 100 m deep, have 10 m branchlines, and applies to vessels >40 ft. This is accomplished by requiring a minimum float line length of 30 m, together with a minimum of 70 m between the float line and any branch lines.

Sea Turtle Handling and Mitigation Measures:

- Adhere to regulations for safe handling and release of sea turtles
- Have on board the vessel all required turtle handling and dehooking gear specified in regulations.
- Comatose sea turtles are to be brought aboard and resuscitation techniques should be performed according to the procedures described at 50 CFR 665.812 and 50 CFR 223.206.

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.

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. Additionally, fishermen are required to remove fishing gear from any oceanic whitetip shark caught in all longline fisheries operating under the Pelagic FEP to reduce the amount of fishing gear (trailing gear) attached to released oceanic whitetip sharks, and requires that the animals should be left in the water and a dehooker or line clippers shall be used to remove trailing gear (50 CFR 665.811).

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 prohibit 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 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, and Swains Island (Figure 2).

On July 9, 2021, 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.

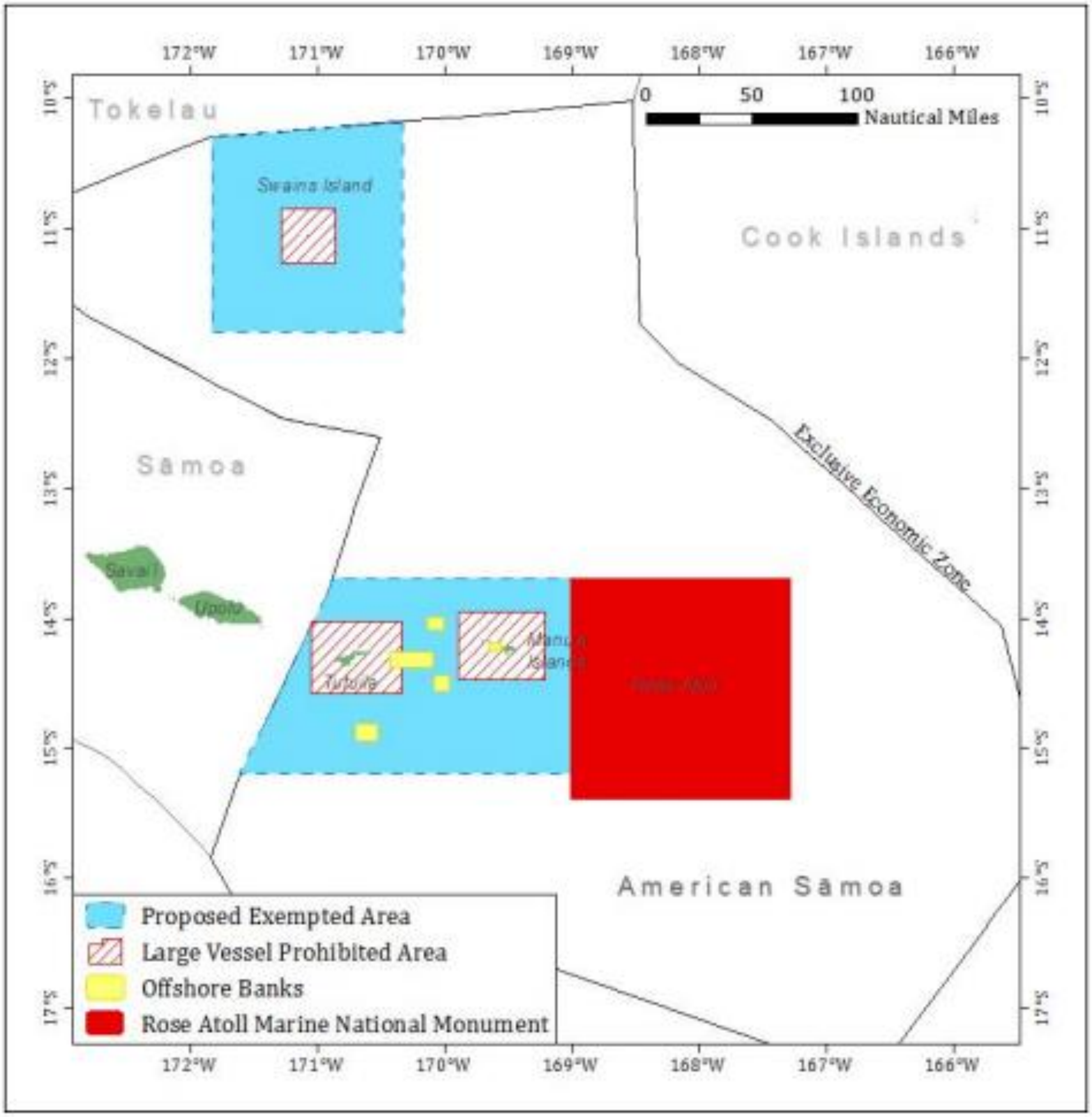


Figure 2. Current LVPA boundaries in waters around American Samoa and the longline vessel exempted areas as defined by 86 FR 36239.

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. 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 longline fishery (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). 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.

Table 1 illustrates the number of active vessels and fishing effort (active vessels, trips, sets, and hooks deployed) in the fishery from 2007 to 2021. 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 representing a decrease of 76% from 2007. In the last 15 years, there were year-over-year increases in effort four times and decreases ten times. Since 2007, the fishery metrics for vessels, trips, sets, and hooks have declined at statistically significant ($p<0.05$) rates.

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 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 (2017-2021); i.e., 13.4 vessels, 1,580 sets, and 4,446,380 hooks.

Table 1. Number of active vessels and fishing effort in the ASLL fishery, 2007-2021 (2007 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. 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 define the word "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.

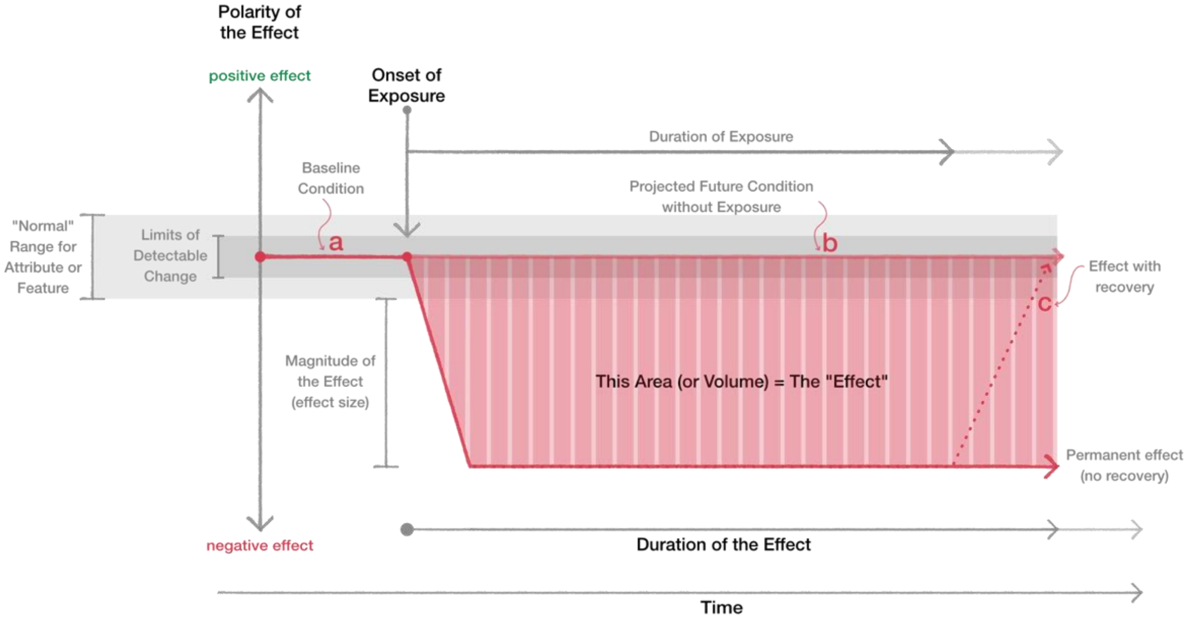


Figure 3. A schematic of the various elements encompassed by the word "effect." The vertical bars in the figure depict a series of annual "effects" (negative changes from a pre-existing or "baseline" condition) that are summed over time to estimate the action's full effect. See text for a more complete 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 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 ESA section 7 regulations define "jeopardize the continued existence of "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 (83 FR 35178, July 25, 2018). This definition requires our assessments to address four primary variables:

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

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

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

The third of these variables—distribution— refers to the number and geographic arrangement of the populations that comprise a species. Jeopardy analyses must focus on populations because the fate of species is determined by the fate of the populations that comprise them: species become extinct with the death of the last individual of the last population. For that reason, jeopardy analyses may consider changes in the number of populations, which provides the strongest evidence of a species' extinction risks or its probability of recovery. Jeopardy analyses also may consider changes in the spatial distribution of the populations that comprise a species because such changes provide insight into how a species is responding to long-term changes in its environment (for example, to climate change). The spatial distribution of a species' populations also determines, among other things, whether all of a species' populations are affected by the same natural and anthropogenic stressors and whether some populations occur in protected areas or are at least protected from stressors that affect 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 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). For the analysis in the biological opinion, we used 40 years for all species (except West Pacific leatherback sea turtles) because we have adequate information about population trends, interactions rates, and effects from interactions to reasonably predict the action's effects over that timeframe. For West Pacific leatherback sea turtles, we note 1) the population growth rate estimated by Martin et al. (2020a; -6.0% [95% CI: -24.1 to 12.2%]) represents a spread of 36% and includes both positive and negative growth rates, and 2) the recent nest numbers from the Jamursba Medi and Wermon index beaches (Lontoh et al. In Prep, see Figure 16) show stable numbers of annual nests from 2017 to 2021, inconsistent with the projected mean decline of Martin et al. (2020a, though within the 95% CI) and used in the PVA of Siders et al. (2023). New nesting data suggests a somewhat more favorable, stabilizing trend than the population growth rates projected by Martin et al. (2020a) and that was used by Siders et al. (2023) to estimate 2021 population sizes. However, this data is not yet available in sufficient detail that will allow an update to the growth trend analysis of Martin et al. (2020a). To date we have only received the graph shown in Figure 16 and detailed data on nest counts per month are needed to update the model. In addition, the data from 2012 to 2021 that show stable to increasing trends may not be of sufficient duration compared to the dramatic declines from the 1980's to 2012 to update our understanding of the current status of this population. From the 1980's to 2013, nesting at the index beaches declined by a factor of 13 (nest numbers in the 1980's were about 13 times higher than in 2013). In contrast, from 2013 to 2021, nesting at the index beaches increased by a factor of about 2. While this is encouraging, it does not ameliorate the historic declines. Thus, we consider the modeling results of Martin et al. (2020a) as the best available estimates for long-term West Pacific leatherback population trends, while acknowledging the uncertainty around these estimates and the need to update that model with the new data once it is available.

This step of our analyses considers data available on the particular populations and species affected by an action. However, this step of our analyses is also 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 or endangered species. 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:

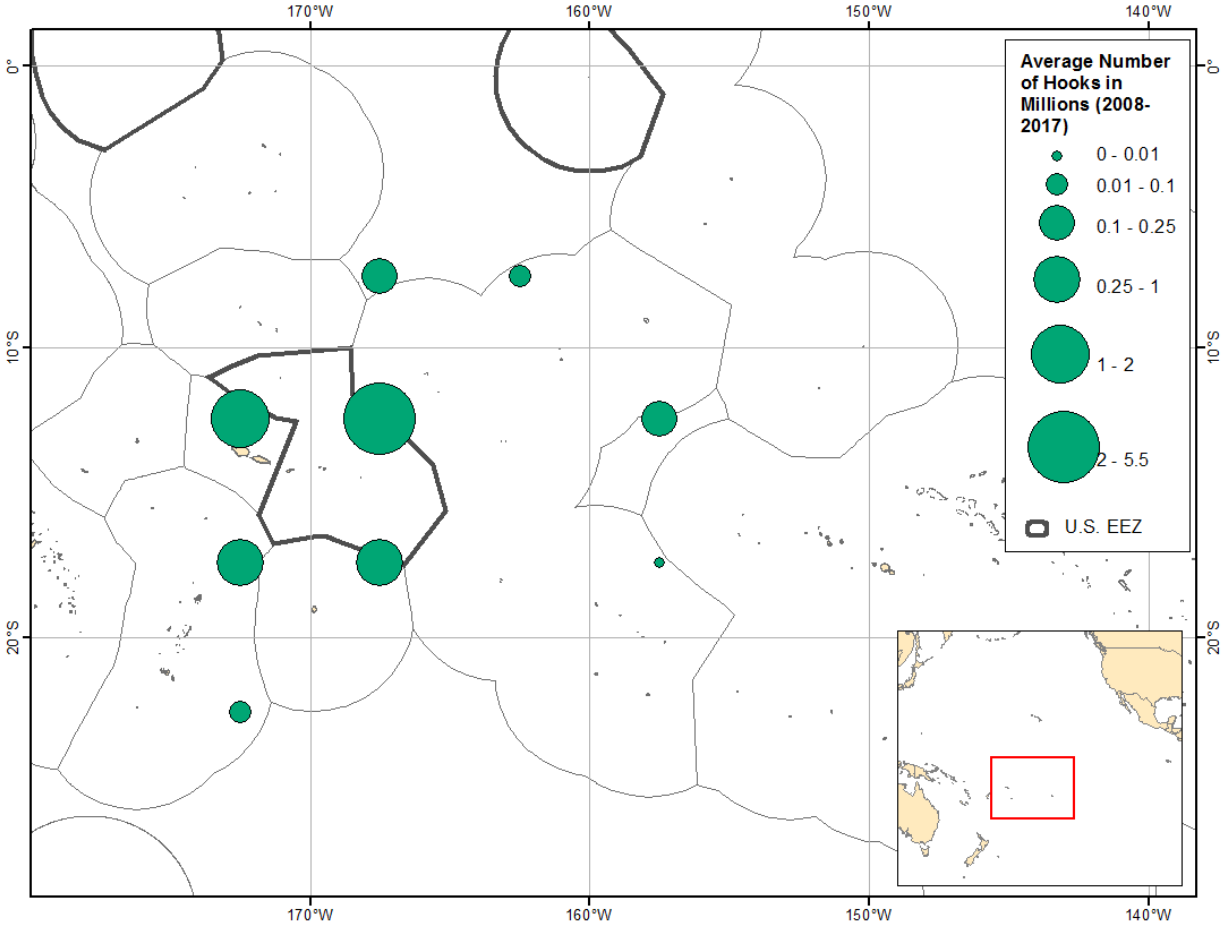
- Interaction with, including capture of non-target species, listed species, or their prey;
- Derelict gear;
- Introduction of oily discharges, cardboard, plastics, and other waste into marine waters;
- Collisions with vessels;
- Vessel noise; and
- Vessel emissions.

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 began by considering the general location of their 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 each species would be exposed to the stressors described above. Where we concluded that the likelihood of exposure is extremely unlikely or the response will not rise to the scale of take, we do not include the stressor further in our exposure or response analyses (these determinations are explained in Appendix A). As a result, we focused our attention on the primary threat, the observed interactions, and characterizing the effects of those interactions on listed resources.

The stressors associated with the 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 each species 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 those listed species and their designated critical habitat.

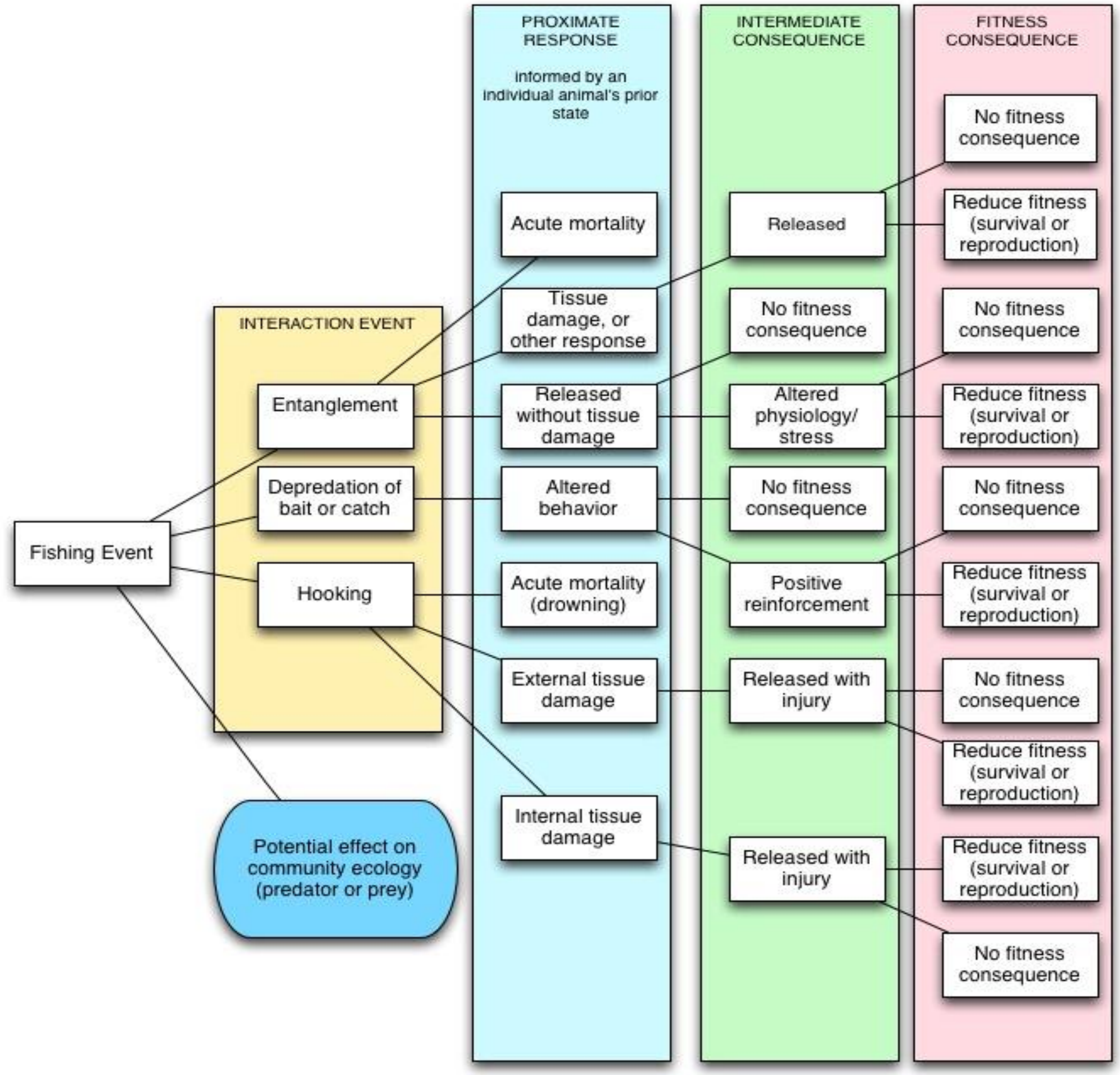


Figure 5. Conceptual model of the potential responses of an animal's exposure in the ASLL fishery and the pathways between exposure (the interaction event the individual animal has with the fishing gear), responses, and potential fitness consequences to the individual.

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 interaction 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. Due to confidentiality rules under section 402 of the Magnuson-Stevens Fishery Conservation and Management Act, the 2020 and 2021 American Samoa observer reports are not available. Furthermore, the number of observed vessel trips during these years were so low the data may not be sufficiently reliable to use in our analysis. The number of hooks set in the ASLL during 2020 and 2021 are slightly lower than the preceding years (WPRFMC 2022). Thus, from what we know about the observer data and the available information about the fishery, the number of interactions in 2020 and 2021 are in line or slightly less than the metrics from 2006 to 2019 we use in this analysis.

From there, we developed mortality estimates for both unobserved historic interactions and anticipated future interactions. Some species like 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 FWS-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 Simulations using Bayesian State-Space Population Viability Analyses for the West Pacific Population of Leatherback Sea Turtles

A Bayesian state-space nesting trend analysis and Population Viability Analysis (PVA) was used to assess the current status of the endangered West Pacific leatherback sea turtle to examine the long term viability of those populations (Martin et al. 2020a, 2020b; note that in this section we use the term ‘population’ to refer the West Pacific leatherback sea turtle). Martin et al. (2020b) specifically considered the impact of the ASLL on West Pacific leatherbacks. The Martin et al. (2020b) model was subsequently updated with 2017 to 2021 DSLL capture data for leatherback sea turtles (Siders et al. 2023). We include Siders et al. (2023) here as it updates the PVA without the impact of the DSLL, which is relevant to our analysis of the current status of the population. In addition, due to the small numbers of leatherbacks that have been captured in the ASLL, Martin et al. (2020b) used data from the DSLL to parameterize the fishery mortality and length distribution of captures in the ASLL. While Siders et al. (2023) did not rerun the Martin et al. (2020b) model for ASLL, the only difference would be lower numbers of annual captures compared to the DSLL and a slightly lower magnitude of impact. The purpose of the analyses are to bring together current knowledge of the ecology of the species of concern using nest count data to forecast the population trends and extinction risks over defined time frames. The Bayesian analysis addresses some of the shortcomings of poor data, imprecise parameter estimates, and sampling and process variance in demographic parameters by exploring the full range of probability distributions for parameters of interest.

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

Annual nest counts for each population were used as the foundational data for these analyses because this represents the most reliable long term census data for these populations. Ideally, we would have key demographic parameters on each of the populations to support robust analyses of abundance and trends, such as: age at maturity, survivorship across life stages, adult and hatchling sex ratios, clutch frequency, nesting success and remigration intervals for each subpopulation that comprise the population. However, comprehensive population-specific data sets are not available for either population. Nest counts provide an index of female abundance for the specific nesting beaches; however, assumptions must be made on clutch frequency and, because females do not return to nest every year, we also make assumptions on remigration intervals to convert nest counts to adult female abundance. Results of the PVA are also only applicable to the portion of each population, or subpopulation, for which we have nest counts. No data are available to provide an index of abundance for other age classes. Due to this data limitation, estimating r from these data does not represent the true population growth rate for the population of interest but only the adult female portion of the population. We also note that the updated leatherback PVA of Siders et al. (2023) did not have new nesting data available to incorporate in the model, and they therefore projected populations based on the end of the time series in Martin et al. (2020a; 2017) to the end of the updated fishery capture data time series (2021) to estimate annual numbers of nesting females.

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

Imputation:

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

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

Trend:

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

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

3. The clutch frequency is constant through time.

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

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

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

7. The index beaches were observed with constant effort or were effort-corrected by the data collectors prior to being provided to NMFS' Pacific Islands Fisheries Science Center (PIFSC) to form the time series used in these analyses.

8. The remigration interval is constant through time.

Projections:

1. The population will face the same threats in the future as it did in the past. The PVA applies the population growth rate from the past, which was influenced by some particular magnitude of threats, to the future trajectory of the population. It does not account for the impacts of new or increasing threats, such as the loss of nesting habitat from climate change. [3]

The PVA model was extended to include a "mortality[4]" model to assess the effect of the ASLL fishery on the West Pacific population of leatherback sea turtles (Martin et al. 2020b). Martin et al. (2020b) used turtle lengths and assigned at-vessel and post-interaction mortality rates from the observer data in the DSLL fishery (due to the limited observer data available for the ASLL) to parameterize a multivariate normal distribution that characterized historical length-mortality interactions for the fishery, and this distribution was used to generate estimates for the length and mortality of turtles that the fishery would interact with in the future.

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

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

Due to the limited number of known lengths and assigned mortality rates for the DSLL and ASLL fisheries, the length-mortality distributions used in the model were much less informed (i.e., less data-rich) compared to the SSLL, thereby leading to greater uncertainty about historical and future population impacts. As a result, Martin et al. (2020b) added stochasticity to the simulations for the population impact of both historical and future fishery interactions. Siders et al. (2023) updated the model of Martin et al. (2020b) and the impact of the DSLL on West Pacific leatherback sea turtles. Key updates included updating the model with observed and estimated leatherback captures from 2018 to 2021, along with subsequent updates to anticipated captures and fishery mortality rates and a revised approach to how maturity was assigned. Siders et al. (2023) increased the survival of smaller (younger) leatherback sea turtles by proportioning annual survival between juvenile and adult survival rates, with adult survival rates representing a larger proportion of total survival rates with increasing size (Siders et al. 2023). Again, we consider this model here because it updates the population's probability and times to reach population thresholds without the DSLL fishery.

Because the PVA considers only adult nesting females, each sea turtle interacting with the fishery was converted to an adult nester equivalent (ANE) to assess the impact to the population from, for adult-sized turtles, the loss of any adult females including the future productivity that female would have contributed to the population; and for juvenile-sized turtles, the future productivity that individual may have contributed to the population. Converting adult-sized turtles to ANE requires an estimate of the population sex ratio to determine the probability that the captured individual was female, and future productivity requires an estimate of adult survival rate and remigration interval. In addition to population sex ratio, adult survival rate and remigration interval, converting juvenile-sized turtles to ANE also requires an estimate of the time remaining until the turtle reaches maturity, and the probability that the turtle survives to maturity. Martin et al. (2020a) details the size-at-age relationship, age at maturity, sex ratio, remigration interval and survival rates used for each population. Siders et al. (2023) used the same size-at-age relationship, sex ratio, remigration interval, and survival rates as Martin et al. (2020a,b) but they increased the minimum size at maturity while allowing for the possibility of smaller turtles to be mature. To account for unobserved historical interactions in the ASLL fishery in the historical take analysis, estimates of total fishery and species specific interactions were taken from the Annual Stock Assessment and Fishery Evaluation Report for U.S. Pacific Island Pelagic Fisheries Ecosystem Plan 2018 (WPRFMC 2019 and McCracken references therein; Martin et al. 2020a). The estimated unobserved interactions (historical) were the difference between the estimated total interactions and the known observed interactions. Each individual of the anticipated interaction received a length and fishery mortality from a bivariate normal distribution, a back-calculated age with years until maturity based on a von Bertalanffy growth model, juvenile survival rate, sex, and a remigration interval. From this, the probability of being female, the probability of surviving until maturity (a combination of size and age at capture and juveniles survival rate), and the probability of the interaction resulting in a mortality were estimated to yield an ANE for each interaction (Martin et al. 2020a, 2020b).

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

1.5.3 Climate Change

Future climate will depend on warming caused by past anthropogenic emissions, future anthropogenic emissions and natural climate variability. NMFS' policy (NMFS 2016a) is to use climate indicator values projected under the Intergovernmental Panel on Climate Change (IPCC)'s Representative Concentration Pathway (RCP) 8.5 when data are available or best available science that is as consistent as possible with RCP 8.5. RCP 8.5, like the other RCPs, were produced from integrated assessment models and the published literature; RCP 8.5 is a high pathway for which radiative forcing reaches >8.5 W/m^2 by 2100 (relative to pre-industrial values) and continues to rise for some amount of time. Since NMFS (2016a), RCP have been updated to a new version called SSP5-8.5 (ICCP 2021). A few projected global values under SSP5- 8.5 are noted in Table 2. Presently, the IPCC predicts that climate-related risks for natural and human systems are higher for global warming of 1.5 ºC but lower than the 1.7 ºC presented in Table 2 (IPCC 2021). Changes in parameters will not be uniform, and IPCC projects that areas like the equatorial Pacific will likely experience an increase in annual mean precipitation under scenario 8.5, whereas other mid-latitude and subtropical dry regions will likely experience decreases in mean precipitation. Sea level rise is expected to continue to rise well beyond 2100 and while the magnitude and rate depends upon emissions pathways, low-lying coastal areas, deltas, and small islands will be at greater risk (IPCC 2021).

Table 2. Projections for certain climate parameters under Representative Concentration Pathway 8.5 (values for temperature from Table 4.2 in IPCC 2021; values for sea level from Chapter 9 of IPCC 2021).

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

Given the limited data available on sea turtle populations, and other listed species like whales, sharks, and rays that are adversely affected by the proposed action, and the inherent challenges with creating population models to predict extinction risks of these species, we are not inclined to add more uncertainty into our assessment by creating climate models with little data to parameterize such models. Since trying to apply a climate based model in 2012 to the SSLL biological opinion (NMFS 20132n), we have learned a few key important lessons: the climate based model incorporating fixed age (lag) is unrealistic given variability ages at sexual maturity for loggerhead and leatherback sea turtles, and fails to consider variation in age of the nesting cohort; studies have shown juvenile loggerhead sea turtles are distributing more widely than thought, and thus are likely impacted in ways not considered under the previous model; a new dispersion model on leatherback sea turtles suggest they too may be dispersing more broadly, and affected differently than previously considered; the model did not account for impacts to more than two life-stages; and arguably, most importantly, the models did not perform as expected because the predictions were wrong for leatherback sea turtles the majority of the time, and predictions for loggerhead sea turtles were wrong half the time (Kobayashi et al. 2008, 2011; Van Houtan 2011; Van Houtan and Halley 2011; Allen et al. 2013; Arendt et al. 2013; Briscoe 2016a, 2016b; Jones et al. 2018; see also Jones memo 2018). Instead, in this assessment we rely on systematic assessments of available and relevant information to incorporate climate change in a number of ways.

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

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

1.5.4 Statistical Intervals and Anticipated Future Captures Used in this Assessment

Throughout our assessment, we calculate a variety of metrics to evaluate the status of the species and the effects of the action. Where we have data to develop statistical inferences, we relied on the credible interval or the confidence interval to estimate the probability and the uncertainty of the estimate. Confidence intervals were used to capture the level of confidence that the true value (i.e., 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 per year on average over time, and provides the most reliable estimate of the expected number of interactions between the fishery and the listed species over an extended time frame. However, in any one year, higher or lower numbers of interactions are likely to occur. 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 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 2019 on the time series of estimated annual 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 captures from 2012 to 2016, and our last value is the average of captures from 2015 to 2019. From that resulting time series of running averages, we find the maximum value and that is the maximum 5-year running average. Multiplying the maximum 5-year running average by five gives the maximum 5-year running sum. We used both of these metrics, maximum 5-year running average and sum, along with the mean and 95th percentile values from McCracken (2019a) in our assessment of the impact of the fishery on the species considered in this biological opinion. Given the year-to-year variability in the number of captured ESA-listed species, a single year estimate of captures that is not routinely exceeded would need to be a large value, such as the 95th percentile. However, using the 95th percentile to describe fishery impacts to ESA-listed species over longer time periods would lead to estimates of impacts that are exaggerated and not reasonably likely to occur. Because of this limitation, recent fisheries biological opinions have relied on rolling 5-year estimates, such as the 5-year running sum described above, to describe the likely impact of fishery captures on ESA-listed species (see NMFS 2020c, 2021d, 2021e as examples).

Our assessment does not focus on the lower limits of the 95% confidence intervals because they represent lesser risk to endangered and threatened species; nevertheless, values between the mean and the lower limit 95% intervals are also compatible with the data and are not ignored in our analyses. However, for risk analyses over longer time intervals (for example, 25 years) we rely largely on the mean estimates because long time series are most likely to include values above and below the mean.

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, except for the West Pacific population of leatherback sea turtles for which we use 20 years. We are not reasonably certain we can reliably predict the effects of the fishery's continued operations beyond that period of time. We have, however, carried the effects of the fishery 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.

1.5.5 Evidence Available for this Consultation

Section 7(a)(2) of the ESA and its implementing regulations require NMFS to use the best scientific and commercial data available during consultations. The most credible and relevant data available for our exposure and response analyses are (1) data NMFS' observer program collects on interactions between the ASLL fishery and endangered or threatened species collected by as bycatch between 2006 and 2019 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 Annual Stock Assessment and Fishery Evaluation Reports (WPRFMC 2019. 2020, 2021, 2022). As a result, these represent the best scientific and commercial data available at the time of consultation. The 2022 Annual Stock Assessment and Fishery Evaluation Report is not available at the time of this writing.

To support our status assessments, assessments of the expected impacts of the environmental baseline on endangered and threatened species and designated critical habitat, response analyses, and risk assessments, the 2016 Report of the Rare Events Bycatch Workshop Series (WPRFMC 2016), the Bycatch Management Information System (BMIS), and recovery plans that have been developed for the endangered or threatened species that may be affected by the ASLL fisheries. 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 the endangered and threatened species this consultation considered? Specifically, we targeted information that would allow us to identify the number of populations and sub-populations that comprise the endangered and threatened species this consultation considers.
2. What effects have been reported for the endangered and threatened species this consultation considers after interactions with longline gear? We were specifically interested in effects reported from the Pacific, but we also collected any at-vessel, post release, and overall or total mortality rates we could identify that may be pertinent to the species under consideration. We considered surrogate species as well when species-specific information was not available.
3. What, if any, patterns are available in the literature for the endangered and threatened species considered in this consultation after an interaction with longline gear? We were specifically interested in differences between gear types and operational characteristics of fisheries that use either monofilament and/or wire leaders that may inform our effects analysis. We considered related surrogate species when species specific information was not available.
4. What post-release changes in reproductive variables have been reported for the endangered and threatened species this consultation considers after interactions with longline gear? We were specifically interested in data on the effects of capture myopathy and stress pathology on the interval between reproductive events and natality (live births) in any of the endangered and threatened species this consultation considers.

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

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

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

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

Finally, we also relied on data from NMFS' observer program of past interactions between the ASLL fishery and sea turtles and other protected species 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) and 4.65% in 2021. In the absence of observer data, McCracken and Cooper (2022) developed an estimated 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. To be clear, data have been collected by observers in the ASLL since 2006; however, observer coverage was less than 10% from 2006 to 2009, after which it was around 20% until 2019, and from 2020 to 2022 coverage was again less than 10% due to the Covid-19 pandemic.

Importantly, observer data is raw data. To be useful for consultation and decision-making in general, raw data requires processing. Once processed, the data becomes useful for decision-making. When a federal action agency has observational data collected from their specific action, as is the case for the United States 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 data available for consultation. We would not be able to examine the past effects or predict future effects from the fishery if we did not transform the data.

2 STATUS OF LISTED RESOURCES

NMFS has determined that the action that NMFS SFD proposes to authorize, the operation of the ASLL fishery as currently managed, may affect the threatened and endangered species listed in Table 3; these listed resources are provided protections under the ESA.

Table 3. Listed resources within the Action Area that may be affected by the proposed action.

Species/common name	**ESA Status**	**Effective Listing Date/FR Notice**	**Recovery Plan**
Chelonia mydas Central West Pacific Green Sea Turtle	Endangered	05/06/2016 81 FR 20057	
Central South Pacific Green Sea Turtle	Endangered	05/06/2016 81 FR 20057	
East Pacific Green Sea Turtle	Threatened	05/06/2016 81 FR 20057	
East Indian-West Pacific Green Sea Turtle	Threatened	05/06/2016 81 FR 20057	
Southwest Pacific Green Sea Turtle	Threatened	05/06/2016 81 FR 20057	
Eretmochelys imbricata Hawksbill Sea Turtle	Endangered	06/03/1970 35 FR 8491	5/22/98 63 FR 28359
South Pacific Loggerhead Sea Turtle	Endangered	10/24/2011 76 FR 58868	5/22/98 63 FR 28359

Species/common name	**ESA Status**	**Effective Listing Date/FR Notice**	**Recovery Plan**
Lepidochelys olivacea Olive Ridley Sea Turtle	Threatened	08/27/1978 43 FR 32800	5/22/98 63 FR 28359
Olive Ridley Sea Turtle; Mexico (Pacific coast) breeding populations	Endangered	08/27/1978	43 FR 32800
Dermochelys coriacea Leatherback Sea Turtle	Endangered	06/03/1970 35 FR 8491	5/22/98 63 FR 28359
Physeter macrocephalus Sperm Whale	Endangered	12/02/1970 35 FR 18319	12/28/10 75 FR 81584
Sphyrna lewini Scalloped Hammerhead Shark Indo West Pacific	Threatened	09/02/2014 79 FR 38213	
Carcharhinus longimanus Oceanic Whitetip Shark	Threatened	03/01/2018 83 FR 4153	
Manta birostris Giant Manta Ray	Threatened	02/21/2018 83 FR 2916	
Corals *Acropora globiceps*	Threatened	10/10/2014 79 FR 53852	
Acropora jacquelineae	Threatened	10/10/2014 79 FR 53852	
Acropora retusa	Threatened	10/10/2014 79 FR 53852	
Acropora speciosa	Threatened	10/10/2014 79 FR 53852	

Species/common name	ESA Status	Effective Listing Date/FR Notice	Recovery Plan
Euphyllia paradivisa	Threatened	10/10/2014 79 FR 53852	
Isopora crateriformis	Threatened	10/10/2014 79 FR 53852	
Nautilus pompilius Chambered Nautilus	Threatened	10/29/2018 83 FR 48976	

2.1 Critical Habitat

The ESA defines critical habitat as "(i) the specific areas within the geographic area occupied by the species, at the time it is listed … on which are found those physical or biological features (I) essential to the conservation of the species and (II) which may require special management considerations or protection; and (ii) specific areas outside the geographical area occupied by the species at the time it is listed… upon a determination by the Secretary that such areas are essential for the conservation of the species" (16 USC §1532 [5][A]). Critical habitat consists of both a geographic area and elements such as plants or natural features within that area.

No critical habitat for any of the species analyzed in this document has been designated within the Action Area, and therefore critical habitat is not considered further in this analysis.

2.2 Listed Resources Not Likely Adversely Affected

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

Based on our evaluation using these criteria, the following species are not likely to be adversely affected by the proposed longline fishery: South Pacific loggerhead sea turtle; sperm whale; chambered nautilus; *Acropora globiceps; Acropora jacquelineae; Acropora retusa; Acropora speciose; Euphyllia paradivisa, and Isopora crateriformis.* We present the basis for these determinations in Appendix A.

2.3 Status of Listed Species That Are Likely to be Affected

The rest of this section of NMFS biological opinion consists of narratives for each of the threatened and endangered species that occur in the Action Area and that may be adversely affected by the 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 appreciably reduce a species' probability of surviving and recovering in the wild. To fulfill that purpose, each species' narrative presents a summary of (1) the species' distribution and population structure (which are relevant to the distribution criterion of the jeopardy standard); (2) the status and trend of the abundance of those different populations (which are relevant to the numbers criterion of the jeopardy standard); (3) information on the dynamics of those populations where it is available (which is a representation of the reproduction criterion of the jeopardy standard); and (4) natural and anthropogenic threats to the species, which helps explain our assessment of a species' likelihood of surviving and recovering in the wild. This information is integrated and synthesized in a summary of the status of the different species.

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

2.3.1 Green Sea Turtle

In 1978, green sea turtles were listed as threatened (43 FR 32800; July 28, 1978) except for breeding populations that occur in Florida and the Pacific coast of Mexico, which were listed as endangered. In 2016, NMFS and the FWS replaced the global green sea turtle listing with 11 DPSs (Figure 6). These 11 DPSs are demographically, spatially, and genetically independent. Eight of these DPSs are listed as threatened (light- colored polygons in Figure 6) and three as endangered (dark-colored polygons in Figure 6). The DPSs of green sea turtles whose individuals are most likely to occur in the range of the ASLL fishery are limited to those that occur in the South Pacific Ocean: East Indian-West Pacific, Central West Pacific, Southwest Pacific, Central South Pacific and East Pacific.

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

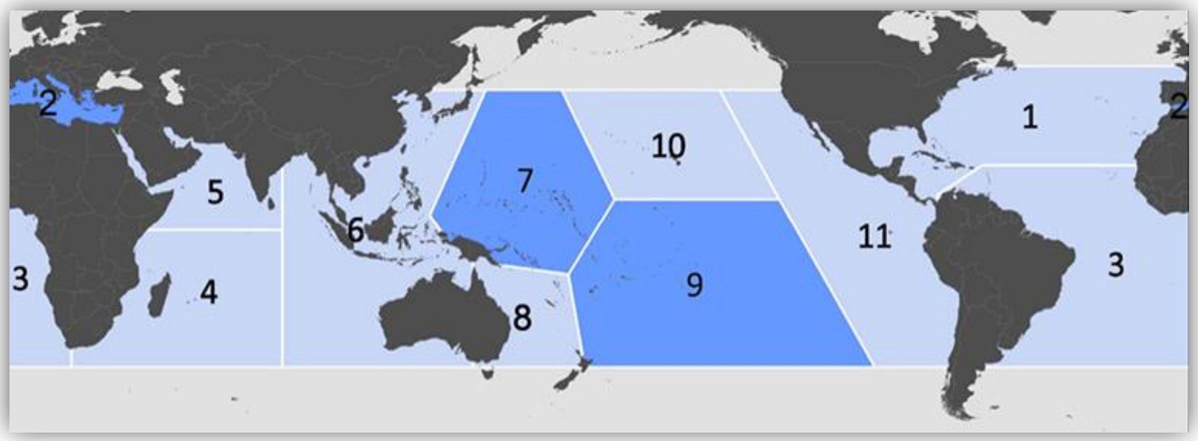

Figure 6. Overview of listed green sea turtle DPSs; (1) North Atlantic; (2) Mediterranean; (3) South Atlantic; (4) Southwest Indian; (5) North Indian; (6) East Indian-West Pacific; (7) Central West Pacific; (8) Southwest Pacific; (9) Central South Pacific; (10) Central North Pacific; and (11) East Pacific.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

PIRO's observer program collected 31 samples from green sea turtles between 2006 and 2017 in the ASLL fishery. Genetic analysis of these samples was conducted by the NMFS Southwest Fisheries Science Center. They used a Bayesian mixed stock analysis (BMSA) of pooled data (P. Dutton pers. comm. July 5, 2018; see Table 4 for summary information from these assessments). Due to the very small sample sizes combined with the presence of haplotypes that are found in multiple rookeries in different DPSs, we cannot determine the percentages of turtles from each of these green turtle DPSs with great certainty. Two haplotypes in particular (CmP20.1 and CmP47.1), are present in three different DPSs (6, 7, 8). Additionally, two orphan haplotypes have not been discovered at nesting sites, only at foraging sites. Therefore, due to the low number of available genetic samples and taking into account the migration and foraging behaviors of the various green sea turtle DPSs in consideration, we used the results from the genetic analysis cautiously. Because the sample size is small and we cannot determine the percentages of green sea turtles from each DPS with great certainty (as evidenced by the large CI, in this case, "CI" refers to "credible interval" rather than "confidence interval"), our analyses considered the impacts of all green sea turtle interactions on each DPS separately.

Table 4. Genetic composition of green turtles sampled from the ASLL fishery between 2006 and 2017 using a mixed stock analysis at the mean and 95% CI (P. Dutton pers. comm. July 5, 2018).

Species	BMSA mean %	95% CI %
Central South Pacific	51.4	30.7-72.9
Southwest Pacific	28.2	11.6-46.5
East Pacific	10.2	2.0-23.2

Species	BMSA mean %	95% CI %
Central West Pacific	8.4	0-24.7
East Indian-West Pacific	1.8	0-11.4

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

General Population Dynamics

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

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

As with other sea turtle species, population censuses are primarily focused on adult females as they can be monitored on nesting beaches. Seminoff et al. (2015) present the best available information for adult female population sizes for green sea turtle species. We parameterized a post-breeding, age-based Leslie matrix model using the fecundity values in Balazs et al. (2015) and the survival rates and age at maturity of Piacenza et al. (2016) and conducted 10,000 simulations drawing from triangular distributions for the survival rates and age to maturity ranges specified in Table 5. All other parameters were held stable as specified in Table 5 across the simulations. The right eigenvector of a Leslie matrix model provides the stable age distribution of the population from which we can derive the percent of the population that are adult females. We then use the adult female population sizes from Seminoff et al. (2015) to estimate the total population size of individuals greater than one year old for each DPS. The results of this analysis are presented in the DPS-specific sections below.

Table 5. Parameters used to estimate total population size for green sea turtle species.

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

Diving and Social Behavior

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

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

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

East Pacific Green Sea Turtle

Distribution and Population Structure

The geographic range of East Pacific green sea turtles extends from the California/Oregon border (USA; 42°N) southward along the Pacific coast of the Americas to central Chile (40°S). The DPS ranges from the aforementioned locations in the United States and Chile to 143°W and 96°W, respectively (Figure 7).This DPS encompasses the Revillagigedos Archipelago (Mexico) and Galapagos Islands (Ecuador).

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

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

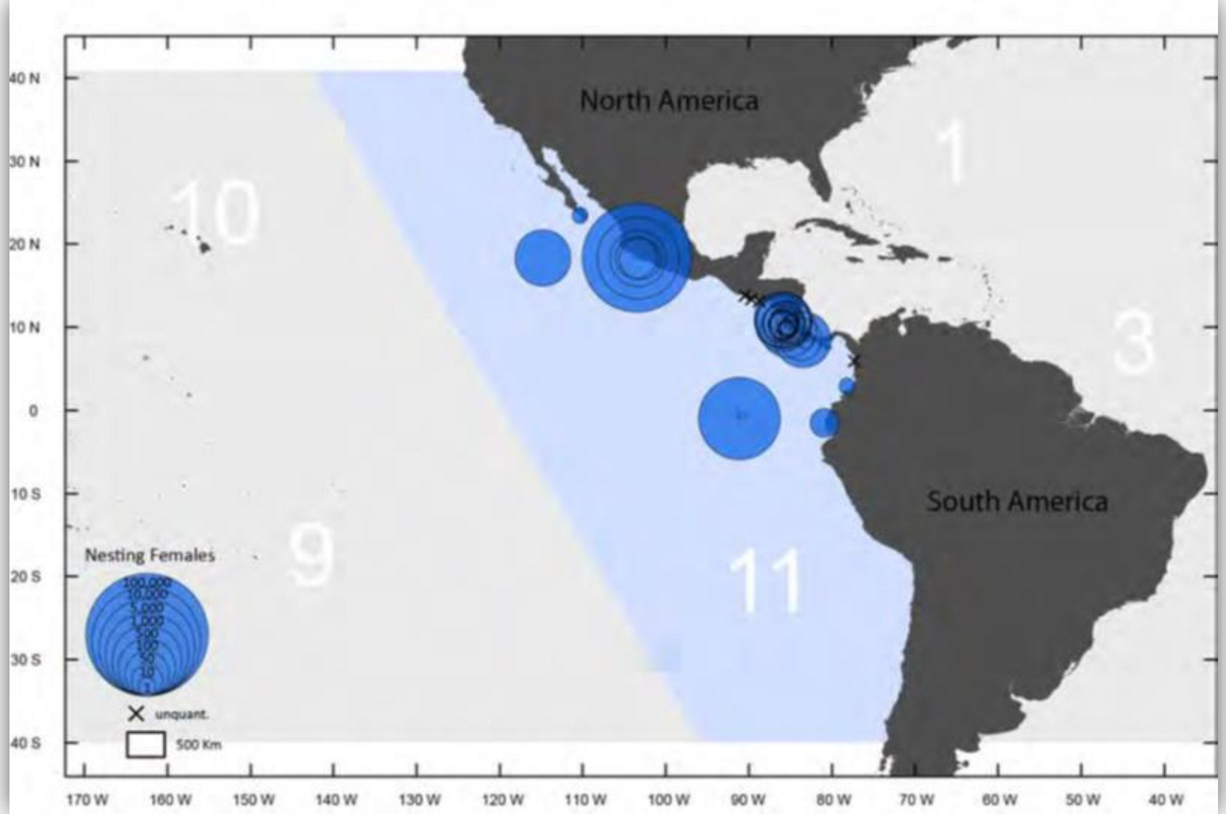

Figure 7. Nesting distribution of East Pacific green sea turtles (blue-shaded area marked with '11'). Size of circles indicates estimated nester abundance.

Status and Trends

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

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

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

Population Dynamics

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

Remigration intervals, numbers of nests per season, and number of eggs per clutch are within the ranges reported in the *General Population Dynamics* narrative for green turtles (subsection 3.3.4.1). Hatching success rate was 92% in Mexico (Xavier et al. 2006 as cited in Seminoff et al. 2015), although this is probably an overestimate because the study did not include nests that show no sign of hatching in these analyses.

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

Threats to the Species

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

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

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

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

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

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

Many United States fisheries capture green sea turtles in the Pacific. However, the Hawaii SSLL and DSLL fisheries do not overlap the Action Area. The Hawaii SSLL fishery interacted with ten green turtles (zero mortalities) from 2004-2018 (NMFS 2019b). Based on genetic samples from 19 green sea turtles captured since the 1990s, NMFS estimates that between 32 and 77% of the turtles caught in the Hawaii SSLL fishery are likely East Pacific green sea turtles (P. Dutton pers. comm. November 20, 2017). This means that between 3 and 8 turtles from this DPS have been captured by the Hawaii SSLL fishery since 2004.

Historically, the Hawaii DSLL fishery has observed 25 green sea turtles between 2004 and 2022 with an estimated 128 green sea turtle interactions with this fishery during this time frame (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021, 2022). NMFS estimates about 95% (95% CI = 77 to 99%) of green turtles captured in the DSLL fishery have been dead when they were hauled back to the fishing vessel based on the number of observed mortalities. Finally, the effects from the United States WCPO purse seine fishery will be discussed further in the Environmental Baseline as it overlaps the Action Area.

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

Central South Pacific Green Sea Turtle

Distribution and Population Structure

The Central South Pacific green sea turtle is distributed north from northern New Zealand to Fiji, Tuvalu, and Kiribati and east to include French Polynesia (Figure 8). Its open ocean polygonal boundary endpoints are (clockwise from the northwest-most extent): 9°N, 175°W to 9°N, 125°W to 40°S, 96°W to 40°S, 176°E, to 13°S, 171°E, and back to the 9°N, 175°W northwest extent. The range of this DPS includes a longitudinal expanse of 7,500 km—from Easter Island, Chile in the east to Fiji in the west, and encompasses American Samoa, French Polynesia, Cook Islands, Fiji, Kiribati, Tokelau, Tonga, and Tuvalu.

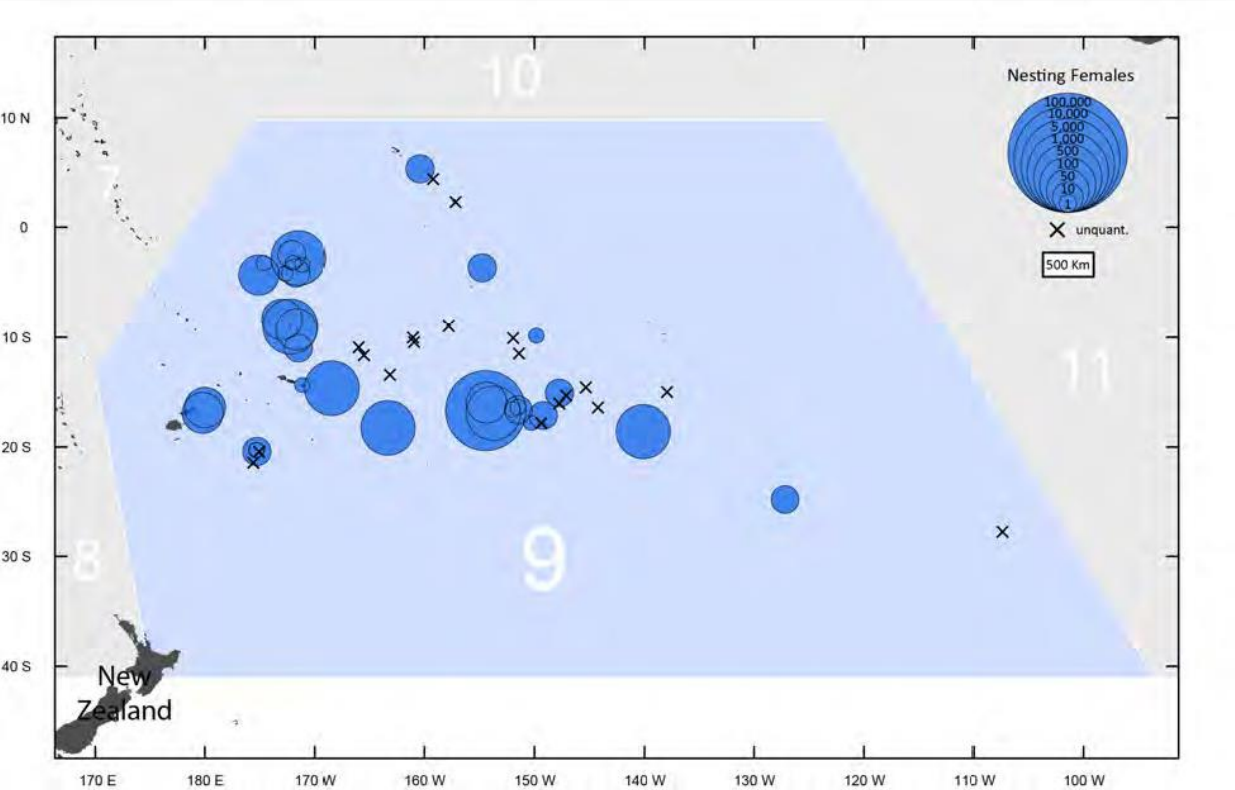

Figure 8. Nesting distribution of Central South Pacific green sea turtles. Size of circles indicates estimated nester abundance. Locations marked with 'X' indicate nesting sites lacking abundance information.

Nesting in this DPS of green turtle has been reported from 57 locations in the DPS' range (Figure 8). The genetic sampling and demographic studies of this population do not provide the data necessary to identify any populations or subpopulations in this green turtle. Foraging areas are known to include green sea turtles from multiple nesting locations. For example, in Fiji, Piovano et al. (2019) analyzed genetic samples from 150 green sea turtles captured in two foraging habitats at Yadua and Makogai Islands, and identified 18 haplotypes with turtles in these habitats originating from American Samoa (72%), New Caledonia (17%) and French Polynesia (7%).

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

Status and Trends

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

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

Threats to the Species

Central South Pacific Ocean green turtles are threatened by being harvested for food and by the loss and degradation of coastal habitat. The exploitation of green turtles for eggs, meat, and parts has occurred throughout the South Pacific Region including, but not limited to, American Samoa, Cook Islands, Fiji Islands, Tuvalu, Tokelau, French Polynesia, and Kiribati (Balazs 1983; Tuato'o-Bartley et al. 1993; NMFS and FWS 1998a; Obura and Stone 2002; Alefaio et al. 2006). At Scilly Atoll (French Polynesia), local residents are allowed to harvest 50 adults per year from a nesting population that might be as low as 300 to 400 turtles. In Tuvalu, harvest of turtles for meat has been reported as a major threat (Alefaio et al. 2006). Elsewhere, subsistence harvests of eggs occur regularly.

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

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

Many United States fisheries capture green sea turtles in the Pacific. However, the Hawaii SSLL and DSLL fisheries do not overlap the Action Area. The SSLL is expected to capture 5 green sea turtles (all species considered (*n*= 6), including the Central South Pacific) annually with one of those turtles expected to die (NMFS 2019b). Between 2004 and 2018, the SSLL fishery captured 10 green sea turtles and we expect less than 2% of those turtles caught could be from the Central South Pacific (NMFS 2019b). All ten green sea turtles caught in the Hawaii SSLL fishery between 2004 and 2018 were released alive with no gear attached.

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

The effects from the United States WCPO purse seine fishery will be discussed further in the Environmental Baseline as it overlaps the Action Area.

Southwest Pacific Green Sea Turtle

Distribution and Population Structure

The distribution of Southwest Pacific green sea turtles extends from the western boundary of Torres Strait (at 142°E longitude), southeast to the eastern tip of Papua New Guinea and out to the offshore coordinate of 13°S, 171°E; the eastern boundary runs from this point southeast to 40°S, 176°E; the southern boundary runs along 40°S from 142°E to 176°E; and the western boundary runs from 40°S, 142ºE north to Australian coast then follows the coast northward to Torres Strait (Figure 9).

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

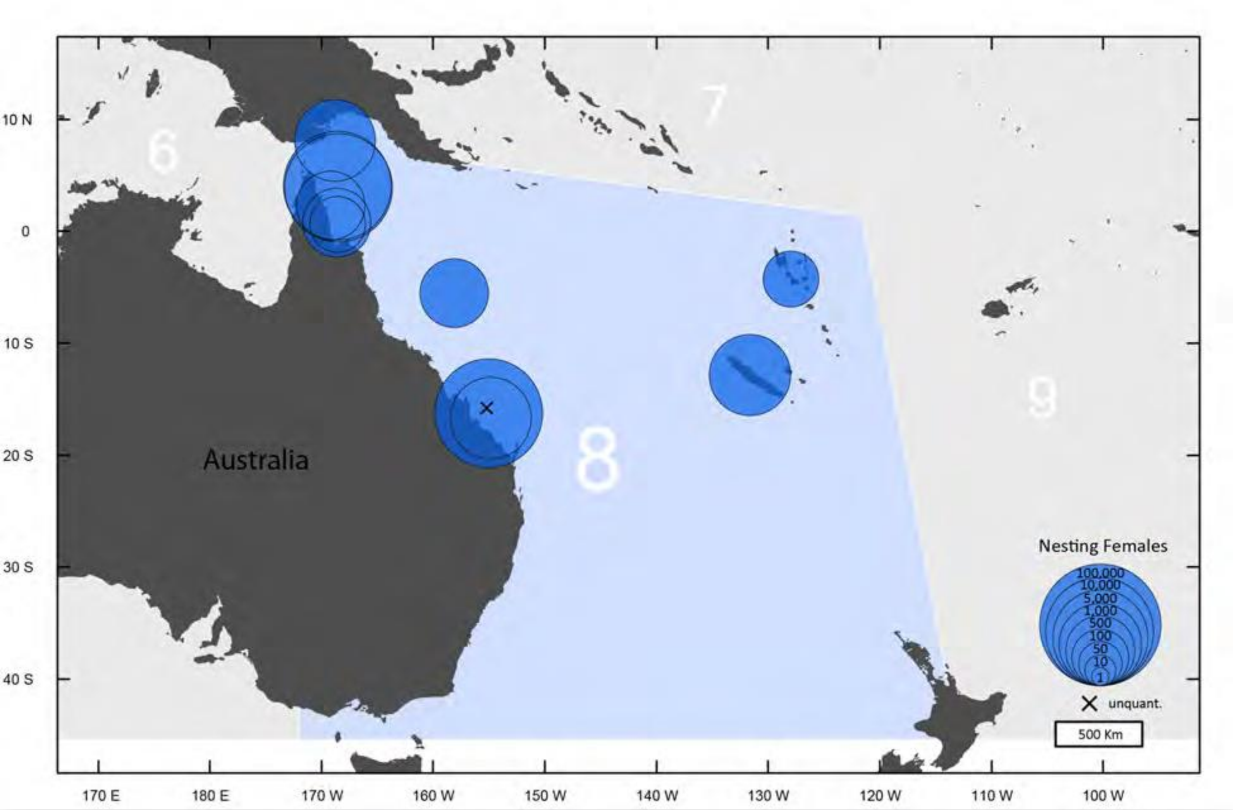

Figure 9. Nesting distribution of Southwest Pacific green sea turtles. Size of circles indicates nesting estimated nester abundance. Locations marked with 'x' indicate nesting sites lacking abundance information.

Status and Trends

Southwest Pacific green sea turtles are listed as threatened. The estimated abundance of nesting females in this population is high with 83,058 adult nesting females (Seminoff et al. 2015). The highest nesting densities for this green turtle occur in Australia, particularly in the northern Great Barrier Reef. Estimates of annual nesters at Raine Island, Australia, vary from 4,000 – 89,000 (Seminoff et al. 2004; NMFS and FWS 2007a; Chaloupka et al. 2008; Limpus 2009). The number of turtles nesting in the Great Barrier Reef area of Australia differs widely from year to year and is well correlated with an index of the Southern Oscillation (Limpus and Nicholls 2000 as cited in Seminoff et al. 2015). Fewer turtles nest in New Caledonia and Vanuatu. In New Caledonia, Pritchard (1994 as cited in Maison et al. 2010) described turtles to be abundant or near saturation levels on several islands in the archipelago. A 2006 and 2007 survey of over 6,000 km of nesting habitat identified nesting locations hosting an estimated 1,000 – 2,000 green turtle females nesting annually (Maison et al. 2010 citing Limpus et al. 2009). In Vanuatu, hundreds of nesting green turtles have been observed on Malekula Island, Southern Epi Island, Santo and Thion Islands, Tegua and Hiu Islands (Maison et al. 2010). We estimate total population size by using the age-based Leslie matrix model described above and the adult female population estimate of 83,058 (Seminoff et al. 2015). As we are reasonably certain the trend for this population is increasing, we restricted the 10,000 population simulations to those demonstrating an increasing population growth rate to estimate a total mean population size of 14,762,154 (range 10,137,513 to 23,139,940) individuals greater than one year old.

Population Dynamics

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

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

Threats to the Species

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

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

Many United States fisheries capture green sea turtles in the Pacific. However, only the United States WCPO purse seine fishery is expected to interact with this DPS within the Action Area. Therefore, the effects from this fishery will be discussed further in the Environmental Baseline. However, the Hawaii SSLL and DSLL fisheries do not overlap the Action Area.

The SSLL is expected to capture 5 green sea turtles (all species considered (*n*= 6), including the Southwest Pacific green sea turtle) annually with one of those turtles expected to die (NMFS 2019b). Between 2004 and 2018, the SSLL fishery captured 10 green sea turtles and we expect between 0 and 8% of those turtles caught could be from the Southwest Pacific green sea turtle DPS (NMFS 2019b). Therefore, the SSLL fishery may have captured one Southwest Pacific green sea turtle since 2004. In total, all ten green sea turtles caught in the Hawaii SSLL fishery between 2004 and 2018 were released alive with no gear attached, therefore mortality is not expected to occur from interactions in this fishery.

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

Central West Pacific Green Sea Turtle

Distribution and Population Structure

The Central West Pacific green sea turtle has as their northern boundary 41°N latitude and is bounded by 41°N, 169°E in the northeast corner, going southeast to 9°N, 175°W, then southwest to 13°S, 171°E, west and slightly north to the eastern tip of Papua New Guinea, along the northern shore of the Island of New Guinea to West Papua in Indonesia, northwest to 4.5°N, 129°E then to West Papua in Indonesia, then north to 41°N, 146°E (Figure 10).

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

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

Japan was reported to have nested in Yap (Cruce 2009).

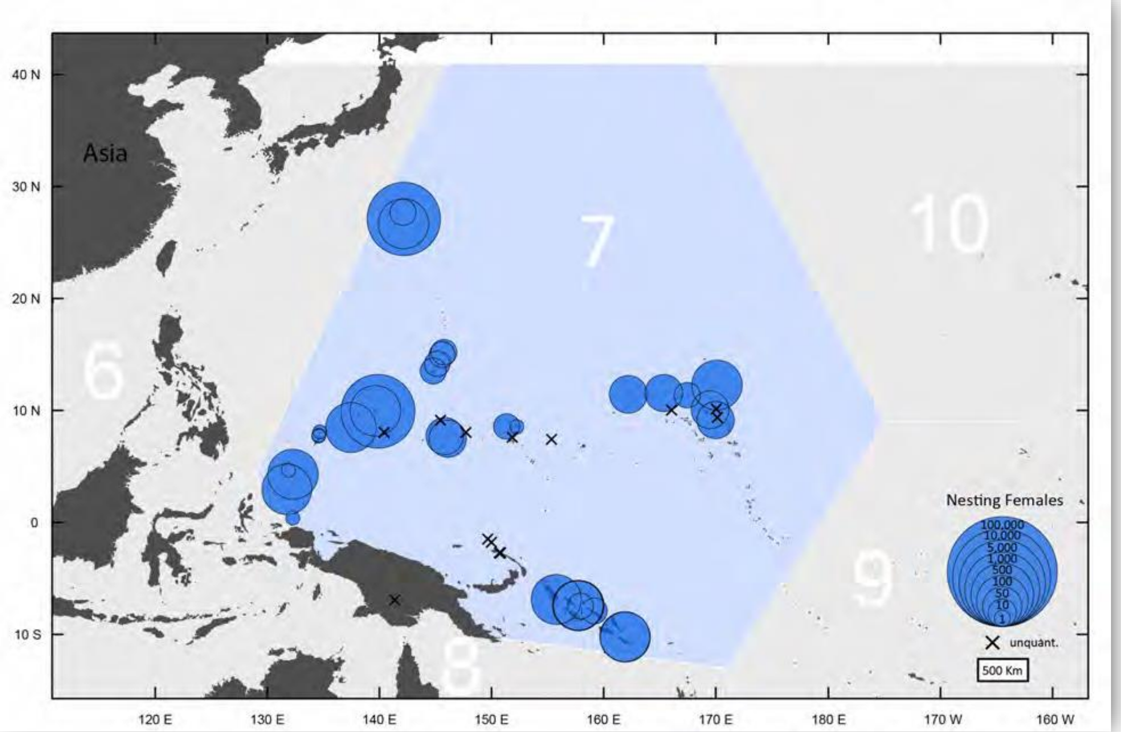

Figure 10. Nesting distribution of Central West Pacific green sea turtles (blue-shaded region). Size of circles indicates estimate nester abundance. Locations marked with 'x' indicate nesting sites lacking abundance information

Status and Trends

Central West Pacific green sea turtles are listed as endangered. The DPS exhibits low nesting abundance, with an estimated total nester abundance of 6,518 females at 51 documented nesting sites (Seminoff et al. 2015). There are a number of unquantified nesting sites, possibly with small numbers of nesting females; however, specifics regarding these sites is unknown. The largest nesting site is in the FSM, and that particular site hosts approximately 22% of the total annual nesting females. The limited available information suggests a nesting population decrease in some portions of the DPS like in the Marshall Islands, or unknown trends in other areas such as Palau, Papua New Guinea, the Marianas, Solomon Islands, or the FSM (Maison et al. 2010). There are a number of unquantified nesting sites, possibly with small numbers, although specifics regarding these sites are unknown. The largest nesting site is in the FSM, and that particular site hosts approximately 22% of the total annual nesting females (Seminoff et al. 2015). In their status review, Seminoff et al. (2015) conducted quantitative PVA modeling to project adult female abundance at nesting sites for which sufficient data were available, including a minimum of 15 years of nesting abundance data that is less than 10 years old. There is only one site for which 15 or more years of recent data are available for annual nester abundance for the Central West Pacific green sea turtle. This is at Chichijima, Japan, one of the major green sea turtle nesting concentrations in Japan (Horikoshi et al. 1994). The PVA of

Seminoff et al. (2015) has limitations, and does not fully incorporate other key elements critical to the decision making process such as spatial structure or threats. It assumes all environmental and anthropogenic pressures will remain constant in the forecast period and it relies on nesting data alone. The PVA suggests the probability approaches zero that this population will fall below the trend reference or absolute abundance reference in 100 years (Seminoff et al. 2015). The population has increased from a mean of approximately 100 females/year in the late 1970s/early 1980s to a mean of approximately 500 per year since 2000. Similarly, Chaloupka et al. (2008) reports an estimated annual population growth rate of 6.8% per year for the Chichijima nesting site.

With the exception of the small Chichijima, Japan nesting aggregation, the overall population trend for the Central West Pacific green sea turtle is not known and given the threats to the DPS, its status as endangered under the ESA and suspected declines in certain portions we assume the DPS has an overall declining trend.

We estimate total population size by using the age-based Leslie matrix model described above and the adult female population estimate of 6,518 (Seminoff et al. 2015). As the trend for this population is uncertain, we considered the mean of all 10,000 simulations to estimate a total mean population size of 1,543,625 (range 821,772 to 3,592,842) individuals greater than one year old.

Population Dynamics

The population dynamics of this DPS are similar to that of the other green turtles discussed in this opinion. However, clutch sizes appear to be more variable: clutch sizes range from 59 to 139 eggs per nest (compared with 80 to 120 eggs per nest in the general pattern; Palacios 2012 as cited in Seminoff et al. 2015). In addition, like East Pacific green turtles, these green turtles appear to mature at smaller sizes. The known mean nester sizes range from 102 cm curved carapace length (CCL) in Palau to 104.5 cm CCL in Tinian, CNMI (Pultz et al. 1999). Finally, hatching success for areas where this information is available is more variable: from 44.1-73.8% (Suganuma 1985).

Threats to the Species

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

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

Central West Pacific Ocean green turtles are threatened by being harvested for food, by the loss and degradation of coastal habitat, and by incidental capture in artisanal and commercial fisheries. Green sea turtle eggs are collected in the CNMI, Federated States of Micronesia, Guam, Kiribati, Papua, Papua New Guinea, Republic of the Marshall Islands, and Palau (Eckert

1993; Hitipeuw and Maturbongs 2002; Philip 2002; Humber et al. 2014). In addition, adult females were reported to have been captured and killed for food on nesting beaches in the Commonwealth of the Northern Marianna Islands, Papua New Guinea (Limpus et al. 2002) and the Solomon Islands (Broderick 1998; Pita and Broderick 2005).

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

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

Many United States fisheries capture green sea turtles in the Pacific. However, only the United States WCPO purse seine fishery is expected to interact with this DPS within the Action Area. Therefore, the effects from this fishery will be discussed further in the Environmental Baseline.

However, the Hawaii SSLL and DSLL fisheries do not overlap the Action Area. The SSLL is expected to capture 5 green sea turtles (all species considered (n= 6), including the Central West Pacific) annually with one of those turtles expected to die (NMFS 2019b). Between 2004 and 2018, the SSLL fishery captured 10 green sea turtles and we expect between 0 and 8% of those turtles caught could be from the Central West Pacific (NMFS 2019b). Therefore, the SSLL fishery may have captured one Central West Pacific green sea turtle since 2004. In total, all ten green sea turtles caught in the Hawaii SSLL fishery between 2004 and 2018 were released alive with no gear attached, therefore mortality is not expected to occur from interactions in this fishery.

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

East Indian-West Pacific Green Sea Turtle

Distribution and Population Structure

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

142°E longitude). The southern boundary is 40°S latitude, encompassing the Gulf of Carpentaria (Figure 11).

East Indian-West Pacific green sea turtles use nesting beaches from northern Australia to southern Asia (Figure 11). Based on studies of 22 rookeries, 16 populations or subpopulations of this DPS have been identified: Northwest Shelf, Scott Reef, Ashmore Reef, and the Gulf of Carpentaria (Australia); West Java, Berau Islands, and Aru (Indonesia); Peninsular Malaysia, Sarawak, Southeast Sabah (Malaysia), Sulu Sea (Malaysia/Philippines); Wan-an Island, and Lanyu Island (Taiwan); Zamami, Iriomote Island, and Ishigaki Island (Japan; Dethmers et al. 2006; Cheng et al. 2008; Hamabata et al. 2009; Nishizawa et al. 2011).

Pelagic habitat use by small juveniles and adults are not generally known; however, green sea turtles with haplotypes found in East Indian-West Pacific green sea turtle rookeries have interacted with the ASLL fishery (NMFS unpublished data) suggesting they migrate as far east as the American Samoa portion of the United States EEZ.

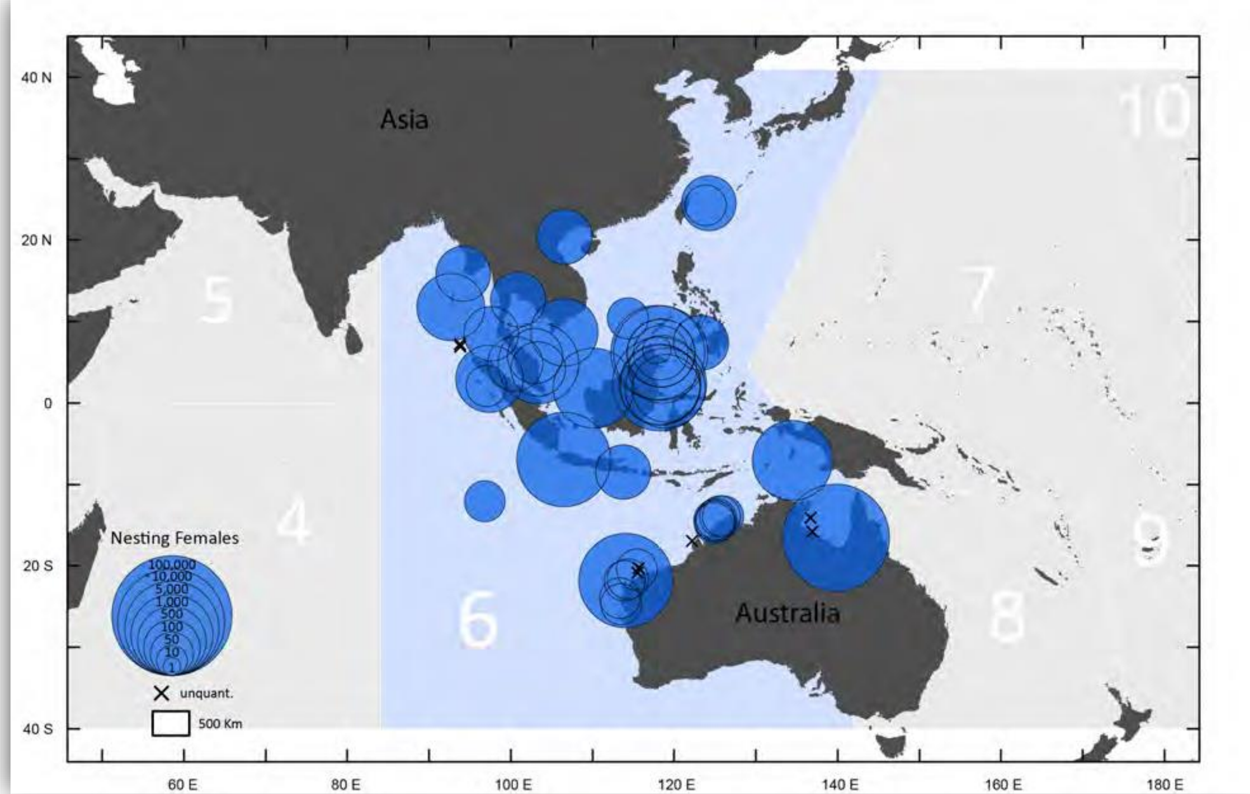

Figure 11. Nesting distribution of East Indian-West Pacific green sea turtle (blue-shaded area). Size of circles indicates estimated nester abundance. Locations marked with 'x' indicate nesting sites lacking abundance information.

Status and Trends

East Indian-West Pacific green sea turtles are listed as threatened. The abundance of nesting females in this DPS is estimated at 77,009 females over 50 nesting sites (Seminoff et al. 2015).

The largest nesting site is in northern Australia and supports about 25,000 nesting females; however, there are no recent data to support estimations of trend (Seminoff et al. 2015). The abundance of adult females at important nesting sites in Malaysia is estimated at 7,000 females and is expected to increase in the foreseeable future. However, the abundance of nesting females at several smaller sites are expected to decrease (Seminoff et al. 2015). Because of these declines, several nesting populations of this DPS have relatively high probabilities of falling below 100 nesting females per year (Seminoff et al. 2015). Mazaris et al. (2017) conducted a meta-analysis of population trends on sea turtles nesting beaches and they summarize trends for 14 East Indian-West Pacific green sea turtle nesting beaches. Restricting these trends to those that concluded within the last 20 years (i.e. 2002) and consisting of at least 10 years of monitoring data, we estimate a mean population decline of 1.11% per year (95% CI: -1.6% to -0.6%).

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

Threats to the Species

Fibropapillomatosis has been found in green sea turtles in Indonesia (Adnyana et al. 1997), Japan (Y. Matsuzawa, Japanese Sea Turtle Association, pers. comm. 2004 as cited in Seminoff et al. 2015), the Philippines (Nalo-Ochona 2000 as cited in Seminoff et al. 2015), Western Australia (Raidal and Prince 1996; Aguirre and Lutz 2004), and on PhuQuoc in Vietnam (Ministry of Fisheries 2003). Epidemiological studies indicate rising incidence of this disease (George 1997), thus the above list will likely grow in the future.

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

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

Many United States fisheries capture green sea turtles in the Pacific. However, the Hawaii SSLL and DSLL fisheries do not overlap the Action Area. However, only the United States WCPO purse seine fishery is expected to interact with this DPS within the Action Area. Therefore, the effects from this fishery will be discussed further in the Environmental Baseline.

The SSLL is expected to capture 5 green sea turtles (all species considered (n= 6), including the East Indian-West Pacific green sea turtle) annually with one of those turtles expected to die (NMFS 2019b). Between 2004 and 2018, the SSLL fishery captured 10 green sea turtles and we expect between 0 and 10% of those turtles caught could be from the East Indian-West Pacific green sea turtle DPS (NMFS 2019b). Therefore, the SSLL fishery may have captured one East

Indian-West Pacific green sea turtle since 2004. In total, all ten green sea turtles caught in the Hawaii SSLL fishery between 2004 and 2018 were released alive with no gear attached, therefore mortality is not expected to occur from interactions in this fishery.

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

Summary of the Status

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

As previously mentioned in the green sea turtle status discussions, sea turtles face various threats throughout each stage of their respective life cycles.

In our analysis of each green sea turtle DPS, we concluded most of these DPSs face the same general threats. For instance, hatchling predation is encountered by all green sea turtles; however, the predatory species may change depending on geographic location. Those dynamics are consistent with the status and trend of the count data: most green sea turtle DPSs appear to be increasing in abundance.

However, as previously noted, trends are positive in certain portions of the Central West Pacific green sea turtle while others are unknown. Similarly, trends are not apparent in the Central South Pacific green sea turtles.

2.3.2 Leatherback Sea Turtle

Distribution and Population Structure

Leatherback sea turtles are listed as endangered under the ESA throughout their global range. The leatherback turtle has the most extensive global distribution of any reptile and is distributed throughout the oceans of the world (Figure 12) from the equator to subpolar regions in both hemispheres. Leatherback turtles spend the majority of their lives at sea, where they develop, forage, migrate, and mate, nesting on beaches on every continent except Europe and Antarctica, and several islands of the Caribbean and the Indo-Pacific (Eckert et al. 2012; NMFS and FWS 2020a). Seven populations are currently recognized: (1) Northwest Atlantic; (2) Southeast Atlantic; (3) Southwest Atlantic; (4) Northeast Indian; (5) Southwest Indian; (6) West Pacific; and (7) East Pacific Ocean populations (NMFS and FWS 2020a).

For purposes of this opinion, we focus only on the West Pacific leatherback sea turtle population. Leatherback sea turtles from the East Pacific Ocean population are not considered likely to be captured in the ASLL fishery because all of the observed interactions occurred west of 162º W and East Pacific leatherback sea turtles use habitats in the South Pacific Gyre east of 140 º W that do not overlap with the Action Area (Figure 13; Willis-Norton et al. 2015; NMFS 2017b).

Therefore, our analysis focuses on the effects of the action on the West Pacific Ocean population of leatherback sea turtles. The Southwest Indian Ocean population is unlikely to occur in the Action Area because nesting females from this population generally either stay in the Indian Ocean or move around the Cape of Good Hope and into the Southeast Atlantic (NMFS and FWS 2020). In addition, based on genetic samples from leatherback sea turtles captured in the ASLL fishery, of those sampled, all were confirmed to be from the western Pacific regional population comprised of Papua Barat, Indonesia, PNG, and the Solomon Islands. (P. Dutton pers. comm. July 5, 2018).

The marine distribution for Pacific leatherback sea turtles extends north into the Sea of Japan, northeast and east across the North Pacific to the west coast of North America (predominantly off California), west to the South China Sea and Indonesian Seas, and south into the high latitude waters of the western South Pacific Ocean and Tasman Sea (Figure 13; Benson et al. 2011).

Figure 12. Map identifying the range of the endangered leatherback sea turtle. From NMFS https://www.fisheries.noaa.gov/species/leatherback-turtle#overview

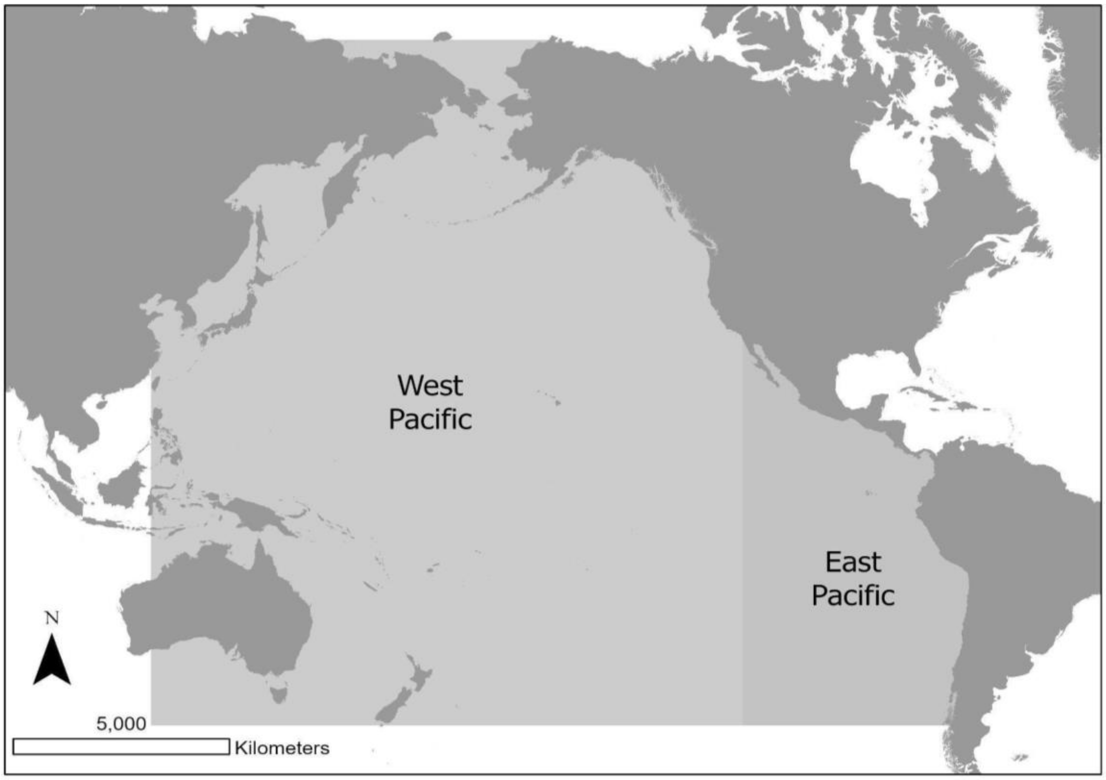

Figure 13. Map identifying the ranges of the East and West Pacific populations of leatherbacks.

We define the West Pacific population as leatherback turtles originating from the West Pacific Ocean, with the following boundaries: south of 71° N, north of 47° S latitudes and east of 120° E, and west of 117.124° W longitudes (NMFS and FWS 2020a). Indonesia, Papua New Guinea, and Solomon Islands have been identified as the core nesting areas for this population (Figure 14; Benson et al. 2007a; Benson et al. 2007b; Benson et al. 2011; Benson 2018). Low levels of nesting are also reported in Vanuatu and the Philippines (NMFS and FWS 2020a). Recently, a new leatherback turtle nesting area was identified at Buru Island, Maluku province of Indonesia where approximately 200 nests are laid annually (NMFS and FWS 2020a; WWF 2018, 2022). However, long-term monitoring data for this population is geographically limited to the Bird's Head Peninsula in West Papua at Jamursba Medi and Wermon nesting beaches which represent an estimated 50% to 75% of all nesting in the West Pacific (NMFS and FWS 2020a).

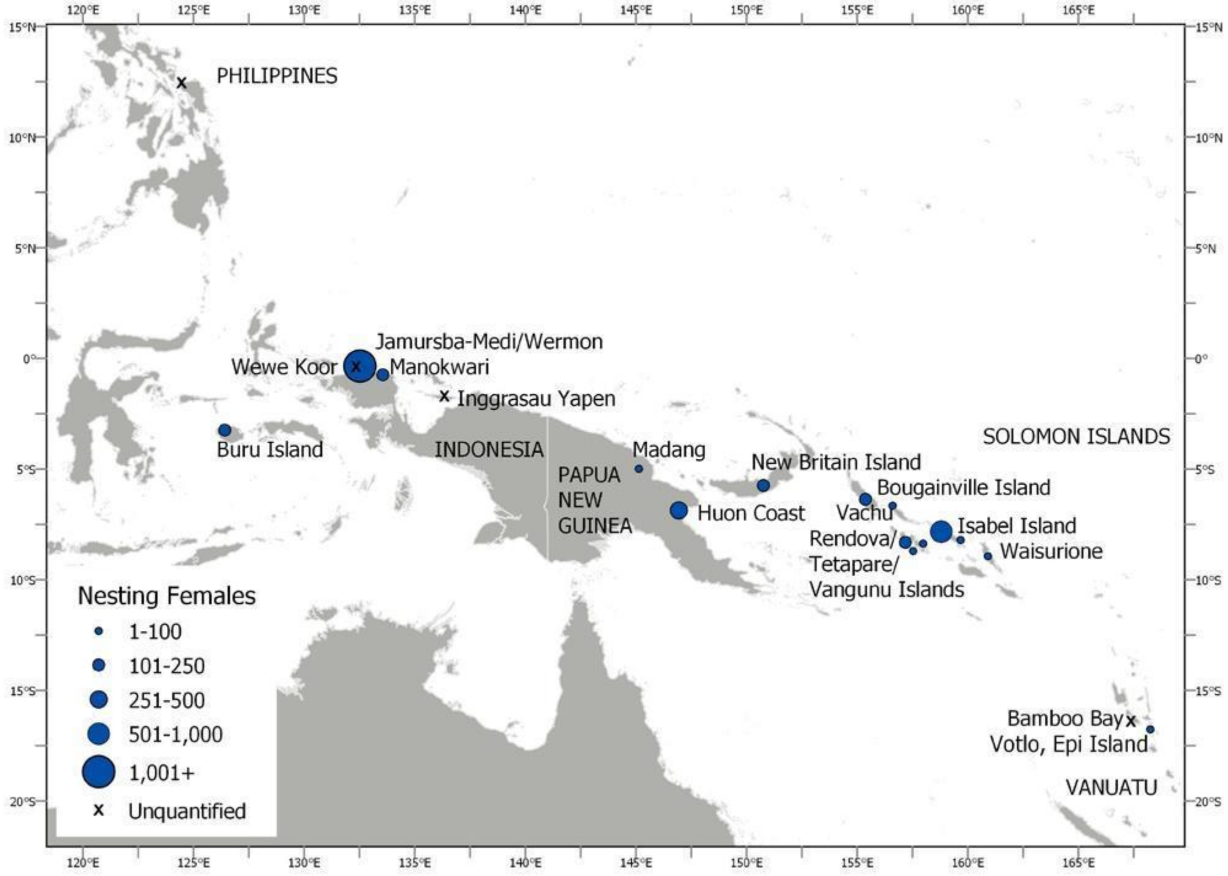

Figure 14. Nesting sites of the West Pacific leatherback population (NMFS and FWS 2020a).

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

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

Migration and foraging strategies vary based on these life history strategies, likely due to prevailing offshore currents and, for hatchlings, seasonal monsoon-related effects experienced as hatchlings (Benson et al. 2011; Gaspar et al. 2012). Summer nesting females forage in Northern Hemisphere foraging habitats in Asia and the North Pacific Ocean, while winter nesting females migrate to tropical waters of the Southern Hemisphere in the South Pacific Ocean (Figure 15; Benson et al. 2011; Harrison et al. 2018). The lack of crossover among seasonal nesting populations suggests that leatherback turtles develop fidelity for specific foraging regions likely based on juvenile dispersal patterns (Benson et al. 2011; Gaspar et al. 2012; Gaspar and Lalire 2017). Stable isotopes, linked to particular foraging regions, confirm nesting season fidelity to specific foraging regions (Seminoff et al. 2012).

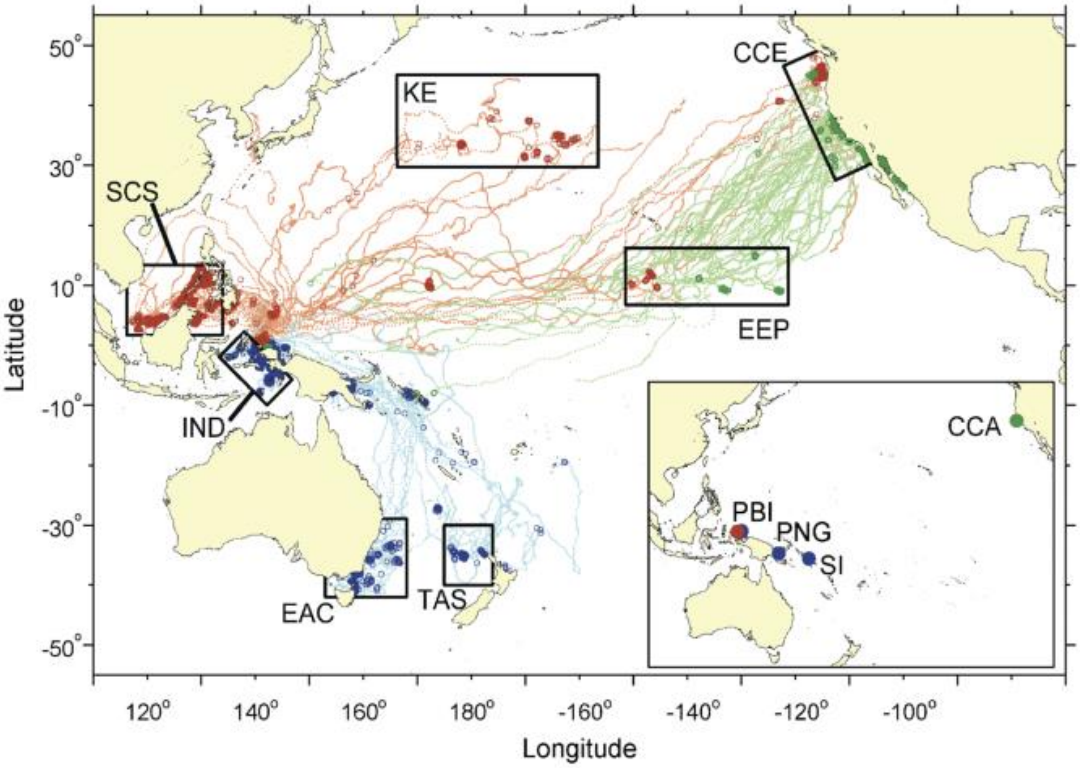

Figure 15. Satellite tracks from 126 West Pacific leatherback sea turtles. Color of track indicates deployment season: red = summer nesters, blue = winter nesters, green = deployments at central California foraging grounds. Inset shows deployment locations; PBI = Papua Barat, Indonesia, PNG = Papua New Guinea, SI = Solomon Islands, CCA = central California. Black boxes represent ecoregions for which habitat associations were quantitatively examined: SCS = South China, Sulu and Sulawesi Seas, IND = Indonesian Seas, EAC = East Australia Current Extension, TAS = Tasman Front, KE = Kuroshio Extension, EEP = equatorial eastern Pacific, and CCE = California Current Ecosystem (from Benson et al. 2011; NMFS and FWS 2020a).

Status and Trends

Leatherback sea turtles are listed as endangered under the ESA throughout their global range. Tiwari et al. (2013) estimated that the western Pacific population had 1,438 mature individuals, which meets the IUCN threshold for endangered. However, because their estimates led them to conclude that this population had a high probability of declining by about 96% by 2040 (to about 57 nests and 260 adult females), under their classification system, the IUCN classified the western Pacific leatherback turtle population as critically endangered (Tiwari et al. 2013; Wallace et al. 2013b).

Tapilatu et al. (2013) found a 78% decline in nesting from 1984 to 2011 at Jamursba Medi and a 62.8% decline in nesting at Wermon from 2002 to 2011. Overall they estimated a 5.9% per year decline in nesting abundance for both index beaches over this time period. Since the nesting decline reported in Tapilatu et al (2013), leatherback nest numbers have fluctuated between 477 and 1531 at Jamursba Medi and between 602 and 1566 at Wermon in the time period from April 2012-March 2013 to April 2021-March 2022 (Lontoh et al. In Prep). While the population does not demonstrate an increasing trend, it is encouraging that the population has not declined further in the past decade (Lontoh et al. In Prep; Figure 16). At a mean of 5.5 nests per female (Tapilatu et al. 2013, Lontoh 2014), these nest numbers equate to approximately 87 to 278 annual nesting females at Jamursba Medi and 109 to 285 annual nesting females at Wermon.

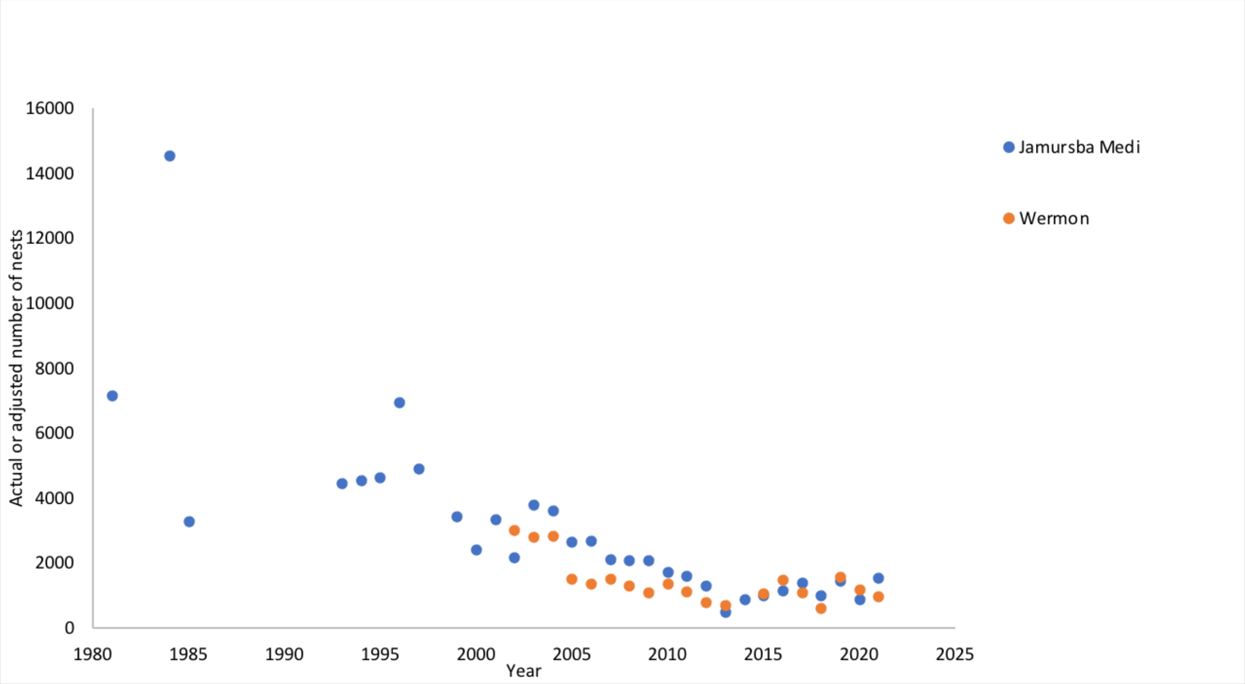

Figure 16. Actual or adjusted number of leatherback nests between 1981 and 2021 at Jamursba Medi and Wermon. Each year represents nests laid from April of one year to March of the following year (Lontoh et al. In Prep).

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

2017 and 2022 appear to be stable (Figure 16), we consider the 2017 abundance to be the best estimate of current (2022) adult females for the index beaches.

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

Additional but lower levels of nesting have been documented elsewhere in Indonesia, including a new monitoring program established in 2017 on Buru Island (World Wildlife Fund [WWF] 2022), plus locations in Papua New Guinea, Solomon Islands, Vanuatu and the Philippines (NMFS and FWS 2020a). Monitoring at most of these additional sites has not been going on long enough to establish trends or abundance; therefore data from those nesting beaches cannot be used to reliably calculate those metrics at this time. An exception to this is the WWF program at Buru Island in Indonesia where data have been consistently collected since 2017 (WWF 2022). While it's only 6 years of data, this does span almost two remigration intervals (3.1 years; Martin et al. 2020a) for leatherbacks. These data indicate an increasing trend of 10.1% per year (Figure 17; 95% Confidence Interval: -26.1% to 46.3%) based on an exponential growth curve.

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

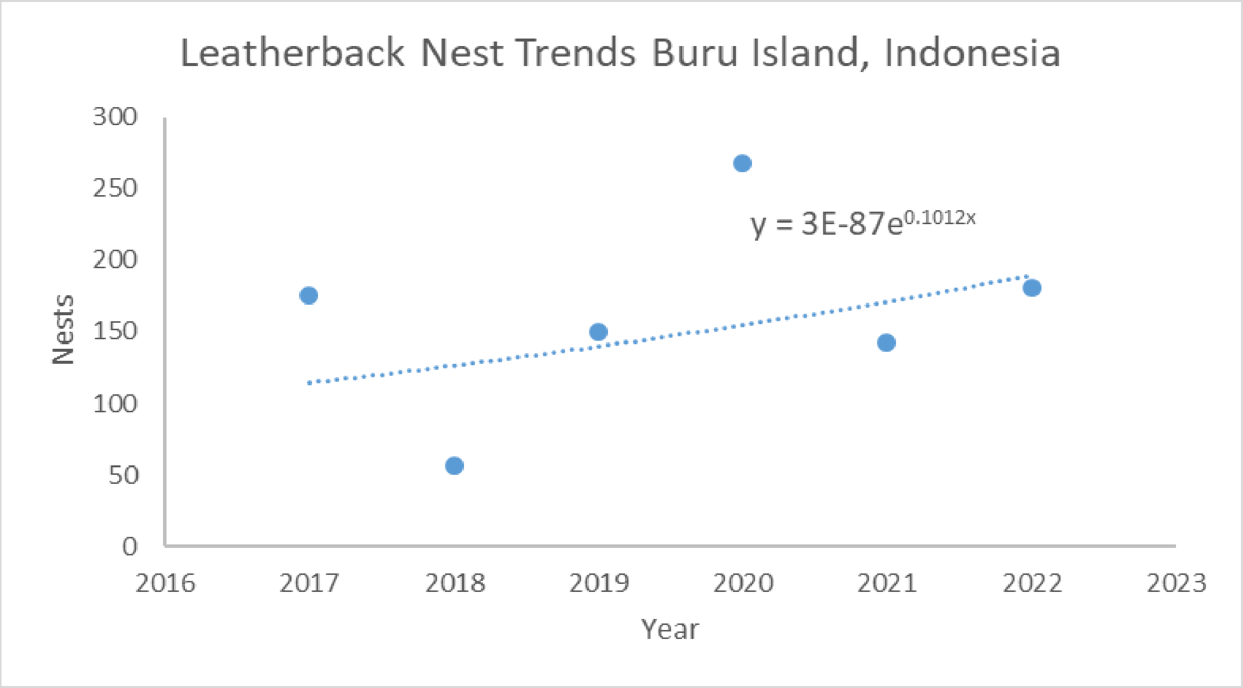

Figure 17. Leatherback nesting trend at Buru Island, data from WWF (2022).

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

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

The median trend in annual nest counts estimated for Jamursba Medi nesting beaches from data collected from 2001-2017 was -5.7 percent annually (95% CI: -16.2% to 5.3%; NMFS and FWS 2020a). The median trend in annual nest counts estimated for Wermon nesting beaches from data collected from 2006 to 2017 (excluding 2013–2015 due to low or insufficient effort) was -2.3 percent annually (95% CI: -19.8 to 14.9%; NMFS and FWS 2020a). Using their Bayesian steady-state model, Martin et al. (2020a) estimate the combined trends for Jamursba Medi and Wermon to be a mean of -6.0% (95% CI: -24.1 to 12.2%) annually. We note that the data from 2018 to 2021 are preliminary and only provided to NMFS from the authors (Lontoh et al. In Prep) as the graph shown in Figure 16. Until we receive the detailed raw monthly data from the nesting beaches, the growth trend analysis of Martin et al. (2020a) cannot be updated, so we do not have any updated modeled estimates of future growth rates based on this new information and we rely on the estimates of Martin et al. (2020a) for population growth rates after 2021. In addition, given the substantial declines in the population from 1984 to 2012, the data from 2012 to 2021 are likely not of long-enough duration to definitely state that the population is now at least stable, but we cannot be certain of this until the raw data are available for analysis.

Benson et al. (2020) assessed in-water trends for leatherbacks using the California Current Ecosystem foraging area (see Figure 15). They found that abundance for the total population is declining by a mean of 5.6% per year (95% CI: -9.8% to -1.5%) over a similar timeframe (1990 to 2017). This indicates the 6.0% per year decline found by Martin et al. (2020a) on the index nesting beaches may be mirrored in the large juvenile and male portions of the mixed North Pacific foraging population. Therefore, we consider the Martin et al. (2020a) trends in annual nest counts an accurate index of the population's growth rate through 2017.

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

A recent report on sea turtle bycatch in New Zealand's shallow-set longline fishery indicates a marked increase in leatherback captures (Figure 18; Dunn et al. 2022). Observed captures have increased from a low of one in 2008 to a high of 50 in 2022. Accounting for effort, the resulting trend in CPUE shows a statistically significant positive trend, increasing at a rate of 19.9% per year (Figure 18; 95% Confidence Interval: 8.4% to 31.3%) based on an exponential growth curve. While fishery captures can be influenced by numerous environmental factors that can disconnect them from population trends, the strength of this trend suggests the potential for more leatherback turtles in the water in recent years, which is supportive of the information we have on nesting beach trends (WWF 2022, Lontoh et al. In Prep) and increased hatchling production (Pakiding et al. 2020) for West Pacific leatherbacks. In summary, we acknowledge that there is a great deal of uncertainty associated with the current status of West Pacific leatherback sea turtles, and assuming a long term decline of 6% per year may underestimate future projected population sizes if the recent nesting beach trends continue. However, until new nesting beach data are available in sufficient detail to reassess the population trend model, we consider the projected declines of Martin et al. (2020a) to be the best available data but acknowledge that these declines did not occur from 2017 to 2021.

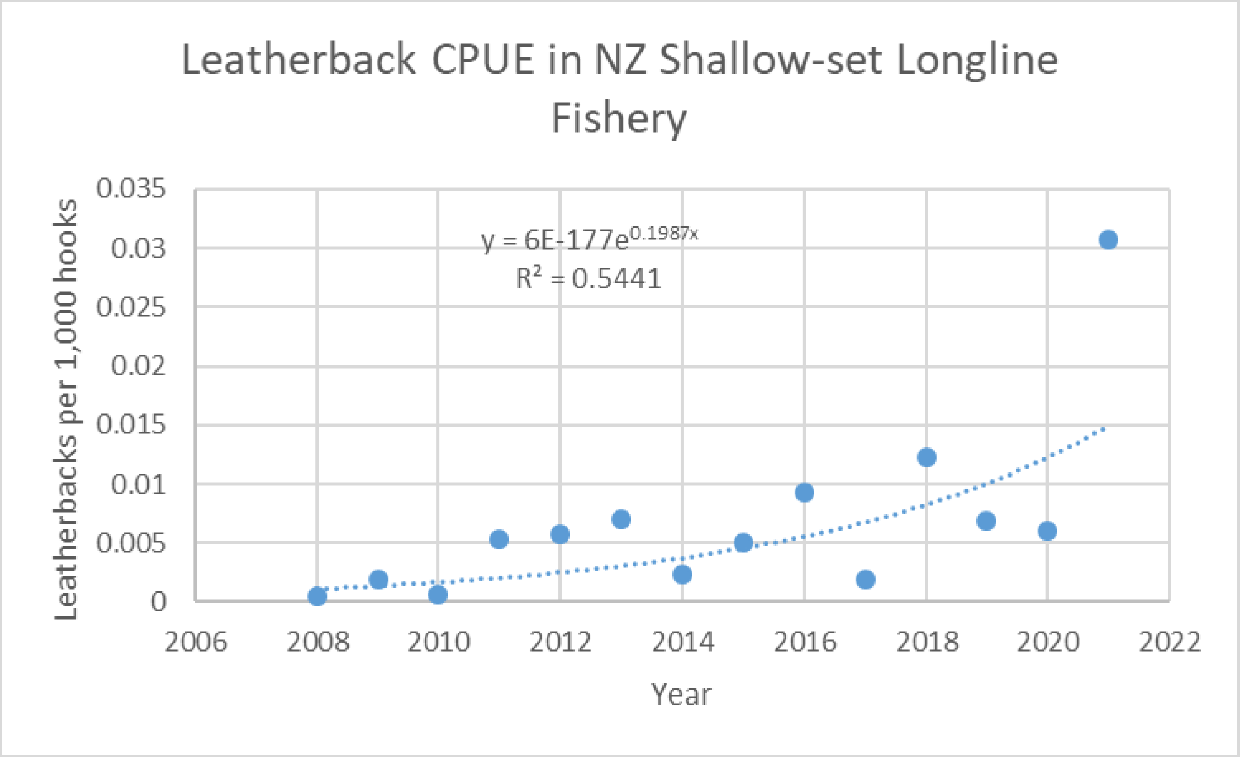

Figure 18. Trend in CPUE of leatherback sea turtles in New Zealand shallow-set longline fishery (data from Dunn et al. 2022).

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

Population Dynamics

Avens et al. (2020) estimate a mean age at maturity of 17 years (range of 12 to 28 years) for Pacific leatherbacks (East and West populations combined). In Indonesia, the mean size of nesting females is 161 cm curved carapace length (CCL) with an observed minimum of 138 cm CCL (Hitipeuw and Maturbongs 2002; Lontoh 2014). The most comprehensive information on clutch frequencies for West Pacific leatherback nesting females is from Jamursba Medi and Wermon beaches (Indonesia), where Tapilatu et al. (2013) estimated an average clutch frequency of 5.5 (+/- 1.6, range 3-10). For the overall West Pacific population, Lontoh (2014) found evidence that clutch frequency and remigration interval may vary with foraging life history. Mean clutch frequency varied among individuals, but was higher for the North Pacific Transition

Zone foraging group (6.1) than the Northeast Pacific (5.6) and South China Sea (4.8) foraging groups and increased with turtle size (curved carapace width) (Lontoh 2014). The range of remigration intervals was 1-6 years, and the mean (± SE) remigration interval was greater for the Northeast Pacific foraging group (4 ± 0.3 years) than the North Pacific Transition Zone foraging group (2.3 ± 0.4 years) and the South China Sea foraging group (2.9 ± 0.4 years) (Lontoh 2014). Mean clutch sizes for West Pacific leatherbacks range from 76 to 94 eggs per nest (Jino et al. 2018; Pilcher 2011; Tapilatu and Tiwari 2007).

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

Diving and Social Behavior

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

Threats to the Species

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

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

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

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

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

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

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

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

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

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

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

The 2004 management measures introduced to the Hawaii longline fisheries have demonstrably reduced leatherback sea turtle interaction rates by 83% (Gilman et al. 2007a; WPRFMC 2009b). Since the Hawaii SSLL fishery re-opened in 2004, 22 leatherback sea turtles have been estimated to have been killed in the fishery. All of the leatherback sea turtles caught were released alive;

mortality estimates come from applying the NMFS post-hooking mortality criteria to interactions (Ryder et al. 2006). NMFS authorized the incidental take of 21 leatherback sea turtles annually with an expectation that 3 would die (NMFS 2019c).

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

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

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

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

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

Conservation

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

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

Community-based conservation projects at Wermon and Jamursba Medi beaches on the Birdshead Peninsula of Indonesia, Buru and Kei Islands within the Maluku Province of Indonesia, Isabel Island of the Solomon Islands, and Epi and Ambrym Islands of Vanuatu have been developed that monitor nesting activity and protect nests and females from harvest and predation. These projects have increased our understanding of population nesting trends, and work to bolster the production of hatchlings from nesting areas and reduce harvest. Specifically, a nest monitoring project has been conducted in the Buru Islands since 2017. In 2017, 59% of leatherback nests were poached and nesting females were known to be harvested. By the 2021/2022 season, only one nest out of 181 was poached with no direct harvest of nesting females observed (WWF 2022). The Kei Island project works to reduce the harvest of subadult and adult leatherback turtles foraging within waters of the Kei Islands. In this project the direct harvest of foraging leatherbacks has declined by an average of 85.8% since 2017 (PIFSC unpublished data). Conservation efforts at Jamursba Medi and Wermon beaches have continued and expanded (Pakiding et al. 2020).

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

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

Summary of the Status

In this section of this biological opinion, we explained that the leatherback sea turtle, which is endangered throughout its range, is a global metapopulation represented by 7 extant populations. We anticipate that the ASLL fishery will only interact with the West Pacific Ocean population. The West Pacific Ocean leatherback sea turtle was estimated to have declined by as much as 78.3% from 1984 to 2011 (Tapilatu et al. 2013) and was estimated to be declining at a rate of approximately 6% per year through 2017 (Martin et al. 2020b). Martin et al. (2020a) estimates that there were approximately 790 adult females in 2017 at the two primary nesting beaches in Indonesia, which represent approximately 75% of the total population.

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

2.3.3 Olive Ridley Sea Turtle

Distribution and Population Structure

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

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

Population structure and genetics are poorly understood for this species, but populations occur in at least the eastern Pacific, western Pacific, eastern Indian Ocean, central Indian Ocean, western

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

Olive ridley sea turtles that have interacted with the ASLL fishery have haplotypes consistent with both eastern Pacific (40%) and western Pacific (60%) olive ridley sea turtles (NMFS 2015a; Dutton and LaCasella 2018) suggesting that sea turtles from both populations migrate as far as the American Samoa EEZ. The Eastern Pacific haplotypes cannot be distinguished between the endangered Mexico population and the East Pacific threatened population, therefore we assume that turtles from both populations may interact with the ASLL.

Status and Trends

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

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

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

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

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

Olive ridley sea turtle nesting has been monitored at Alas Purwo National Park, East Java, Indonesia since 1983 where they have increased from less than 10 annual nesters to more than 1,100 in 2018 (Kurniawan and Gitayana 2020; Figure 19). We digitized Figure 1 from (Kurniawan and Gitayana 2020; Figure 19 below) and estimate the trend at 9.8% per year (95% CI: 8.1 to 11.6%). Assuming a 2 year remigration interval (van Buskirk and Crowder 1994) and 50% sex ratio (Maulany et al. 2017), the total adults for this location is approximately 4,400. On Hamadi beach, Jayapura Bay in June 1999, an estimated several hundred ridleys were observed nesting (NMFS and FWS 2014). On Buru Island, Indonesia leatherback and olive ridley sea turtles have recently been discovered nesting, with 211 olive ridley sea turtle nests identified in 2017 (WWF 2018).

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

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

Heppell et al. (2005), excluding 0 to 1 year old hatchlings, 7.7% of the total population are breeding adults. Based on the 4.4% we estimate a total population size of 554,545, and of those, 316,883 are juveniles and adults older than 1 year old. Therefore, eliminating 0 to 1 year old hatchlings as this size class is not captured in the ASLL, we estimate the population size to be approximately 316,883 juveniles and adults based on an adult population size of 24,400 with 7.7% of the population occurring in that stage (24,400/[7.7/100]).

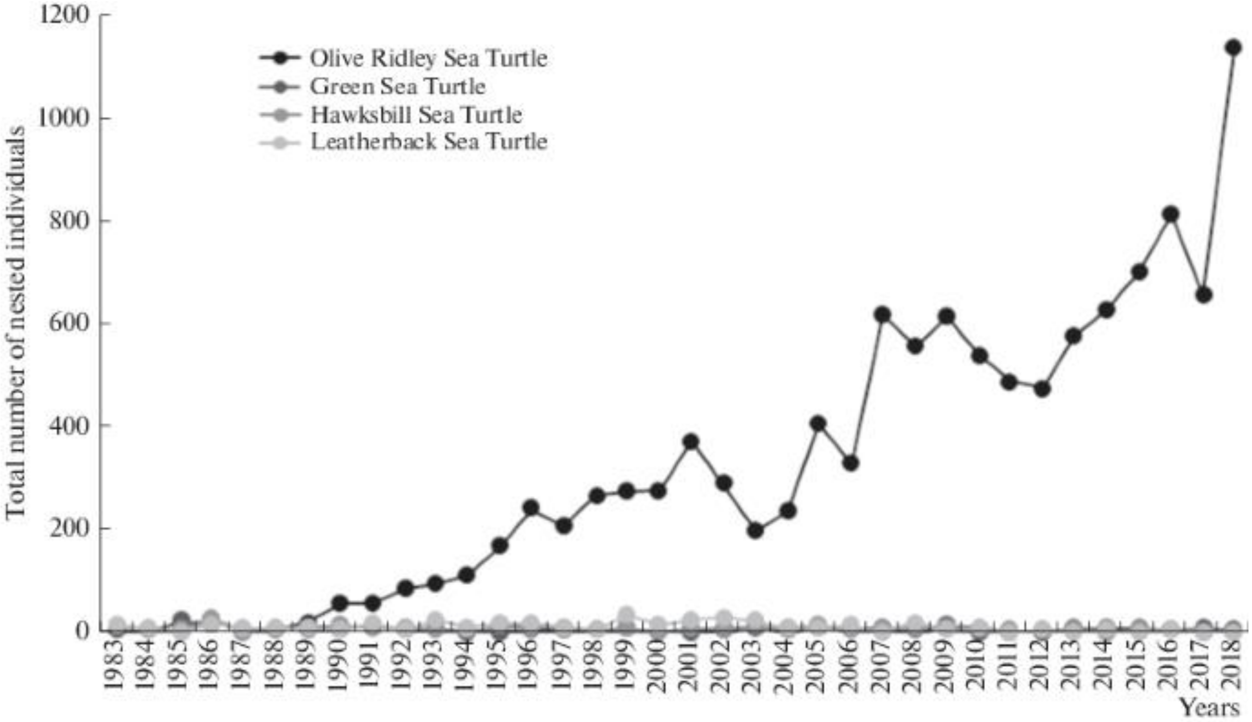

Figure 19. Trends in annual nesting sea turtles at Alas Purwo National Park, East Java, Indonesia (Kurniawan and Gitayana 2020).

Population Dynamics

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

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

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

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

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

Diving and Social Behavior

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

Threats to the Species

Impacts from climate change, especially due to global warming, are likely to become more apparent in future years (IPCC 2022; NMFS and FWS 2014). Based on the available information, climate change will affect olive ridley habitat and biology. Based on substantial new evidence, observed changes in marine systems are associated with rising water temperatures, as well as related changes in ice cover, salinity, oxygen levels, and circulation.

These changes include shifts in ranges and changes in algal, plankton, and fish abundance (IPCC 2007), which could affect olive ridley prey distribution and abundance.

However, olive ridleys in the east Pacific Ocean are highly vagile, and seemingly adaptable to fluctuating environmental conditions. They possess the ability to shift from an unproductive habitat to one where the waters are biologically productive, which may minimize the impacts of climate change (Plotkin 1994, 2010). As with leatherback turtles nesting in the eastern Pacific, olive ridley sea turtles may also be affected by the occurrence of El Nino events. It is possible that the variation in numbers of turtles in the Ostional arribadas are also affected by changes in productivity in their foraging areas, because olive ridley females also need time to amass sufficient nutrients to support their metabolic, migratory, and reproductive activities (Valverde et al. 2012).

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

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

Historically, an estimated 10 million olive ridleys inhabited the waters in the eastern Pacific off Mexico (Cliffton et al. 1982 as cited in NMFS and FWS 1998d; NMFS and FWS 2014). However, human-induced mortality caused this population to decline. From the 1960s to the 1970s, several million adult olive ridleys were harvested by Mexico for commercial trade with Europe and Japan (NMFS and FWS 1998d). Although olive ridley meat is palatable, it was not widely sought after; its eggs, however, are considered a delicacy. Fisheries for olive ridley turtles were also established in Ecuador during the 1960s and 1970s to supply Europe with leather (Green and Ortiz-Crespo 1982).

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

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

Between 2004 and 2022 there were 222 observed olive ridley interactions in the Hawaii DSLL fishery. When these reports are adjusted to account for the percentage of observer coverage, about 1,104 olive ridley turtles are likely to have been captured (McCracken and Cooper 2022). Based on the genetic samples, 75% are from either the endangered Mexico population or threatened subpopulation in the eastern Pacific, and 25% are from the threatened western Pacific subpopulation of olive ridley sea turtles. The Hawaii SSLL fishery rarely interacts with olive ridley sea turtles and since 2004 they have caught 10 and all were released alive, with no mortality (NMFS 2019b). The California Oregon drift gillnet fishery has an ITS for up to 2 anticipated olive ridley interactions and 1 anticipated estimated mortality every five years (NMFS 2013). Since 2001 no olive ridley sea turtles have been captured in the California Oregon drift gillnet fishery and only one has been observed since 1990 (NMFS unpublished data).

Conservation

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

Summary of the Status of the Olive Ridley Sea Turtle

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

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

2.3.4 Hawksbill sea turtle

Distribution and Population Structure

Hawksbill sea turtles are globally distributed occurring in at least the insular and western Caribbean, southwestern and eastern Atlantic, the southwestern, northwestern, and central/ eastern Indian Ocean, and the western, central, and eastern Pacific. In the Pacific Ocean, NMFS and FWS (2013a) define three populations, West, Central and East Pacific. Of these, hawksbill sea turtles from the Central Pacific population are most likely to interact with the ASLL given the overlap with the Action Area, with the exception of Hawaii where hawksbill turtles are not expected to migrate from Hawaii to the Action Area. We also consider hawksbill sea turtles from a portion of the West Pacific population, specifically Australia's Great Barrier Reef and Papua New Guinea to be reasonably likely to interact with the fishery. Together this area is generally termed as Oceania as described by Maison et al. (2010) but excluding Hawaii in this case. We refer to this area to describe the region containing the nesting beaches from which the hawksbill sea turtles that may be affected by the fishery are sourced. Hawksbill sea turtles nest broadly throughout Oceania, with the largest nesting concentration occurring on remote islands in Australia's Great Barrier Reef World Heritage Area (GBR), Australia's Torres Strait area, and Arnavon Islands in the Solomon Islands.

Although recent research using mtDNA markers has shown several distinct hawksbill nesting populations in the West, Central, and East Pacific (Gaos et al. 2016, 2018; Vargas et al. 2016; Banerjee et al. 2019; NMFS unpublished data), these studies have also revealed an exceptionally large proportion of shared mtDNA haplotypes among nesting beaches in the Pacific Ocean, which complicates analysis and limits robust assessments of nesting population contributions (Gaos et al. 2016). Furthermore, not all nesting areas have been genetically surveyed in the Pacific, so the possibility remains that specimens collected on foraging grounds come from unsampled rookeries rather than those to which they were provisionally assigned by mtDNA (Avise 2007). The confounding issues posed by shared mtDNA nesting haplotypes, large confidence intervals and limited sample sizes suggest that continued research is needed to characterize the population structure and natal origins of hawksbills at foraging habitats in the Pacific Ocean. In addition to the use of mtDNA, the incorporation of nuclear DNA (nDNA) markers will be important to improve researchers' ability to identify the natal origins of turtles at foraging grounds (Komoroske et al. 2017). A recent study using nDNA included new Pacific sampling sites: Hawaii, American Samoa, CNMI, Palau, and Australia (Banerjee et al. 2019). They found support for at least three populations, West Pacific, East Pacific and Atlantic. Their results suggest finer subpopulation structure in the West Pacific but they indicate a need for increasing sample sizes to confirm this separation.

It is possible that hawksbill sea turtle subpopulations are present within the Action Area. For example Wallace et al. (2010a) suggest six Regional Management Units for hawksbill sea turtles in the Pacific: East Pacific, North Central Pacific (Hawaii), South Central Pacific, Southwest Pacific, West Central Pacific, and West Pacific/Southeast Asia based on known nesting. However, of these only the East Pacific and Southwest Pacific have associated genetic data to support them, the rest are considered putative (Wallace et al. 2010a). Given the limited information on genetic structuring in the region and our inability to assess which potential subpopulations of hawksbill sea turtles interact with the ASLL fishery, for the purpose of this assessment, we consider all hawksbill sea turtle interactions with the fishery to be from the area of Oceania described above (Banerjee et al. 2019).

Status and Trends

Hawksbill sea turtles were listed globally as endangered in 1970. There is nesting information for the hawksbill sea turtle at nine primary locations within Oceania (excluding Hawaii) including: GBR, Papua New Guinea (PNG), Solomon Islands, Vanuatu, Fiji, the FSM, Republic of Palau, and the Samoan Islands (Independent Samoa and American Samoa) spanning the

Central Pacific population excluding Hawaii and the eastern portion of the West Pacific population. Hawksbill sea turtle nesting may occur elsewhere within this region, but any such nesting is thought to be in very low numbers. Thus, the total number of annual nesting females in Oceania is estimated based on information from the nine locations mentioned above at 5,400 – 6,160 females annually, with an overall downward trend (NMFS and FWS 2013a). We note that much of the information on abundance and trends in Table 6 are based on anecdotal information however it represents the best available information (NMFS and FWS 2013a). As trends are considered decreasing for most nesting aggregations we assume an overall decreasing trend for Oceania hawksbill sea turtles. NMFS and FWS (2013a) indicate that quantitative trend data over a period of at least 20 years are available for 11 sites worldwide and of these only the Milman Islands, Great Barrier Reef, Australia is within the Oceania region. The trend for this nesting aggregation is estimated to be declining at a rate of 4.9% per year (Limpus 2009; Mazaris et al. 2017) and we consider this the best available information on trends for Oceania hawksbill sea turtles.

Table 6. Summary of best currently available nesting information for hawksbill sea turtles in Oceania, which spans the Central Pacific (CP) population excluding Hawaii and the eastern portion of the West Pacific (WP) population described by NMFS and FWS (2013a).

Location	**Annual nesting females**	
	Estimate	**Trend**
Australia ^{WP}	4,000	Decreasing
Papua New Guinea^{WP}	~500-1,000	Decreasing*
Solomon Islands^{CP}	200-300	Increasing+
Vanuatu^{CP}	>300	Unknown
Fiji^{CP}	100-200	Decreasing*
Palau^{CP}	15-25	Decreasing*
Federated States of Micronesia^{CP}	~300	Decreasing*
Samoan Islands^{CP}	>10-30	Decreasing*
Mariana Islands**^{CP}	5-10	Decreasing*
Total	5,430-6,165	Decreasing

*Trend information is based on documented anecdotal evidence from local residents, not on long term nesting beach monitoring data sets.

** The Mariana Islands is referenced in NMFS and FWS (2013a) and therefore included in this summary table; however, no nesting activity has been documented in the Mariana's over the last decade.

+ Nesting activity in the Arnavon Islands is currently increasing but still a fraction of historic levels (Hamilton et al. 2015).

We estimate the total population of the hawksbill sea turtles in Oceania at 2,592,331 sea turtles (juveniles greater than one-year-old and adults). Given the uncertainties in these estimates including how long ago the data were collected and the short time frames over which they were collected (NMFS and FWS 2013a), we base this estimate on the lower nester abundance level of 5,430 annual nesting females per year, a remigration interval of 3.5 years, average of 3.2 nests per year, with 175 eggs per nest, a nest survival rate of 0.69 (metrics from Seitz et al. 2012 for Hawaii hawksbill sea turtle nests), a sex ratio of 7:1 (females to males; Brunson et al. 2017), first year survival of 0.44, small juvenile survival of 0.64, and large juvenile survival of 0.85 (using survival rates estimated for green sea turtles in Australia as a proxy; Chaloupka 2002), and lastly 22 years to reach maturity (Snover et al. 2013). We estimated 2,098,152 hatchlings successfully emerge from the nest in a year, but due to the high rate of mortality, we used juveniles (greater than one-year-old) and adults as our abundance metric.

Population Dynamics

Compared to green and loggerhead sea turtles, hawksbill sea turtles may exhibit faster growth rates and earlier maturation ages. Analysis using skeletochronology estimated that Hawaii hawksbills reach sexual maturity at 17–22 years, at a size of 78.6 cm SCL (Snover et al. 2013). However, using bomb ^{14}C dating technique, Van Houtan et al. (2016) estimated age at sexual maturity of hawksbills in Hawaii to be 29 years (range 23–36). In Hawaii, adult female hawksbill sea turtles nest every 3.5 years on average (range two to 10 years), laying one to six clutches in a nesting year, with an average of 175 eggs per clutch (Seitz et al. 2012). In American Samoa, Tagarino (2015) estimated clutch sizes of 114 ±43 eggs. As with other sea turtles, hawksbill sea turtles demonstrate strong nest site fidelity, returning to the region of their hatching beach upon maturation and returning to the same beaches in subsequent years (Kamel and Mrosovsky 2006).

Diving and Social Behavior

Hawksbill diving behavior has been studied at several sites (van Dam and Diez 1997a, 1997b, 1998; Houghton et al. 2003; von Brandis 2010; Martin et al. 2018). Dive patterns are influenced by complex biological and environmental factors (Blumenthal et al. 2009; Gaos et al. 2012), thus factors such as benthic topography, oceanic characteristics, prey availability, diel period, and life stage would likely affect diving behavior. Unlike other marine turtles, hawksbills are not generally deep divers, which may be a reflection of the shallow depths of their primary food—sponges and macroalgae. Hawksbills actively forage during the day and tend to rest at night (Blumenthal et al. 2009; Okuyama et al. 2010; Witt et al. 2010; Hart et al. 2012). However, Gaos et al. (2012) documented foraging activity at night as well as during the day and thought it might be due to possible overlap of foraging and resting areas.

Threats to the Species

Anthropogenic climate change is likely affecting hawksbill sea turtles, although no published accounts or direct observations have been documented to date (NMFS 2016a). However, over the long-term, climate change-related impacts will likely influence biological trajectories in the future on a century scale (Parmesan and Yohe 2003). In the future, climate change-related increasing temperatures, sea level rise, changes in ocean productivity, and increased frequency of storm events as a result of climate change are all potential threats to hawksbill sea turtles because these could lead to a reduction or cessation of male hatchling production (Hawkes et al. 2009;

Poloczanska et al. 2009), and a restriction of suitable nesting habitat. Additionally, because hawksbill sea turtles typically inhabit and depend on coral reef communities for food and shelter, they are vulnerable to changes that affect these communities including bleaching events, increased occurrence of disease, and weakening of coral skeletons as a result of global climate change (Langdon et al. 2000; Ohde and Hossain 2004; McWilliams et al. 2005).

The various habitat types hawksbill sea turtles occupy throughout their lives exposes these sea turtles to a wide variety of natural and anthropogenic threats. The beaches on which hawksbill sea turtles nest and the nests themselves are threatened by hurricanes and tropical storms as well as the storm surges, sand accretion, and rainfall that are associated with hurricanes. Hawksbill sea turtle remains have been found in the stomach content of tiger sharks near the Philippines (Witzell 1987). Witzell (1987) also notes that Japanese tuna long-line fishermen in the Solomon Islands have been known to open incidentally captured sharks to look for valuable hawksbill turtle shells (Vaughan 1981).

A wide variety of human activities adversely affect hatchlings and adult female turtles when they are on land, including beach erosion, beach armoring and nourishment; artificial lighting; beach cleaning; human presence on nesting beaches; beach driving; coastal construction and fishing piers that alter patterns of erosion and accretion on nesting beaches; exotic dune and beach vegetation; and poaching.

The primary threat to hawksbill sea turtles globally has been the direct exploitation and harvest of turtles for their shells ('tortoiseshell') and eggs (Frazier 2003; Pita and Broderick 2005; Kinch 2007; Mortimer and Donnelly 2008; Hamilton et al. 2015; Miller et al. 2019). The largest source of mortality identified for south Pacific hawksbill sea turtle has been harvest for food and tortoiseshell in the broader Coral Sea region (Allen 2007; Limpus and Miller 2008).

Fisheries bycatch in artisanal and industrial fishing gear also affects hawksbill sea turtles. Since hawksbills may not occupy pelagic waters for pronounced periods of time (Van Houtan et al. 2016b), they are not primarily (or typically) impacted by high-seas commercial fisheries, but are particularly susceptible to bycatch in nearshore artisanal fisheries gear (Brunson et al. 2017; Liles et al. 2017; DMWR unpublished stranding data). These fishery practices include drift netting, set netting, hook-and-line, and trawl fisheries, and their adverse impacts on sea turtles have been documented in marine environments throughout the world (Lutcavage et al. 1997; Epperly 2003; Wallace et al. 2013b; Liles et al. 2017).

In Australia, five stranded hawksbill sea turtles in 2011 were determined to have injuries from boat strikes (Meager and Limpus 2012). Increasing boat traffic may increases the likelihood of boat strikes (NMFS and FWS 2007b, 2013a).

Conservation

Numerous conservation programs are being implemented throughout the Pacific Ocean to protect nesting habitat and reduce harvest and fisheries bycatch of all sea turtle species (e.g., East Pacific Hawksbill sea turtle Initiative, Hawaii Island Hawksbill turtle Project, Arnavons (ACMCA), etc.; NMFS and FWS 2013a, 2018; Hamilton et al. 2015). Additionally, numerous regulatory mechanisms are in place at international, regional, national and local levels to protect sea turtles (including the South Pacific Regional Environment Programme and the Inter-American Convention for the Protection and Conservation of Sea Turtles). Many of these programs may help hawksbill sea turtles, but the species continues to decline in the Pacific and Indian Ocean areas due, in large part, to past and ongoing unsustainable harvest for food (eggs) and tortoiseshell, predation, habitat loss and climate change (Limpus and Miller 2008; Mortimer and Donnelly 2008; NMFS and FWS 2013a; Miller et al. 2019).

Summary of the Status

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

2.3.5 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 20); 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. 2011; Hall and Roman 2013; Tolotti et al. 2013; Young et al. 2017).

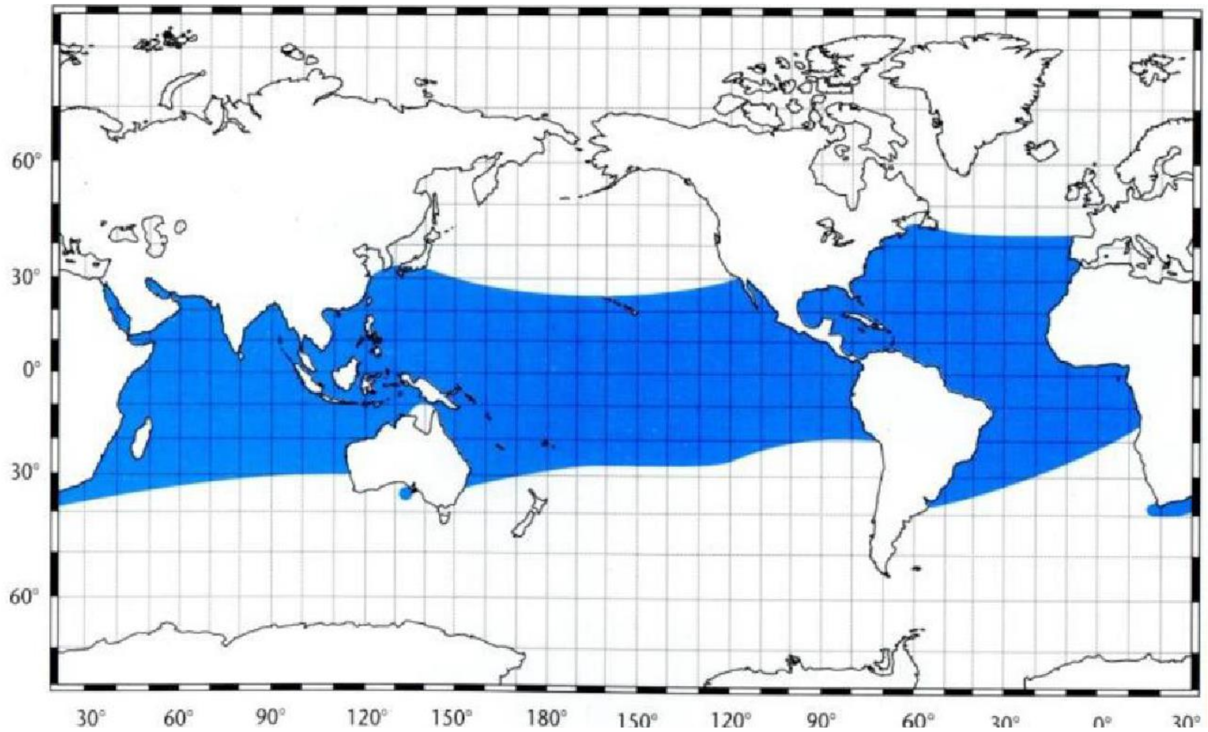

Figure 20. Geographical distribution of the oceanic whitetip shark (Last and Stevens 2009).

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. However, to take a cautionary approach, 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.

Status and Trends

Oceanic whitetip sharks were globally listed as threatened in 2018. Historically, oceanic whitetip sharks were 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 and continued declines throughout its distribution. 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), 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, (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 2020a). 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 21; 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). The implementation of CMM-2022-04 is anticipated to improve the survival of released sharks throughout the WCPO by eliminating wire leaders and shark lines.

We believe this new information provided by Bigelow et al. (2022) constitutes the best available. However, Bigelow et al. (2022) do not provide specific population trends, only indicating that the trends in spawning biomass ratios are anticipated to be positive (Figure 21). 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.

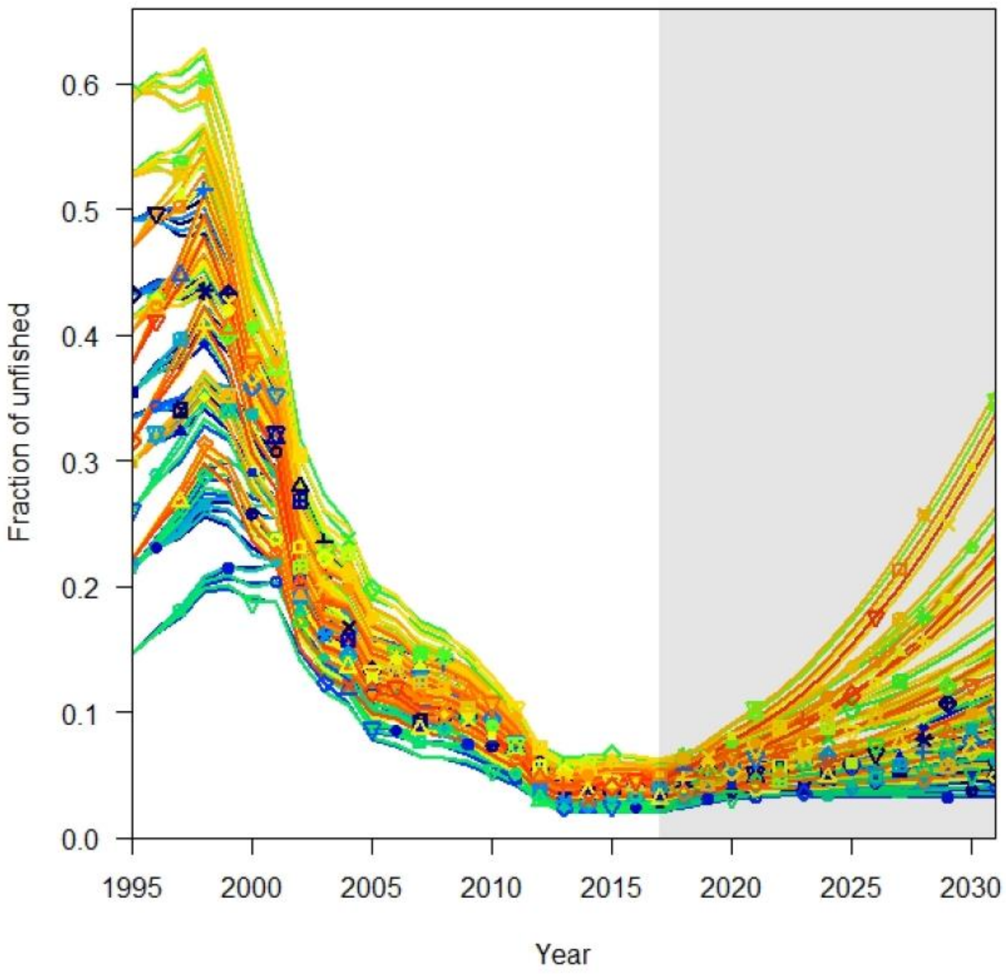

Figure 21. Projected ratios of spawning biomass (projected to 2031) to the equilibrium unfished spawning biomass for WCPO oceanic whitetip sharks with updated at-vessel and post-release mortality rates and the prohibition of wire branchlines and shark line (Figure 7 in Bigelow et al. 2022).

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 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 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.

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 86% 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 2018b). In total, there were 6,139 observed interactions in the DSLL fishery from 2004 to 2022 (NMFS unpublished data).

When these data are adjusted to account for the percentage of observer coverage, approximately 31,467 oceanic whitetip sharks are likely to have been incidentally captured in the HI DSLL fishery in the 19-year period from 2004 to 2022 (McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021, 2022). Finally, 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 allow fishermen to remove shark fins provided that the weight of the fins does not exceed 5% of the total weight of shark carcasses landed or found onboard. The WCPFC has implemented several conservation and management measures for sharks with the following objectives (Clarke 2013): (1) promote full utilization and reduce waste of sharks by controlling finning (perhaps as a means to indirectly reduce fishing mortality for sharks); (2) increase the number of sharks that are released alive (in order to reduce shark mortality); and (3) increase the amount of scientific data that is collected for use in shark stock assessments. Also, specific to oceanic whitetip sharks, CMM 2011-04 prohibits WCPFC vessels from retaining onboard, transshipping, storing on a fishing vessel, or landing any oceanic whitetip shark, in whole or in part, in the fisheries covered by the Convention. This CMM was later replaced in 2019 by CMM-2019-04 which was in-turn was replaced in 2022 by CMM-2022-04 for all sharks. The measure retains the retention prohibition for oceanic whitetip sharks, and includes additional measures on minimizing bycatch (including some gear restrictions), implementing safe release practices, and prohibiting wire leaders and shark lines for longline fishing.

Summary of the Status 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; Young et al. 2017). 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.

2.3.6 Indo-west Pacific Scalloped Hammerhead Shark

Distribution and Population Structure

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

Scalloped hammerhead sharks are highly mobile, partly migratory, and are likely the most abundant of the hammerhead DPSs (Maguire et al. 2006). Females show site fidelity with no trans-oceanic migration while males have been shown to disperse over vast distances (Daly-Engel et al. 2012; Francis 2016). These sharks have been observed making migrations along continental margins as well as between oceanic islands in tropical waters (Kohler and Turner 2001; Duncan and Holland 2006; Bessudo et al. 2011; Diemer et al. 2011; Prus 2013; Chin et al. 2017).

Indo-West Pacific scalloped hammerhead sharks may potentially make long range movements. However, most of the literature focuses on scalloped hammerhead sharks in other locations around the world. For instance, the median distance between mark and recapture of 3,278 tagged adult sharks along the eastern United States was less than 100 km (Kohler and Turner 2001). Along the east coast of South Africa, average distance moved by *S. lewini* was 147.8 km (data from 641 tagged scalloped hammerhead sharks; Diemer et al. 2011). In Kaneohe Bay, Hawaii, sharks travelled as far as 5.1 km in the same day but the mean distance between capture points was 1.6 km (data from 151 recaptured juveniles; Duncan and Holland 2006).

These tagging studies reveal the tendency for scalloped hammerhead sharks to aggregate around and travel to and from core areas or "hot spots" within locations (Holland et al. 1993; Duncan and Holland 2006; Hearn et al. 2010; Bessudo et al. 2011; Nalesso et al. 2019), however they are also capable of traveling long distances (1941 km, Bessudo et al. 2011; 1671 km, Kohler and Turner 2001, Hearn et al. 2010; 629 km, Diemer et al. 2011; 710 km, Prus 2013; 710 km, Nalesso et al. 2019). In addition, in many of these tagging studies scalloped hammerhead sharks were tracked leaving the study area for long periods of time, ranging from 2 weeks to several months (Hearn et al. 2010; Bessudo et al. 2011; Prus 2013; Nalesso et al. 2019) to almost a year (324 days) (Duncan and Holland 2006), but eventually returning, displaying a level of site fidelity to these areas.

Hence, scalloped hammerheads are likely organized into localized aggregations that could become depleted under heavy fishing pressure, but these aggregations are not genetically distinct given the male-mediated gene flow between aggregations. In support of this, Duncan et al. (2006) found genetic structure between ocean basins but little population structure along continental margins.

Status and Trends

Indo-west Pacific scalloped hammerhead sharks are listed as threatened because of overharvesting. Although range-wide trends in the abundance of this DPS are unknown, CPUE data suggest that local abundance throughout the range of the DPS have declined significantly (Miller et al. 2014). For example, the hammerheads in Australia's northwest marine region has been estimated to have declined between 58-76% between 1996 and 2005 (Miller et al. 2014). Similarly, catch rates of *S. lewini* in beach mesh programs in South Africa have declined by 99%, 86%, and 64% from 1952-1972, 1961-1972, and 1978-2003, respectively (Dudley and Simpfendorfer 2006; Ferretti et al. 2010). Estimates of the decline in Australian hammerhead abundance range from 58-85% (Heupel and McAuley 2007).

Data from protective shark meshing programs off beaches in New South Wales (NSW) and Queensland also suggest significant declines in hammerhead abundance off the east coast of Australia. From 1973 to 2008, the number of hammerheads caught per year in NSW beach nets decreased by more than 90%, from over 300 individuals to fewer than 30 (Reid and Krogh 1992; Williamson 2011; Miller et al. 2014). Similarly, data from the Queensland shark control program indicate declines of around 79% in hammerhead shark abundance between the years of 1986 and 2010, with *S. lewini* abundance fluctuating over the years but showing a recent decline of 63% between 2005 and 2010 (QLD DEEDI 2011as cited in Miller et al. 2014).

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

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

Population Dynamics

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

Diving and Social Behavior

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

Threats to the Species

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

Contributing to the scalloped hammerhead shark's biological vulnerability is the fact that these sharks are obligate ram ventilators and suffer very high at-vessel fishing mortality from fisheries where they are not able to continually swim forward (Morgan and Burgess 2007; Macbeth et al. 2009; Miller et al. 2014; Dapp et al. 2016). For example, between 92 to 94% of the hammerhead sharks captured in bottom longline fisheries die at vessel and this does not include post release mortality (Morgan and Burgress 2007). Considering purse seine fisheries, while Hutchinson's (2015) study focused on silky sharks, the study showed that sharks confined in the sack portion of the net just prior to loading suffered much higher mortality with only a 6.67% chance of survival after brailing. This highlights the consequences of restricting the movement of hammerhead shark DPSs given their respiratory mode (i.e., obligate ram ventilation). Compared to other chondrichthyans, scalloped hammerhead sharks appear to sustain a higher level of fishing mortality (Miller et al. 2014). Miller et al. (2014) further ranked high at-vessel mortality as the most serious threat to the species.

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

United States fisheries appear to have less influence on this DPS' status when compared to foreign fisheries. United States fisheries in Alaska and California, and the Hawaii SSLL fishery do not overlap with the DPS' range. However, the Hawaii DSLL fishery interacts with up to an estimated 4 Indo-west scalloped hammerheads per year (95th percentile; McCracken 2019a; McCracken and Cooper 2020a, 2020b, 2021, 2022). While the DSLL fishery overlaps the range of the DPS, it does not overlap the Action Area. Thus, the greatest influence on the decline of this DPS is from foreign fisheries throughout the DPS' range in the western Pacific.

Conservation

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

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

Summary of the Status

In this section of this biological opinion, we explained that the Indo-West Pacific scalloped hammerhead shark is threatened, and that the DPS' trend appears to be decreasing. We used our knowledge of the DPS' demography and population ecology to capture the primary factors that appear to determine the Indo-West Pacific scalloped hammerhead shark population dynamics.

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

2.3.7 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 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 22). 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 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 22).

Several authors have reported that giant manta ray likely occur in small regional subpopulations (Lewis et al. 2015; Stewart et al. 2016a; Marshall et al. 2018; Beale et al. 2019) and may have distinct home ranges (Stewart et al. 2016a). The degree to which subpopulations are connected by migration is unclear but is assumed to be low (Stewart et al. 2016a; Marshall et al. 2018) so regional or local populations are not likely to be connected through immigration and emigration (Marshall et al. 2018), making them effectively demographically independent.

While NMFS' concluded that the species is likely to become endangered within the foreseeable future throughout a significant portion of its range (the Indo Pacific and eastern Pacific), NMFS did not find the species met the criteria to list as a DPS (83 FR 2916, and 82 FR 3694). This decision is unique to the listing process, and does not mean that NMFS should not or would not consider the potential role that populations play in evaluating whether a proposed action is likely to result in appreciable reduction in numbers, distribution or reproduction, or whether such reductions may affect the viability of the putative populations that comprise the listed species. The preponderance of current evidence, combined with expert opinion suggest the species likely has a complex population structure, and while it may occasionally be observed making long distance movements, it likely occurs in small spatially separated subpopulations, though to be viable the abundance of each subpopulation likely needs to be at least 1,000 individuals (Frankham et al. 2014). This structure is further supported by studies described by Beale et al. (2019) that have documented fisheries-induced declines in several isolated subpopulations (Lewis et al. 2015; Stewart et al. 2016b; Moazzam 2018).

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

The Status Review (Miller and Klimovich 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 7 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 22. 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 IUCN lists them as endangered 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 7[5]). CITES (2013) three giant manta ray subpopulations that have been studied and population estimates provided, and counts for more than ten aggregations (Table 7). CITES (2013) also discusses an additional approximately 25 aggregations where species-level information (i.e., *M. birostris* vs *M. alfredi*) does not exist and, while actual abundance estimates are not available, it is assumed they consist of very small number of individuals. This information was compiled from O’Malley et al. (2013), Heinrichs et al. (2011), Lewis et al. (2015), and Fernando and Stevens (2011).

The most comprehensive of these is O’Malley et al. (2013) that presents an overview of the economic value of manta ray watching tourism. They highlight 23 sites globally, and within the Action Area, 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;

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[5] 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.*

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 7). 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 7 may not be indicative of population size.

The number of individually identified giant manta ray for each studied aggregation ranges from less than 50 in regions with low survey effort or infrequent sightings to more than 1,000 in some regions with targeted, long-term studies. However, ongoing research including mark-recapture analyses suggests that typical subpopulation abundances are more likely in the low thousands (e.g., Beale et al. 2019) and in rare cases may exceed 10,000 in areas with extremely high productivity (pers. comm. Joshua Stewart, Manta Trust to A. Garrett, NMFS PRD, 2021). Of the 12 studied subpopulations identified in Table 7, statistical analyses of sightings/photo-identification data to estimate total population size have 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 7). 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 captures in the WCPO longline and purse seine fisheries. Giant manta ray were not reliably identified to species by observers in the WCPO purse seine fishery until about 2011 (NMFS 2021a). In their analysis, Tremblay-Boyer and Brouwer (2016) found increasing trends in CPUE from 2005 to 2016 for giant manta rays but they caution that these trends represent increases in compliance with reporting the species and does not represent an index of abundance. CPUE trends in the longline fisheries indicate that giant manta rays are observed less frequently in recent years compared to 2000-2005, suggesting a decline in abundance (Tremblay-Boyer and Brouwer 2016).

Table 7. Numbers of recorded individuals and subpopulation estimates of giant manta ray at identified locations adapted from CITES (2013a) and updated with supplementary references as specified.

Location	**Recorded Individuals**	**Subpop. 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 inter-pregnancy period which could depend on the availability of resources. Additionally, sex ratios may differ between populations. Beale et al. (2019) noted a statistically significant female-biased sex ratio of 2.62(f):1 in Raja Ampat. However, Pate and Marshall (2020) did not find a statistical difference in Florida with a sex ratio of 1:1 and Stewart et al. (2018) noted a ratio of 1.3(f):1 in the Flower Garden Banks of the Gulf of Mexico. Differences between locations may be due to unique threats to each population.

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

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

Based on the elasticity analysis, population growth rate was most sensitive to changes in the survival rate of adults regardless of yearling and juvenile survival rates (Smallegange et al. 2016). In other words, in order to prevent populations from declining further, Smallegange et al. (2016) found that increases in adult survival rates would have the greatest impact, such as through protection of adult aggregation sites or a reduction in fishing of adult manta rays (Smallegange et al. 2016). However, their results also show that low yearling and juvenile survival can result in declining populations even if adult survival remains high, indicating that 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 (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 commercial longline, trawl, purse-seine and gillnet 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 47 interactions with giant manta rays from 2004-2022 with an estimated 239 interactions (NMFS unpublished data). Observer data (100% coverage) for the Hawaii SSLL fishery recorded 21 interactions with giant manta rays from 2004-2022 (NMFS Observer data).

Conservation

Domestic fishery regulations prohibit the retention of manta rays in United States fisheries. 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 WCPFC CMM in January 2021. CMM 2019-05 (effective January 1, 2021) prohibits all fishing vessels operating in the high seas and/or exclusive economic zones of the Convention area and flagged to Members, Cooperating Non-Members and Participating Territories of the WCPFC from targeted fishing or intentional setting on mobulid rays; from retaining on board, transshipping, or landing any part or whole carcass of mobulid rays; fishing vessels must promptly release animals alive and unharmed that will result in the least possible harm to the individuals captured. The U.S. issued a rule to put the handling practices in CMM 2019-05 into regulation for U.S. fisheries (88 FR 30671).

Summary of the status

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

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

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.

The past and present impacts of human and natural factors leading to the status of the 11 ESA-listed marine species addressed by this biological opinion within the Action Area include fishery interactions, vessel strikes, climate change, pollution, marine debris, and entanglement. The environmental baselines for these species addressed by this biological opinion 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, 2010, 2015a, 2021). We also used the 2017 pelagics report (WPRFMC 2018), the green sea turtle 5-year status review (Seminoff et al. 2015), the olive ridley sea turtle 5-year status review (NMFS and FWS 2014), 2017 status review report for oceanic whitetip sharks (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, 2022). Climate change will likely have its most pronounced effects on vulnerable species whose populations are already in tenuous positions (Williams et al. 2008). As such, we expect the risk of extinction for ESA-listed species to rise with the degree of climate shift associated with global warming. Increasing atmospheric temperatures have already contributed to documented changes in the quality of freshwater, coastal, and marine ecosystems and to the decline of endangered and threatened species populations (Mantua et al. 1997; Karl et al. 2009).

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. Notably, leatherback sea turtles were predicted to gain core habitat area, whereas loggerhead sea turtles were predicted to experience losses in available core habitat. Such range shifts could affect sea turtle foraging success as well as sea turtle reproductive periodicity (Kashner et al. 2011; Pike 2014).

Significant impacts to sea turtles from ocean acidification may be indirectly tied to foraging opportunities resulting from ecosystem changes (Busch et al. 2013; Haigh et al. 2015; Chan et al. 2017). Nearshore waters off California have already shown a persistent drop in pH from the global ocean mean pH of 8.1 to as low as 7.43 (Chan et al. 2017). Ocean acidification may cause a shift in phytoplankton community composition and biochemical composition that can impact the transfer of essential nutrients to predators that eat plankton (Bermudez et al. 2016). Increased ocean acidification may also have serious impacts on fish and larval development and behavior (Raven et al. 2005, Bignami et al. 2013) that could impact fish populations (Munday et al. 2009). In turn this could negatively impact piscivorous threatened and endangered species that rely on those populations for food.

Sea turtles occupy a wide range of terrestrial and marine habitats, and many aspects of their life history have been demonstrated to be closely tied to climatic variables such as ambient temperature and an increase in storms (Hawkes et al. 2009). Pike et al. (2006) concluded that warming sea surface temperatures may lead to potential fitness consequences in sea turtles resulting from altered seasonality and duration of nesting. Sea turtles may also expand their range as temperature-dependent distribution limits change (McMahon and Hays 2006; Poloczanska et al. 2009).

Sea turtles have temperature-dependent sex determination, and many populations produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Newson et al. 2009; Patricio et al. 2017; Jensen et al. 2018). Genetic analyses and behavioral data suggest that populations with temperature-dependent sex determination may be unable to evolve rapidly enough to counteract the negative fitness consequences of rapid global temperature change (Hays 2008 as cited in Newson et al. 2009). Altered sex ratios have been observed in sea turtle populations worldwide (Mazaris et al. 2008; Reina et al. 2008; Robinson et al. 2008; Fuentes et al. 2009). This does not yet appear to have affected population viabilities through reduced reproductive success, although average nesting and emergence dates have changed over the past several decades by days to weeks in some locations (Poloczanska et al. 2009). A fundamental shift in population demographics may lead to increased instability of populations that are already at risk from several other threats. In addition to altering sex ratios, increased temperatures in sea turtle nests can result in reduced incubation times (producing smaller hatchling), reduced clutch size, and reduced nesting success due to exceeded thermal tolerances (Fuentes et al. 2009, 2010a, 2010b, 2011; Azanza-Ricardo et al. 2017).

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

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

Changing patterns of coastal erosion and sand accretion, combined with an anticipated increase in the number and severity of extreme weather events, may further exacerbate the effects of sea level rise on turtle nesting beaches (Wilkinson and Souter 2008). For instance, after Hurricane Walaka went through the Central North Pacific in October of 2018 much of East Island in French Frigate Shoals was left largely inundated. Roughly 50% of the Central North Pacific green sea turtles (not considered in this biological opinion) nest at East Island and the future impacts from the inundation of the majority of East Island is unclear at this time. Similarly, Hurricane Andrew impacted South Florida in 1992 with widespread effects to sea turtle nests, primarily due to associated storm surge inundating nests across 90 miles of the State (Milton et al. 1994). Modification in the beach topography also caused delayed mortality in surviving hatchlings due to sand accretion, which resulted in their subsequent drowning (Milton et al. 1994).

Climate change is expected to affect the intensity of hurricanes through increasing sea surface temperatures, a key factor that influences hurricane formation and behavior (EPA 2010). Extreme weather events may directly harm sea turtles, causing "mass" strandings and mortality (Poloczanska et al. 2009). Studies examining the spatio-temporal coincidence of marine turtle nesting with hurricanes, cyclones and storms suggest that cyclical loss of nesting beaches, decreased hatching success and hatchling emergence success could occur with greater frequency in the future due to global climate change (Hawkes et al. 2009).

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

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. Several ESA-listed species and habitats 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. The primary species of this consultation affected by these practices are sea turtles and elasmobranchs.

3.2.1 Sea Turtles

Green Sea Turtles

Most harvest of sea turtles occurs on nesting beaches (adult females and eggs). Hence, for the green sea turtle species considered in this assessment, direct harvest within the Action Area will primarily be for those species that have nesting habitats within the Action Area, which are the Central South Pacific, Central West Pacific, and East Indian-West Pacific green sea turtles. Direct consumption of green sea turtle meat and eggs is known or suspected to still occur in numerous nations within the Central South Pacific including American Samoa, Cook Islands, Fiji, Independent Samoa, Kiribati, Tonga, Tuvalu, and Tokelau (Balazs 1983; Tuato'o-Bartley et al. 1993; Bell et al. 1994; NMFS and FWS 1998a; Obura and Stone 2002; Prescott et al. 2004; Alefaio et al. 2006; Batibasaga et al. 2006; Bell et al. 2009; Havea and MacKay 2009; Laveti and MacKay 2009; Bradshaw and Bradshaw 2012; White 2012; White and Galbraith 2013).

Directed harvest of turtles and their eggs is likely the primary threat to the Central West Pacific green sea turtle and is a past, current, and ongoing threat at all nesting and foraging habitats of the region, including: the CNMI, FSM, Guam, Kiribati (Gilbert Islands chain), Papua, Papua New Guinea, Republic of the Marshall Islands, and Palau (Eckert 1993; NMFS and FWS 1998a; Hitipeuw and Maturbongs 2002; Philip 2002; Maison et al. 2010; Humber et al. 2014; Seminoff et al. 2015; Tapilatu et al. 2017; Summers et al. 2018a, 2018b).

Egg harvest remains legal in several of the countries within the range of the East-Indian-West Pacific green sea turtle (Humber et al. 2014). In West Java, Indonesia, the mean annual egg harvest was 2.5 million eggs in the 1950s and 400,000 eggs in the 1980s (Schulz 1987 as cited in Seminoff et al. 2015). However, this apparent decline could be reflective of a decline in egg collection efforts rather than a decline in egg production.

Egg harvesting in Indonesia occurred for decades until 1999. Illegal egg harvesting continues, but there is an increased effort to fully protect green sea turtles from harvest in Indonesia (Reischig et al. 2012). Nesting females continue to be killed in countries within Southeast Asia and the Indian Ocean (Fleming 2001; Cruz 2002). Local islanders in Indonesia have traditionally considered turtles, especially green sea turtles, as part of their diet (Hitipeuw and Pet-Soede 2004). About 25,000 green sea turtles were being exploited for meat each year toward the end of the 20th century (Dethmers et al. 2010). In addition, in the 1960s and 1970s, Indonesia exported 25,000 to 50,000 stuffed turtles annually with the green sea turtle being the most common species.

Hawksbill Sea Turtles

A primary threat to hawksbill sea turtles globally has been the direct exploitation and harvest of turtles for their shells (‘tortoiseshell’) and eggs (Frazier 2003; Kinch 2007; Pita and Broderick 2005; Mortimer and Donnelly 2008; Hamilton et al. 2015; Miller et al. 2019). The largest source of mortality identified for hawksbill sea turtles has been harvest for food and tortoiseshell in the broader Coral Sea region (Allen 2007; Limpus and Miller 2008). Between 1950 and 1992, approximately 1.3 million hawksbill sea turtle shells were collected to supply tortoiseshell to the Japanese market, the world’s largest. Japan stopped importing tortoiseshell in 1993 in order to comply with CITES. Miller et al. (2019) found that wildlife trade networks concentrated in Southeast Asia harvested 9 million turtles, over six times previous estimates provided by Mortimer and Donnelly (2008). While nearly all CITES signatories have agreed to an international trade ban, legal domestic exploitation in several countries still exists (Humber et al. 2014).

Leatherback Sea Turtles

Leatherback sea turtle egg collection occurs in many countries around the world (NMFS and FWS 2013b). Despite conservation efforts, egg harvest continues at certain levels in Indonesia, Papua New Guinea, Solomon Islands and Vanuatu (Bellagio Steering Committee 2008). Harvest of females remains a matter of concern on many beaches (NMFS and FWS 2013b). A traditional harvest of subadult and adult leatherbacks occurs in the Kei Islands in Indonesia (Lawalata et al. 2006; Suarez and Starbird 1996). However, most villages in the Kei Islands have abandoned the harvest and the annual estimate of interactions is between 10 and 15 leatherbacks (C. Hitipeuw, World Wildlife Fund Indonesia, 2013 personal communication as cited in NMFS and FWS 2013b). Local communities are interested in developing ecotourism programs that rely on healthy leatherback populations, and there is hope that the harvest levels will remain low into the future (C. Hitipeuw, World Wildlife Fund Indonesia, 2013 personal communication as cited in NMFS and FWS 2013b).

Olive Ridley Sea Turtles

The largest harvest of sea turtles in human history most likely occurred on the west coasts of Central and South America in the 1950s through the 1970s, when millions of adult olive ridley sea turtles were harvested at sea for meat and leather, simultaneously with the collection of many millions of eggs from nesting beaches in Mexico, Costa Rica and elsewhere. Unsustainable harvest led to extirpation of major arribadas, such as at Mismaloya and Chacahua in Mexico by the 1970s, prompting listing of these nesting aggregations as endangered under the ESA and their protection in Mexico since 1990. Globally, legal harvest of olive ridley sea turtle adults and eggs was reduced in the late 1980s and early 1990s, but legal harvest of eggs continues in

Ostional, Costa Rica. Illegal harvest of eggs is common throughout Central America, western Pacific, and India (Cornelius et al. 2007 as cited in NMFS and FWS 2014).

3.2.2 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 and provisionally in 2022-04), 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). Lastly, CMM 2022-04 will prohibit wire leaders and shark lines.

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 23).

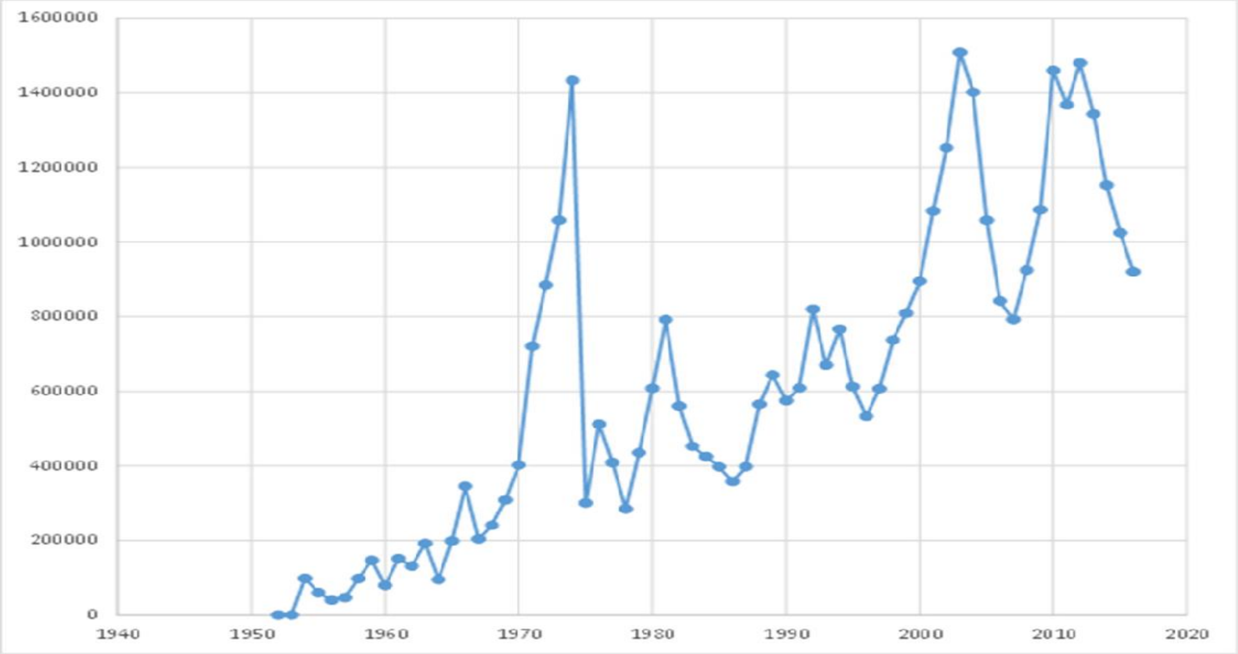

Figure 23. Pacific longline effort levels south of 20˚S in hundred hooks, 1952–2016 (SPC 2019).

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, 2022a) and pelagic troll fishery (NMFS 2009) are managed under the American Samoa Archipelago Fishery Ecosystem Plan and the Pacific Pelagic Fishery Ecosystem Plan respectively. The American Samoa bottomfish Biological Opinion found the action not likely to adversely affect sea turtles, oceanic whitetip sharks, or giant manta rays (NMFS 2022a). However, the American Samoa troll fishery is expected to interact with four green sea turtles annually. There are no expected interactions with elasmobranchs; therefore, these fisheries will not be discussed in further detail. We also note, other United States bottomfish fisheries

adversely affect 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 24; 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 and foreign purse seine. Regulation of the United States WCPO purse seine fishery is under the Western and Central Pacific Fisheries Convention Implementation Act (WCPFCIA; 16 U.S.C. 6901 *et seq*.) and implementing regulations (50 CFR Subpart O), High Seas Fishing Compliance Act (HSFCA; 16 U.S.C. 5501 *et seq*.) and implementing regulations (50 CFR 300 Subpart R), Tuna Conventions Act, as amended, and implementing regulations (50 CFR Subpart C), and regulations implementing the Fishery Ecosystem Plan for Pacific Pelagic Fisheries of the Western Pacific Region (Pelagics FEP) pursuant to the MSA (50 CFR Part 665; NMFS 2021a). 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 25; WCPFC 2021). The United States WCPO purse seine fishery is typically observed at 100%, under the SPTT, the observer program is administered by the 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 153*)*-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).

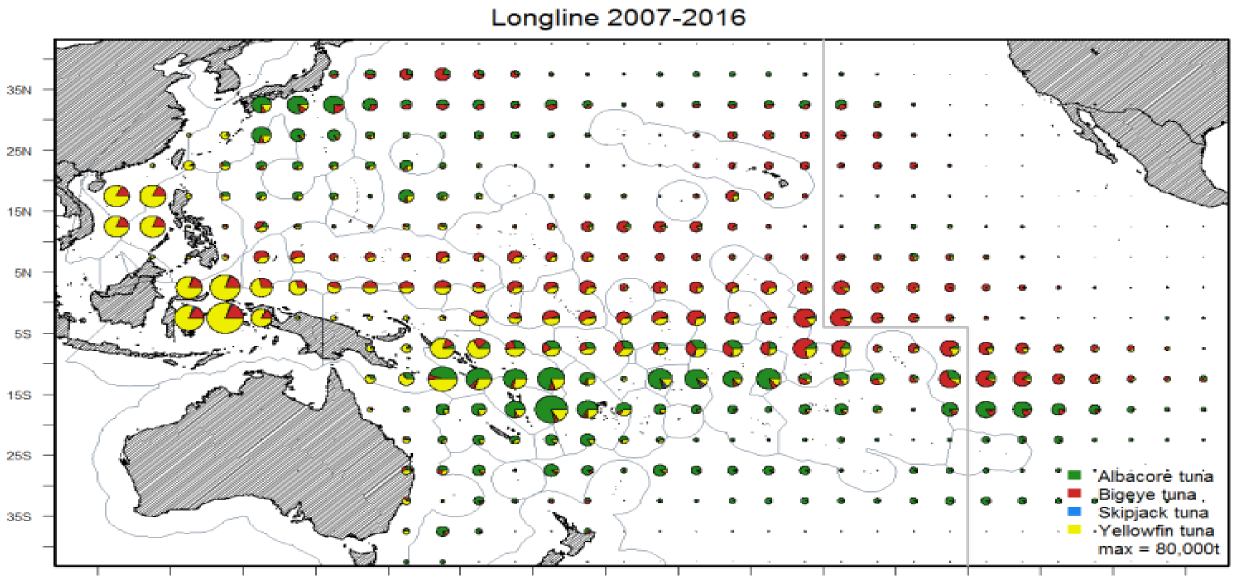

Figure 24. Spatial distribution of tuna catch for the West Central Pacific longline fleets (from Brouwer 2017).

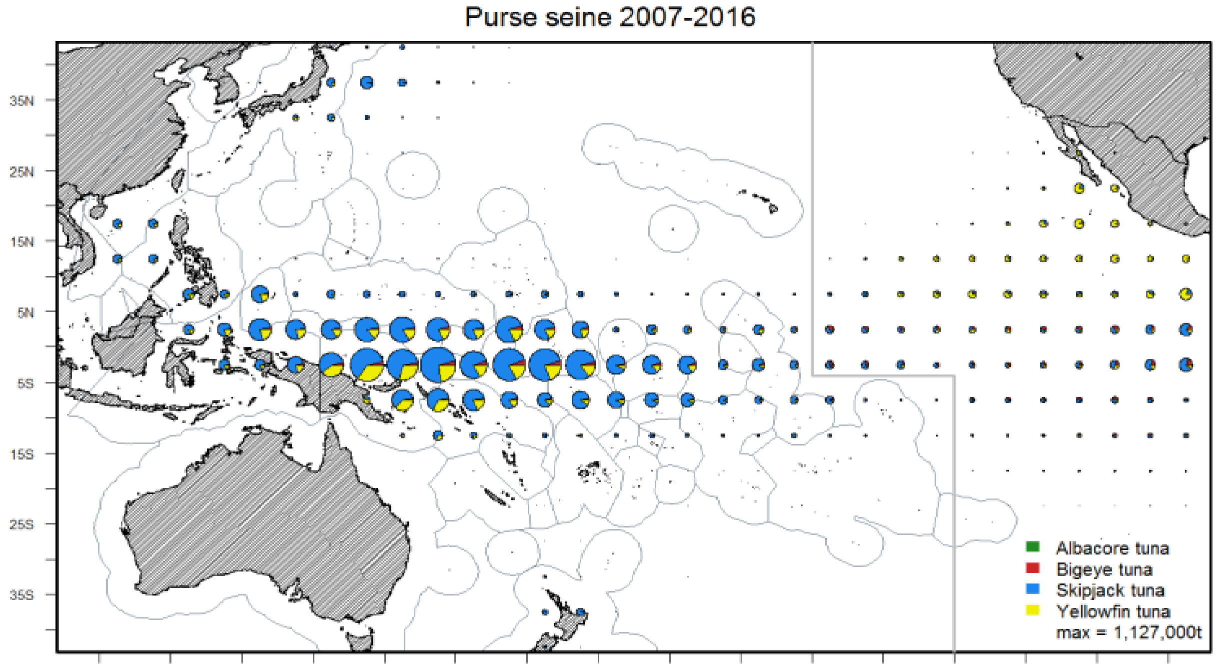

Figure 25. Spatial distribution of tuna catch for the West Central Pacific purse seine fleets (from Brouwer 2017).

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

3.3.1 Sea Turtles

The American Samoa-based bottomfish and pelagic troll fisheries

The American Samoa bottomfish Biological Opinion found the action is not likely to adversely affect sea turtles (NMFS 2022a). However, the American Samoa troll fishery is expected to interact with four green sea turtles annually (NMFS 2009).

Foreign WCPO longline Fisheries

Estimates of sea turtle bycatch is difficult to obtain when observer coverage is low. However, several studies using different methods have analyzed bycatch data from longliners in the WCPO and we have summarized this information in this section. The United States longline data and non-United States longline data is aggregated, however United States fishery interactions are specifically noted above in Section 4.3.2.2. The following information is presented to relay the magnitude of effects in the Region and pertains to the Pacific to the best of our ability. Accordingly, we draw upon the best available scientific and commercial information on fishery impacts Pacific-wide in this discussion.

Lewison et al. (2004) collected fish catch data from 40 nations and turtle bycatch data from 13 international observer programs to estimate global longline bycatch of loggerhead sea turtle and leatherback turtles in 2000. In the Pacific, they estimated 2,600 – 6,000 loggerhead sea turtle juvenile and adult mortalities from pelagic longlining in 2000 (Lewison et al. 2004). They estimated 1,000 – 3,200 leatherback mortalities from pelagic longlining in 2000 (Lewison et al. 2004). However, using effort data from Lewison et al. (2004) and bycatch data from Molony (2005), Beverly and Chapman (2007) estimated loggerhead sea turtle and leatherback longline bycatch to be approximately 20% of that estimated by Lewison et al. (2004), or 520 – 1,200 juvenile and adult loggerhead sea turtles and 200 – 640 juvenile and adult leatherbacks annually. An estimate of 626 adult female leatherback mortalities from pelagic longlining in 1998 was made by Kaplan (2005), or roughly 2,500 juveniles and adults.

In 2015, a workshop was convened to analyze the effectiveness of sea turtle mitigation measures in the tuna RFMOs and 16 countries provided data on observed sea turtle interactions and gear configurations. From 1989-2015, those sixteen countries reported there were 2,323 observed sea turtle interactions with approximately 5% observer coverage. The United States reported 27% of those interactions, which we described in detail in the previous section. Reported interactions and estimated total interactions are given in Table 8 (Common Oceans (ABNJ) Tuna Project 2017). As mortality rates were not estimated, we cannot assess the impact of these interactions at the population level.

Table 8. Turtle interactions reported for the Common Oceans (ABNJ) Tuna Project 2017 workshop from 16 countries from 1989-2015 and expanded based on 5% coverage in the WCPFC*.

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

*This table includes interactions from all longline fleets both United States and foreign. Observer coverage is expanded based on 5% for international fleets. The ASLL fleet has approximately 20% observer coverage. The metrics specific to the United States fishery are presented separately to provide better clarity specific to interactions in the Action Area.

Table 9 shows the average median annual estimates for sea turtle interactions with longliners from 10°S to 10°N from 2003 to 2017, summarized from Table 32b in Peatman et al. (2018b). It should be noted that these data already include United States data. Additionally, caveats apply as observer coverage range from 1 to 4.5% of the total hooks set (Peatman et al. 2018b). The data have wide confidence intervals for sea turtles (Peatman et al. 2018b). Peatman (2018b) states that these estimates are likely overestimated after comparing results from the ABNJ report for olive ridley sea turtles where their model accounted for estimates of sea turtle distributions and also had additional observer data provided for the workshop. Again, mortality rates were not estimated therefore we cannot assess the impact of these interactions at the population level.

Table 9. Median annual estimate and 95% confidence interval for median annual estimate for numbers of sea turtles captured in the region 10°S to 10°N from 2003-2017 in WCPO longline fisheries (summarized from Table 32b in Peatman et al. 2018b).

Species	Median (95% confidence interval) Annual Estimated Interactions	Total Estimated Interactions 2003 - 2017
Green Sea Turtle	1,863 (1,381 – 2,345)	27,949
Leatherback Sea Turtle	557 (439 – 676)	8,362
Loggerhead Sea Turtle	497 (288 – 706)	7,453
Olive Ridley Sea Turtle	8,086 (5,758 – 10,415)	121,292

Finally, bycatch estimates of key species were summarized from annual reports by the WCPFC (2021a). Data for sea turtles are summarized in Table 10. This information is not exclusive of United States fishery data. Figure 26 shows the trends in CPUE (captures per hook) from 2013 to 2019 for sea turtles captured in WCPO longline fisheries. CPUE was calculated as observed turtle captures divided by observed hooks.

Table 10. Average and (95% confidence interval) of the annual numbers of sea turtles reported captured/killed from 2013 to 2020 by participating countries reporting catch data to the Western and Central Fisheries Commission for longline fisheries operating in the Action Area, data were reported in 5°x 5°bins. Estimated Interactions and Mortalities calculated using the Expansion Factor ([100/Observer Coverage] x [Observed Interactions] (WCPFC 2019a).

Species	**Observed Interactions**	**Observed Mortalities**	**Estimated Interactions**	**Estimated Mortalities**	**Total Captures 2013 to 2020**	**Total Mortalities 2013 to 2020**
Green Sea Turtle	34 (28 – 39)	17 (14 – 21)	1,260 (1,091 - 1,429)	644 (531 – 758)	10,078	5,156
Hawksbill Sea Turtle	14 (9 – 18)	6 (5 – 7)	495 (376 – 314)	226 (171 – 281)	3,959	1,807
Leatherback Sea Turtle	18 (14 – 22)	1 (1 – 3)	722 (468 – 976)	76 (16 – 136)	5,779	606
Olive Ridley Sea Turtle	92 (46 – 138)	71 (32 – 110)	3,101 (1,955 – 4,246)	2,337 (1,405 – 3,268)	24,806	18,695
Percent Observer Coverage (2013-2018): 2.8% (2.2 – 3.4%)						

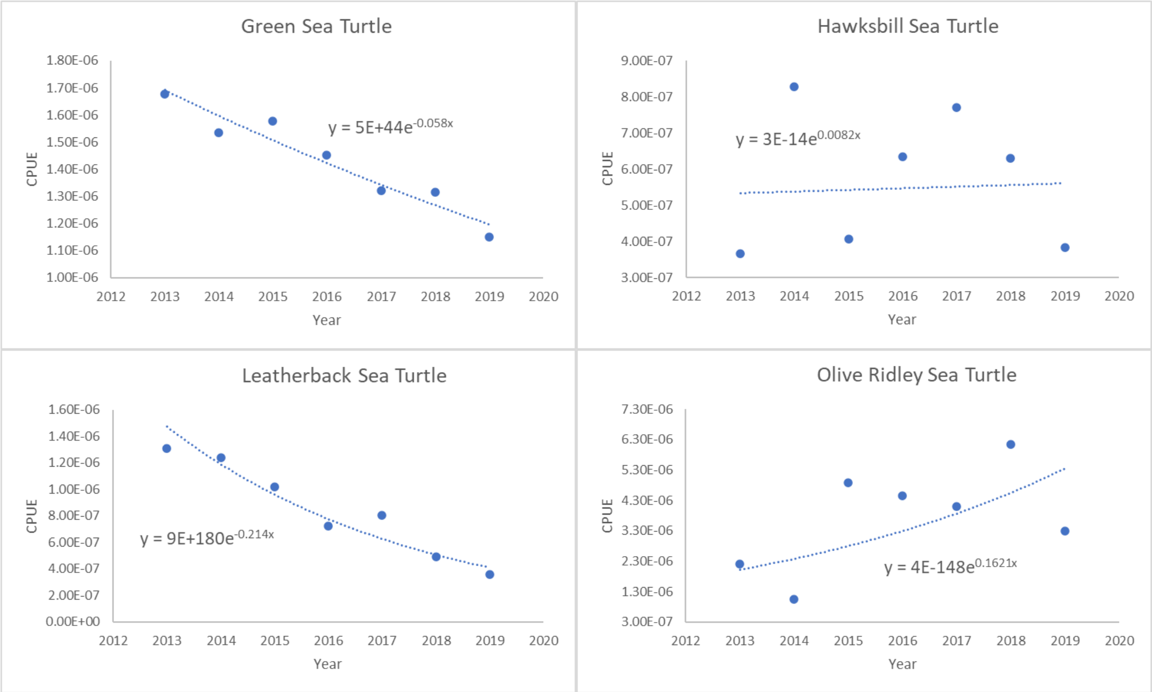

Figure 26. Trends in CPUE for sea turtles captured in the WCPO longline fisheries. Dotted lines are fitted exponential curves, equations describe these curves and the exponent in the equations is the mean percent per year change in CPUE.

United States WCPO Purse Seine

No genetic data has been collected from green sea turtles interacting with the United States WCPO purse seine fishery. While genetic data available from the ASLL fishery exists, this fishing area overlaps with only a small a portion of the Action Area of the United States WCPO purse seine fishery. Therefore, NMFS (2021) considered the impacts of all interactions and potential mortalities to each potentially affected population. The potentially affected population are the Central South Pacific, the Southwest Pacific, the East Pacific, the Central West Pacific, and the East Indian West Pacific.

Sea turtle interactions in the purse seine fishery are categorized into three general interaction types: (1) direct interaction/capture during an active set (Free or FAD Set), (2) interaction and possible entanglement from FADs that are observed not associated with an active set (FAD Interaction), and (3) interactions that occur due to the presence of the vessel, small boats or helicopters used by the fishery but not associated with an active set (No-Set Interaction). Since 2008, a total of 273 sea turtles were observed captured by the United States WCPO purse seine fishery during an active set of the fishery (Table 11 and Table 12; NMFS 2021a). Most sea turtle interactions with active fishery sets occurred on Free-sets, accounting for 58%, followed by 35% of sea turtle interactions on FAD sets, and 7% were Unknown Set Type.

Table 11. Number of the different turtle species to have been captured in the United States WCPO purse seine fishery from 2008 to 2018, inclusive of all interaction types including FAD interactions (Observed and Total Estimated rows). Numbers in parentheses for the total estimated turtles are the 95th percentiles. Reported values for interaction types are the numbers of observed sea turtles (NMFS 2021a).

Interaction Type	Green*	Leatherback	Loggerhead	Olive Ridley	Hawksbill	Unidentified	Total
Observed	97	8	68	57	59	8	297
Total estimated	173 (200)	10 (20)	121 (144)	105 (118)	99 (124)	-	508 (606)
Free-Set	47	5	43	29	27	6	157
FAD Set	31	3	17	24	19	2	96
Unknown Set Type	9	-	5	1	5	-	20
Total	87	8	65	54	51	8	273

*Central South Pacific, Southwest Pacific, East Pacific, Central West Pacific, and East Indian-West Pacific green sea turtles.

Table 12. Number of sea turtles expected to interact with the United States WCPO purse seine fishery each year (mean, 95th percentile and maximum 5-year running average) and the running sum over 5 years (maximum 5-year running sum).

Species	Annual			5-Year Max Running Average of Captures	5-Year Max Running Sum of Captures
	Mean Number Captured	Upper Estimate (95th percentile)	Upper Estimated Mortality		
Hawksbill sea turtle	4	6	1 to 2	4.53	23
Leatherback sea turtle	1	1	0	0.47	3
South Pacific loggerhead sea turtle	5	7	1 to 2	4.96	25
***Olive ridley sea turtle (all species)**	4	6	0 to 1	6.29	32
****Green sea turtle (all species)**	7	9	1 to 2	7.55	38

*Endangered Mexico olive ridley sea turtle and threatened global olive ridley sea turtles (exclusive of the Mexico species)

**Central South Pacific, Southwest Pacific, East Pacific, Central West Pacific, and East Indian-West Pacific green sea turtles

Fish Aggregating Devices

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 percent of associated sets has ranged from 41-67% in the WCPO between 2007 and 2015 (NMFS unpublished data). 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). Going forward, the United States WCPO purse seine fishery anticipates no more than 1,581 of the 3,100 annual sets to be FAD sets (NMFS 2021a).

Designs for man-made FADs, have been developed with the aim of maximizing the potential catch of purse seine target species (Pilling et al. 2018). Ropes and nets associated with FADs pose an entanglement risk to sea turtles and oceanic whitetip sharks (Filmalter et al. 2013; Bourgea et al. 2014). Direct impacts of this entanglement for sea turtles depend on whether they are entangled on top of the FAD or beneath it. Sea turtles entangled on top may be injured attempting to escape or die if they are unable to escape or are not released by fishermen. Sea turtles entangled underneath the FAD may drown if they are unable to surface to breath; they may also be predated by sharks.

The WCPO purse seine fishery is moving towards the use of non-entangling and biodegradable FADs, which should reduce entanglement risk for sea turtles.

In the United States WCPO purse seine fishery, most (83%) of sea turtles observed in FAD interactions are considered uninjured (NMFS unpublished data). They were either resting on or swimming near the FAD and were startled by the presence of the vessel coming to inspect the FAD. On at least six occasions, a sea turtle was observed entangled, and either freed itself or was disentangled and freed by the fishermen and/or observer. These turtles likely experienced a higher stress level than untangled sea turtles but the response is expected to be short in duration. If the turtle was entangled for a substantial amount of time it may have suffered exhaustion and dehydration which would have sub-lethal impacts of longer duration. Two green sea turtles were reported as injured as a result of FAD interactions. One hawksbill sea turtle associated with FAD interactions was dead upon observation (Table 13). Table 14 contains the number of anticipated sea turtle mortalities associated with FADs based on 3,100 annual sets.

Table 13. Cumulative number of sea turtle mortalities for interactions with United States WCPO purse seine fishery FADs, 2008 through 2018. Mortalities assume a 0% mortality rate for uninjured sea turtles, no injured or dying sea turtles have been observed associated with FADs (NMFS 2021a).

Species	**A1 Alive and healthy**	**A2 Alive but injured**	**A3 Alive but unlikely to survive**	**D Dead**	**Total Observed Mortalities**
Green	8	2	0	0	0
Leatherback	0	0	0	0	0
Loggerhead	3	0	0	0	0
Olive Ridley	3	0	0	0	0
Hawksbill	7	0	0	1	1
Grand Total	21	2	0	1	1

Table 14. Mean and 95th percentile of sea turtle posterior estimated FAD interaction rates, anticipated exposures based on 3,100 sets per year, and anticipated mortalities for sea turtles from FAD interactions (NMFS 2021a).

	Per Set Interaction Rate		**Anticipated Exposures**		**Anticipated Mortalities**	
Species	**Mean**	**95th Percentile**	**Mean**	**95th Percentile**	**Mean**	**95th Percentile**
Green	0.00022	0.00034	2	2	0	0
Leatherback	0.00	0.00	0	0	0	0
Loggerhead	0.000066	0.00014	0	1	0	0
Olive Ridley	0.000066	0.00014	0	1	0	0
Hawksbill	0.00018	0.00029	1	2	0	1
Grand Total			3	6	0	1

Foreign WCPO Purse Seine Fisheries

There are fewer analyses of sea turtle bycatch available for the foreign purse seine fleet when compared to longline. Identification to species increased in 2006 after changes to data collection requirements changed and training in turtle identification improved. Peatman et al. (2018a) provided a summary of reported bycatch at the regional scale for purse seiners, which also includes the United States interactions (Table 15). The WCPFC (2021a) summarized bycatch estimates of key species for the WCPO purse seine fleets from annual reports. Data for sea turtles, inclusive of the United States fleet, are summarized in Table 16, and these are considered to be minimum interaction numbers.

Table 15. Median annual estimate and 95% confidence interval for median annual estimate for numbers of sea turtles captured in the WCPO purse seine fisheries from 2003 to 2017. Summarized from Table 9 in Peatman et al. (2018a).

Species	**Median (95% confidence interval) Annual Estimated Interactions**	**Total Estimated Interactions 2003 - 2017**
Green Sea Turtle	54 (39 – 68)	803
Hawksbill Sea Turtle	36 (28 – 45)	543
Leatherback Sea Turtle	7 (5 – 9)	104

Species	Median (95% confidence interval) Annual Estimated Interactions	Total Estimated Interactions 2003 - 2017
Loggerhead Sea Turtle	47 (32 – 62)	706
Olive Ridley Sea Turtle	52 (38 – 67)	784

Table 16. Numbers of sea turtles reported captured/killed by participating countries reporting catch data to the WCPFC for purse seine fisheries operating in the Action Area from 2013-2020; data were reported in 5°x 5°bins. Estimated interactions and mortalities calculated using the expansion Factor ([100/Observer Coverage] x [Observed Interactions] WCPFC 2021a.).

Species	Observed Interactions	Observed Mortalities	Estimated Interactions	Estimated Mortalities	Total Est Captures 2013 to 2020	Total Est Mortalities 2013 to 2020
Green Sea Turtle	45 (28 – 62)	1 (0 – 2)	78 (52 – 103)	2 (0 – 3)	622	12
Hawksbill Sea Turtle	29 (15 – 42)	0 (0 – 1)	50 (28 – 72)	0 (0 – 1)	401	6
Leatherback Sea Turtle	6 (4 – 7)	0 (0 – 1)	11 (8 – 13)	1 (0 – 1)	84	4
Loggerhead Sea Turtle	34 (22 – 45)	1 (0 – 1)	59 (41 – 76)	1 (0 – 2)	469	8
Olive Ridley Sea Turtle	39 (31 – 48)	1 (0 – 2)	72 (60 – 84)	2 (0 – 4)	574	14
Percent Observer Coverage (2013-2020): 54.8% (46.5 – 63.1%)						

3.3.2 Elasmobranchs

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. (2018b) for multiple regions of the WCPFC using longline observer data. Median shark and ray catch estimates were modeled by Peatman et al. (2018b) for multiple regions of the WCPFC using longline observer data. Table 17 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 17 is an excerpt from Peatman et al. (2018b). 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. 2018b). 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 and scalloped hammerhead sharks from 2013 to 2018 (Table 18).

Table 17. Median oceanic whitetip shark catch estimates (‘000 individuals) between South of 10°S (Peatman et al. 2018b).

Oceanic Whitetip Shark (reported in ‘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 18. Mean and (95% confidence interval) of annual numbers of oceanic whitetip and scalloped hammerhead sharks 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 2018 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 2021a).

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
Scalloped Hammerhead Shark	15 (8 – 22)	7 (3 – 10)	570 (232 – 908)	251 (101 – 400)	4,561	2,004
Hammerhead Shark (spp. not identified)	7 (5 – 9)	2 (1 – 3)	279 (191 – 367)	68 (35 – 100)	2,232	541
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 19).

Table 19. 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. (2018b), 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. 2018b). Observer coverage over the whole Convention Area (Figure 27) 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. 2018b). 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.

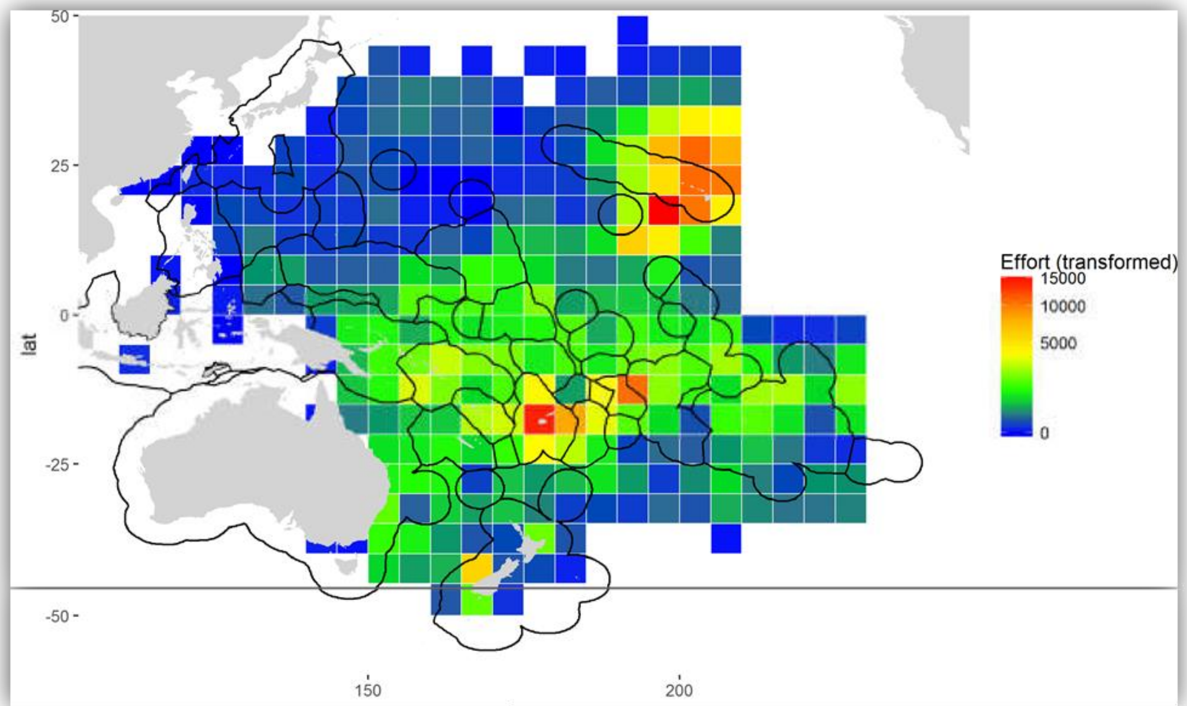

Figure 27. Observed effort in number of hooks (square root transformed) for longliners during the 2003-2017 time period in the WCPFC convention area (Peatman et al. 2018b).

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 28, the total number of manta and mobula ray interactions documented by observers from 2003-2017 totals 1,800.

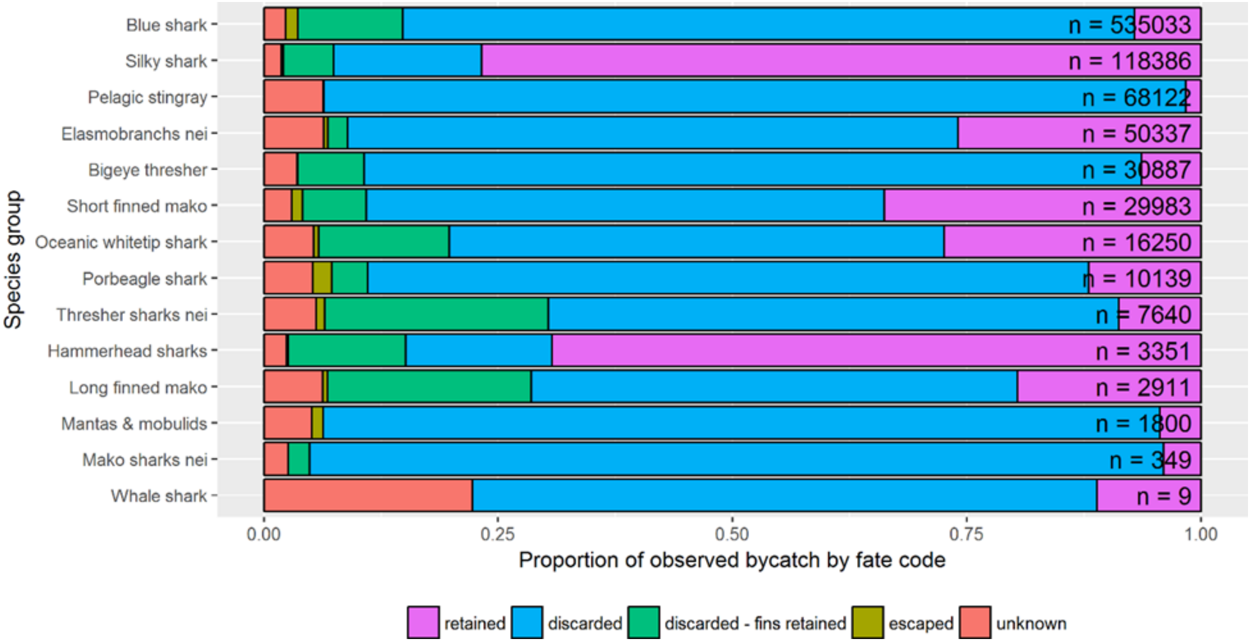

Figure 28. Recorded fate of observed sharks and rays catch by species/species group, as a proportion of total observed catch (number of fish) for the species/species group in the longline fisheries. The number of records is provided (Peatman et al. 2018b).

As evident in the above figure, very few mantas or mobulas were retained, the majority captured were discarded. In Figure 29 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).

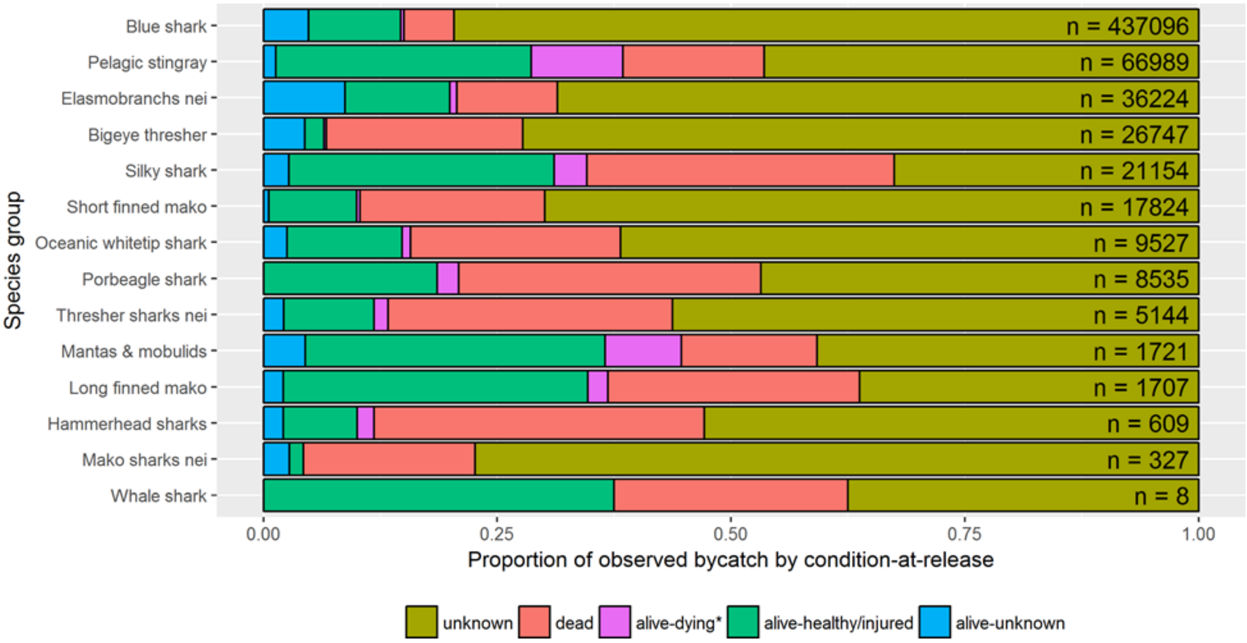

Figure 29. Recorded condition at release of observed sharks and rays catch by species/species group, as a proportion of total observed catch (number of fish) for the species/species group in the longline fisheries. The number of records is provided for each species/group. Note – alive-dying* is individuals that were alive but considered unlikely to survive (Peatman et al. 2018b).

United States WCPO Purse Seine

The observed interactions, mortalities and anticipated number of interactions and mortalities of oceanic whitetip sharks, Indo-West Pacific scalloped hammerhead sharks, and giant manta rays that occurred from 2008-2015 in United States WCPO purse seine fishery are listed below by species. 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 of sea turtles interacting with the United States WCPO purse seine fishery from 2008 to 2018 for oceanic whitetip sharks, 2012 to 2018 for Indo-West Pacific scalloped hammerhead sharks and 2010 to 2018 for giant manta ray.

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 20 below lists the number of oceanic whitetip interactions and mortalities and Table 21 lists the anticipated number of exposures and mortalities.

Table 20. Observed interactions of oceanic whitetip sharks described by the observer data from 2008-2018 (NMFS 2021a).

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	2,284	925

Table 21. 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

Indo-West Pacific Scalloped Hammerhead Sharks

Between 2008 and 2018, a total of 14 Indo-West Pacific scalloped hammerhead sharks were observed in the United States WCPO purse seine fishery (Table 22). No interactions were recorded from 2008 to 2012 and may not have been reliably identified to species before 2012 (D.

O'Brien pers. comm. 2019; NMFS 2021a). Table 23 lists the anticipated exposures and mortalities for the Indo-West Pacific scalloped hammerhead shark.

Table 22. Observed interactions of Indo-West Pacific Scalloped Hammerhead sharks described by the observer data from 2008-2018 (NMFS 2021a).

Year	**Observed Interactions**	**Mortalities**
2009	0	0
2010	0	0
2011	0	0
2012	3	3
2013	2	2
2014	4	4
2015	2	2
2016	1	1
2017	1	1
2018	1	1
Total	14	14

Table 23. Maximum 5-year running average and 95th percentile of scalloped hammerhead shark posterior estimated exposure and mortality in terms of anticipated annual interactions with fishing gear (NMFS 2019c, 2021).

	Maximum 5-year Running Average	**95th percentile**
Predicted Annual Exposures	2.0	2
Predicted Mortality Estimates	2.0	2

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 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 24). 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 25).

Table 24. Number of estimated interactions* with giant manta rays from 2008 to 2018 with estimated number of mortality events based on the observed data (NMFS 2021a).

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 25. 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 2021a).

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 which are most likely a result of decreased number of FADs and may be indicative of population declines in the area (Peatman et al. 2017). Since 2009, bycatch levels have been relatively low and stable but with slightly increased catch estimates from 2018 to 2020 (Peatman and Nicol 2021; Figure 30).

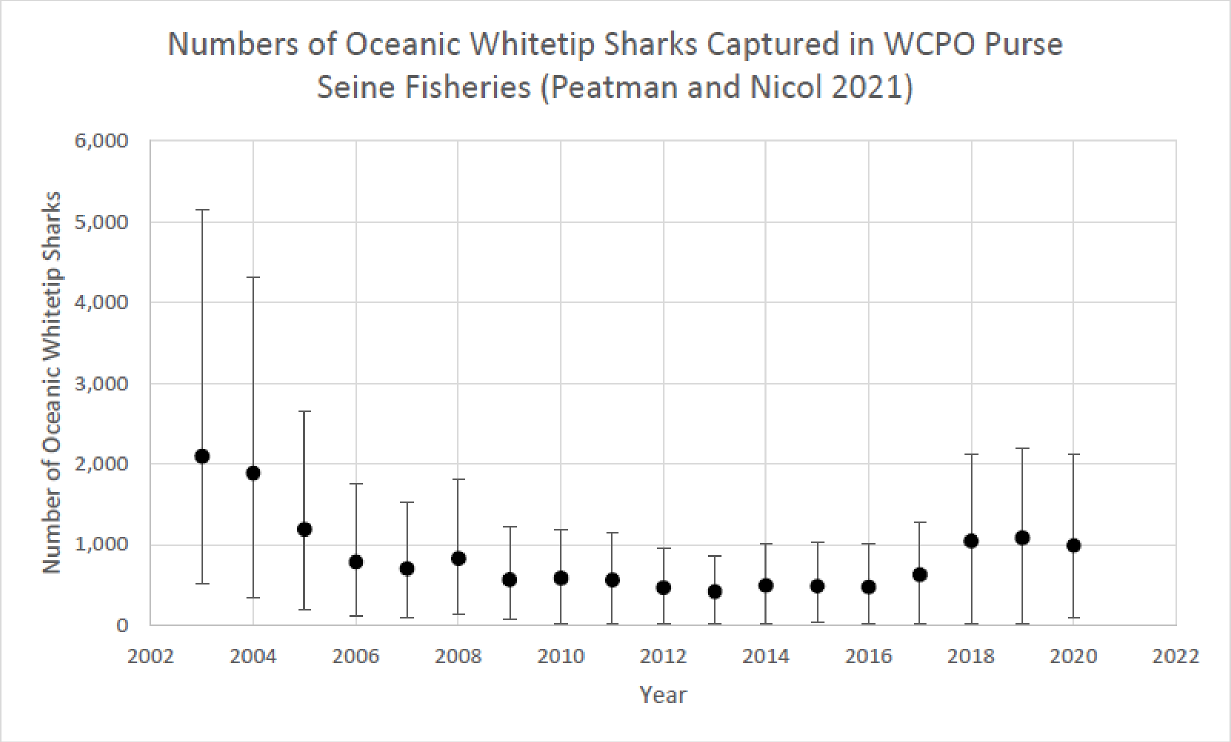

Figure 30. Predicted total annual oceanic whitetip bycatch (numbers) by year for large-scale purse seine fleets between 2002 and 2020. Ninety-five percent confidence intervals (95% CI) shown in boxplot whiskers. Source: Peatman and Nichol 2021.

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.

Scalloped hammerhead sharks are not common in the purse seine fisheries. We accessed the public domain bycatch data served by the WCPFC to estimate the numbers of annual interactions and mortalities of both oceanic whitetip sharks and scalloped hammerhead sharks from 2013 to 2017 (Table 26). Based on how the data is amalgamated by the WCPFC, United States data is presumed to be present although not specifically identified as such.

Table 26. Mean and (95% confidence interval) of annual numbers of scalloped hammerhead sharks reported captured/killed by participating countries reporting catch data to the Western and Central Pacific Fisheries Commission for purse seine fisheries operating in the Action Area from 2013 to 2018; 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 2020).

Species	**Observed Interactions**	**Observed Mortalities**	**Estimated Interactions**	**Estimated Mortalities**	**Total Est Captures 2013 to 2020**	**Total Est Mortalities 2013 to 2020**
Oceanic Whitetip Shark	435 (300 – 570	75 (30 – 120)	798 (610 – 986)	135 (65 – 204)	6,383	1,078
Scalloped Hammerhead Shark	14 (7 – 21)	0 (0 – 0)	27 (15 – 40)	0 (0 – 0)	217	0
Hammerhead Shark (spp not identified)	3 (0 – 5)	0 (0 – 0)	1 (0 – 3)	0 (0 – 0)	9	0
Percent Observer Coverage (2013-2018): 54.8% (46.5 – 63.1%)						

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 2019). 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 ?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 some species like the oceanic whitetip shark or Indo-West Pacific scalloped hammerhead shark 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 and as some ESA-listed species populations recover and populate new areas or areas where they were previously extirpated (Swingle et al. 1993; Wiley et al. 1995). As vessels continue to become faster and more widespread, an increase in vessel interactions is expected.

Sea turtles must surface to breathe and several species are known to bask at the surface for long periods making them more susceptible to ship strikes. Ship strikes have been identified as one of the important mortality factors in several nearshore turtle habitats worldwide (Denkinger et al. 2013). However, available information is sparse regarding the overall magnitude of this threat or the impact on sea turtle populations globally. Although sea turtles can move somewhat rapidly, they apparently are not adept at avoiding ships that are moving at more than 4 km per hour; most ships move far faster than this in open water (Hazel and Gyuris 2006; Hazel et al. 2007; Work et al. 2010). Hazel et al. (2007) suggests that green turtles may use auditory cues to react to approaching ships rather than visual cues, making them more susceptible to strike as ship speed increases. Since turtles that were previously killed or injured as a result of some other stressor (e.g., fishing net entanglement or disease) may be more susceptible to a ship strike, it is not always known what proportion of ship wounds were sustained ante-mortem versus post mortem (or post injury).

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 Indo-Pacific scalloped hammerheads 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 27 as reported by the USCG (2013) and supplemented as noted. Where available, information on the impacts to natural resources is summarized.

Table 27. 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)
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.

Date	Spill Name/Location	Oil Type and Volume	Natural Resource Impacts
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 does 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, polycyclic aromatic hydrocarbons (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; Balcioˇglu 2016; Zychowski and Godard-Codding 2017). PAHs can bioaccumulate in prey items leading to toxic effects in those species (Moore et al. 1989) which affect sea turtles (Camacho et al. 2012). Some PAHs provide serious adverse effects in the marine environment over 17 months after an oil spill event (Perez et al. 2008) and the EPA designated 16 PAHs as High Priority Pollutants because of their potential toxicity, prevalence, and persistence within the environment (40 CFR Part 423, Appendix A; Hussar et al. 2012). Therefore, PAHs are also considered Persistent Organic Pollutants, or POPs. PAHs appear to show greater partitioning and storage in tissues than other categories of POPs with some species at greater risk of long-term bioavailability although varying risks given life stage and sex (Munoz et al. 2021).

Additionally, POPs are maternally transferrable in some species, like sea turtles, and affect neonates (Ehsanpour et al. 2014; De Andres et al. 2016; Munoz and Vermenien 2020 and references cited therein). Barraza et al. (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 sea turtle populations (Camacho et al. 2012). Hutchinson and Simmonds (1992) suggested fibropapillomatosis may also be influenced by PAH exposure in sea turtles while long-term effects from chronic PAH exposure can result in cancers (Collins et al. 1998; Camacho et al. 2012). However, data linking PAHs and POPs to cancers in elasmobranchs is lacking.

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

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

Studies on exposure to oil and dispersed oil in the available literature have mixed results from no effects to observed effects. Stacy (2012) and Stacy et al. (2017) did not observe effects noted in the literature by previous authors (Vargo et al. 1986; Lutcavage et al. 1995; Camacho et al. 2013; Harms et al. 2014). Yet later studies better address exposure to dispersants which resulted in statistically significant alterations for 15 different hematological variables associated osmotic, electrolyte, mineral, and hydration; which ultimately affected hatchling weights (Harms et al. 2019). Shaver et al. (2021) on the other hand documented increased abnormalities in embryos and nesting trends in the same population assessed by Stacy (2012) and Stacy et al. (2017). Additionally, Harms et al. (2019) noted indications of a hypothalamic-pituitary-adrenal stress response and potential hepatocellular toxicity. The authors discuss that some results were inconsistent with previous findings, particularly in relation to anemia and hematological assessment methods used by previous authors. Ultimately, Harms et al. (2019) suggests anemia is likely a result of chronic exposure to dispersants and chemically dispersed oil. Three values suggested considerable concern for morbidity or potential pending mortality including uric acid, potassium, and phosphorus (Harms et al. 2019). Exposure to sea turtle hatchlings would be less likely given the number of nests expected to occur in American Samoa during any given year when compared to adults in the Region.

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

An elasmobranch or sea turtle's response to oil or contaminant exposure will be dependent on a large number of variables. However, length of exposure, concentration of the contaminant(s), and the specific type of 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).

Marine debris is a significant concern for sea turtles and fish. The early developmental stages of all turtle species are spent in the open sea. During this time both juvenile turtles and their buoyant food are drawn by advection into fronts (convergences, rips, and drift lines). The same process accumulates large volumes of marine debris, such as plastics and lost fishing gear, in ocean gyres (Carr 1987). An estimated four to twelve million metric tons of plastic enter the oceans annually (Jambeck et al. 2015. It is thought that some sea turtles eat plastic because it closely resembles jellyfish, a common natural prey item (Schuyler 2014. Ingestion of plastic debris can block the digestive tract which can cause turtle mortality as well as sub-lethal effects including dietary dilution, reduced fitness, and absorption of toxic compounds (Lutcavage et al. 1997; Laist et al. 1999).

Santos et al. (2015) found that a small amount of plastic debris was sufficient to block the digestive tract and cause death in sea turtles. They reported that 10.7% of green turtles in Brazilian waters were killed by plastic ingestion, while 39.4% had ingested enough plastic to have killed them. These results suggest that debris ingestion is a potentially important source of turtle mortality, one that may be masked by other causes of death. Gulko and Eckert (2004) estimated that between one-third and one-half of all sea turtles ingest plastic at some point in their lives.

A more recent study by Schuyler et al. (2015) estimates that 52% of sea turtles globally have ingested plastic debris. Schuyler et al. (2015) synthesized the factors influencing debris ingestion by turtles into a global risk model, taking into account the area where turtles are likely to live, their life history stage, the distribution of debris, the time scale, and the distance from stranding location. They found that oceanic life stage turtles are at the highest risk of debris ingestion. Based on this model, olive ridley turtles are the most at-risk species; green, loggerhead, and leatherback turtles were also found to be at a high and increasing risk from plastic ingestion (Schuyler 2014). The regions of highest risk to global turtle populations are off the east coasts of the United States, Australia, and South Africa, the East Indian Ocean, and Southeast Asia.

In addition to ingestion risks, sea turtles can also become entangled in marine debris such as fishing nets, monofilament line, and fish-aggregating devices or FADs (NRC 1990a, 1990b; Lutcavage et al. 1997; Laist et al. 1999). Turtles are particularly vulnerable to ghost nets due to their tendency to use floating objects for shelter and as foraging stations (Kiessling 2003; Dagorn et al. 2013).

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

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

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

3.6 Anthropogenic Sound

The ESA-listed species 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). The species considered in this opinion 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).

ESA-listed species 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 sea turtles, elasmobranchs, and cetaceans, may result in short-term behavioral or physiological responses (e.g., avoidance, stress) but is not quantifiable in the Action Area at this time.

ESA-listed sea turtles may exhibit a variety of different responses to sound fields associated with seismic airguns and echosounders. Avoidance behavior and physiological responses from airgun exposure may affect the natural behaviors of sea turtles (McCauley et al. 2000).

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 training exercises are conducted in the Hawaii-Southern California Training and Testing Study Area or the Mariana Islands Training and Testing Area which neither 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, cracked or crushed carapaces, amputations, or other broken body parts; any of which could result in an animal's death. Air-breathing turtles and mammals could be injured in a manner that impairs their ability to reach the surface, and subsequently drown (Duronslet et al. 1986). 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 effects to ESA-listed species are not expected as aircraft would be at altitude where auditory effects 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 2019e, 2019f).

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 of all proposed federal projects in the Action Area that have already undergone formal or early consultation, and state or private actions that are contemporaneous with this consultation. The baseline conditions affecting species include global climate change, fisheries and fisheries bycatch, vessel strikes, pollution from chemicals and marine debris, ocean noise from various 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 or endangered, and/or ability of species to recover.

- Oceanic Whitetip Shark- Overutilization is a listing factor for oceanic whitetip shark 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).
- Pacific Leatherback Sea Turtle- The greatest of these threats worldwide are incidental capture in fishing gear and harvest of leatherback eggs and adults. The Pacific leatherback populations are most at-risk for extinction (NMFS 2020a) https://www.fisheries.noaa.gov/species/leatherback-turtle.
- Green Sea Turtle - The primary threats facing green turtles are bycatch in commercial and recreational fishing gear, direct killing of turtles and harvest of eggs, vessel strikes, loss and alteration of nesting habitat, degradation and loss of foraging habitat, and entanglement in or ingestion of marine debris (NMFS 2020a) https://www.fisheries.noaa.gov/species/green-turtle.
- Olive Ridley Sea Turtle – The primary threat and factor for decline is harvest of eggs or killing of nesting adults. Incidental capture in fishing gear—primarily in longlines and trawls, but also in gill nets, purse seines, and hook and line—is a serious ongoing source of mortality that adversely affects the species' recovery. Also, vessel strikes can injure or kill sea turtles. Injuries or death may be caused by propellers and blunt force trauma from the vessel’s hull (NMFS 2020a) https://www.fisheries.noaa.gov/species/olive-ridley-turtle.
- Hawksbill sea turtle – the main threat is habitat loss due to costal development; however, one of the major threats to sea turtles is getting caught in fishing gear and drowning (bycatch). Sea turtle bycatch is a worldwide problem. Hawksbills are primarily a coastal species so nearshore coastal fisheries tend to be their greatest threat (NMFS 2020a) https://www.fisheries.noaa.gov/species/hawksbill-turtle.
- Indo-west Pacific Scalloped Hammerhead Shark - Factors supporting its status as ‘threatened’ include overutilization, inadequacy of existing regulatory mechanisms and other factors, specifically: (1) Decreases in CPUE of sharks off the coasts of South Africa and Australia and in longline catch in Papua New Guinea and Indonesian waters, suggesting localized population declines, (2) high susceptibility to overfishing, especially given its schooling behavior, in artisanal fisheries and industrial/commercial fisheries; (3) high at-vessel mortality rate associated with incidental capture in fisheries (resulting in further reduction of population productivity and abundance); (4) popularity of the species in the shark fin trade; and (5) inadequate regulatory mechanisms and/or weak enforcement of current regulations in many areas, resulting in frequent reports of illegal fishing of the species and contributing to the further decline of this species. Therefore, we the Indo-West Pacific scalloped hammerhead shark was listed as threatened under the ESA (79 FR 38213; Sept 2, 2014).

- 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 2020a) 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 ESA-listed species, which we present below, but the greatest concern is for leatherback sea turtles. The western Population is anticipated to fall below threshold population levels after 100 years with 100% certainty (Martin et. al 2020). Leatherbacks face a high probability of extinction as a result of both environmental and demographic stochasticity. Demographic stochasticity, or chance variation in the birth or death of an individual of the population, is facilitated by the increases in mortality rates of leatherback populations resulting from the premature deaths of individual sea turtles associated with human activities (or that die as a result of being captured in fisheries) or incidental capture and mortality of individuals in various fisheries.

The information available suggests that leatherback sea turtles have a high probability of becoming extinct in the Pacific Ocean unless they are protected from the combined threats of entanglement in fishing gear and overharvest. The number of individuals that continue to be captured and killed in fisheries in the Action Area contributes to the increased extinction risk of the species. Martin et al. (2020) provided an analysis of the combined effects of the three United States fisheries on two sea turtle populations, evaluating the influence on the abundance and productivity of each. They determined that "take" associated with the three fisheries has "little impact" on the timing at which key reductions in populations will be reached, based on a review of the projected 10-year annual nester abundance projections. The combined reduction from the fisheries is 1.6[6] annual nesters at 10 years, of which 0.5 is attributed to the ASLL fishery, 0.9 from the Hawaii DSLL fishery, and 0.2 from the Hawaii SSLL fishery. While this combined effect of the baseline is small, it is not zero. We can reasonably interpolate that if the proposed action were not to proceed, the number of adult nesters at the end of the 10 year could be greater by 1 fecund member.

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 leatherback sea turtles. However, leatherback sea turtles occur in the pelagic waters within the Action Area where their density is sparse in comparison to nearshore turtle habitats where as a result of a higher density of turtles, 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 in the pelagic environment but in nearshore areas the risk of vessel strike likely increases.

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[6] Animals cannot be reduced by a fraction. For the purpose of evaluating this reduction relative to conservation goals, we use standard rounding, where if the number is greater than 0.5 we round up to the next whole number. In this case the reduction at 10 years would be 2 WP leatherback turtles.

Of the different types of pollution that can adversely affect leatherback sea turtles, the ingestion and entanglement risk of marine debris may pose the most serious threat. Ingestion of plastic debris can block the digestive tract which can cause turtle mortality as well as sub-lethal effects including dietary dilution, reduced fitness, and absorption of toxic compounds (Lutcavage et al. 1997; Laist et al. 1999). Entanglement in marine debris could result in substantial wounds, including cuts, constriction, or bleeding on any body part. In addition, entanglement could directly or indirectly interfere with mobility, causing impairment in feeding, breeding, or migration. 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.

We understand little about the impacts of noise or the underwater noise thresholds for sea turtles. However, their hearing ranges are generally limited and sea turtle ears are considered more primitive than that of marine mammals and less capable of detecting sound pressures (Popper et al. 2014). Since leatherback sea turtles in the pelagic waters within the Action Area are generally moving and their exposure to a moving source of sound would be relatively short lived, ocean noise in the Action Area is not expected to contribute to the increased extinction risk of the species.

The stress regime created by activities discussed in this Environmental Baseline continue to be a threat for the Central South Pacific, Central West Pacific, Southwest Pacific, East Pacific, and East Indian-West Pacific green sea turtles. Threats associated with climate change related to loss of nesting habitat, vessel strikes, and marine pollution pose a risk to these species. Even with increasing populations in some green sea turtle species, a risk of extinction due to environmental stochasticity and the threats from fishing gear and vessel strikes contribute to the risk. They are most likely impacted to a greater extent further away from the Action Area.

The stress regime created activities discussed in this Environmental Baseline are also a threat for the giant manta ray. 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 high a 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 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 tract (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 stressors discussed in this Environmental Baseline are also a threat for the oceanic whitetip and Indo-West Pacific scalloped hammerhead sharks. Both the oceanic whitetip and the Indo-West Pacific scalloped hammerhead 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 and Indo-West Pacific scalloped hammerheads suffer a very high at-vessel mortality rate in most fisheries, particularly longlines. The information available suggests that both species 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 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 and Indo-West Pacific scalloped hammerhead 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 only 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 of the endangered and 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 endangered and 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 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 exclude data from 2010 and 2011. For the mean and 95th percentile numbers we rely on the anticipated number of interactions estimated by McCracken (2019a) from 2012-2017 and for the maximum 5-year running average and sum we rely on data from 2012 to 2019.

4.2 Potential Stressors

Potential stressors associated with the proposed action include:

1. Interaction with including capture of non-target species, such as listed species, or their prey;
2. derelict gear;

3. introduction of oily discharges, cardboard, plastics, and other waste into marine waters;
4. collisions with vessels;
5. vessel noise; and
6. vessel emissions.

We determined that vessel noise, vessel collisions, introduction of discharges and other wastes, derelict gear, and vessel emissions are not likely to adversely affect ESA-listed resources (See Appendix A, Stressors Not Likely to Adversely Affect Listed Resources). 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.

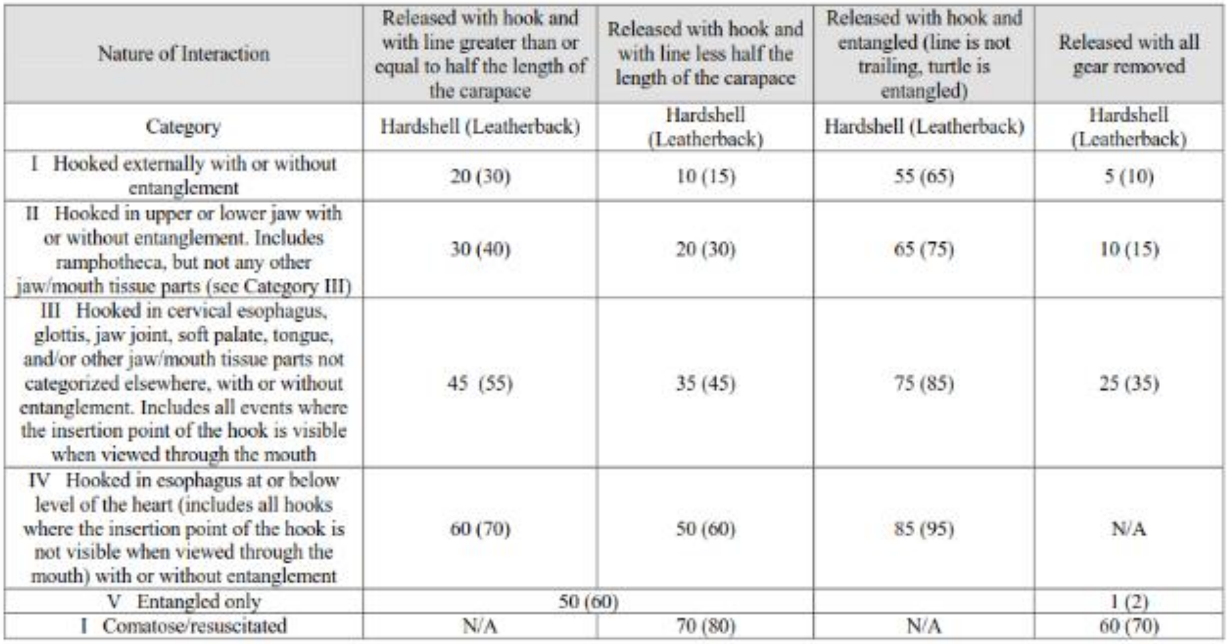
4.3 Summarized Responses of ESA-listed Species to the Fishery

The most significant hazard the ASLL fishery presents to listed species results from hooking and entanglement by gear, which can injure or kill sea turtles and elasmobranchs. If hooked or entangled, air-breathing species can drown after being prevented from surfacing for air; alternatively, all listed species that are hooked or entangled, but do not immediately die from their wounds 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 captured in longline fisheries, long-term effects are nearly impossible to monitor; therefore, a quantitative measure of the effect of longlining on sea turtle, Indo-west Pacific scalloped hammerhead shark, oceanic whitetip shark, and giant manta ray populations is very difficult. Even if listed species are not injured or killed after being entangled or hooked, these interactions can be expected to elicit stress responses that can have longer-term physiological or behavioral effects. The following discussion summarizes the information on how sea turtles, Indo-west Pacific scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays are likely to respond to these interactions with fishing gear.

For sea turtles, we used Ryder et al. (2006) to assign post-interaction mortality rates to sea turtles that were alive when they were brought alongside the fishing vessel. Ryder et al. (2006) assign post-interaction mortality rates based on the location of the hook, whether the turtle was released with or without the hook, the amount of trailing line remaining at release, and whether or not the turtle was entangled. Table 28 specifies the release conditions and hook locations for the associated post-interaction mortality rates.

Table 28. Post-interaction mortality rates for sea turtles interacting with longline fisheries from Ryder et al. (2006)*.



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

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

4.3.1 Entanglement in Longline Gear

Sea Turtles

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

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

Sharks

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

Giant Manta Ray

Similar to sea turtles, when giant manta rays interact with longline gear, they are particularly prone to being entangled in fishing gear because of their body configuration and behavior. The giant manta ray tends to be more vulnerable to entanglement and foul hooking as opposed to being hooked in the mouth (Sales et al. 2010; Domingo et al. 2012). If entangled in a monofilament 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 are so similar they 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, perished and were immediately consumed by sharks. Although mooring lines are not used in this fishery, the material is similar to polypropylene float line, which can entangle these animals, and shows that predators are quick to take advantage of an entangled animal.

4.3.2 Hooking

Sea Turtles

In addition to being entangled in a longline, sea turtles are also injured and killed by being hooked. Sea turtles are either hooked externally - generally in the flippers, head, beak, or mouth - or internally, where the animal has attempted to forage on the bait, and the hook is ingested into the gastrointestinal tract, often a major site of hooking (E. Jacobson in Balazs et al. 1995b). Hooking can occur as a result of a variety of scenarios, some of which will depend on foraging strategies and diving and swimming behavior of the various species of sea turtles. For example, necropsied olive ridleys have been found with bait in their stomachs after being hooked; therefore, they most likely were attracted to the bait and attacked the hook. Similarly, a turtle could concurrently be foraging in or migrating through an area where the longline is set and could be hooked at any time during the setting, hauling, or soaking process.

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

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

Sharks

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

As with other marine species, even if the hook is removed, which is often possible with a lightly hooked shark, the hooking interaction is believed to be a significant event. As previously mentioned, 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)

Sea Turtles

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

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

Sharks

Excessive trailing gear could directly or indirectly interfere with 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. Currently, no CMM's are specific for Indo-west Pacific scalloped hammerhead sharks.

Giant Manta Ray

Given their size, giant manta rays are seldom boarded, and similar to leatherback sea turtles, 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 threatened and endangered species that we previously concluded are likely to be adversely affected by the fishery. For our exposure analyses we estimate the number, age (or life stage), and gender of the individuals that are likely to be exposed to stressors associated with the fishery, if the data are available in sufficient detail.

McCracken (2019a) projected the probable number of the different species that are likely to be incidentally captured in the ASLL fishery in the future based on the number and rate of individuals 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. As noted in Section 1.5.3, we calculated maximum 5-yr. running averages and running sums based on the annual point estimates of McCracken (2019a), and McCracken and Cooper (2020a, 2020b). The following narratives include tables that present estimates of the number of the different species that are likely to be captured in the fishery based on these metrics.

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 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.

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

Interpretation of the intervals presented in these analyses. All of the species' 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 have the potential to occur, although those at the edges have lower likelihoods than those towards the center of the interval. The point estimate (in our case, the mean value) is the value that best describes anticipated effects over a longer term.

For example, Table 30 presents annual estimates of the number of green sea turtles that are likely to be incidentally captured in the ASLL fishery (annual mean = 14; 95th percentile = 45). The estimates in Table 30 should be interpreted as "the data suggest that the number of green sea turtles incidentally captured in any given year should average about 14 and rarely will be higher than 45; over any 5-year period we expect the average to be no more than 19.2 green sea turtles with up to 96 over 5 years."

The upper limits of 95% confidence intervals, or 95th percentile, are important because they are compatible with the data (although less so than the mean), represent greater risks for endangered and threatened species. Although these upper limits will always be higher than the average values, they 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.

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

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

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

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

which can be solved for population growth rate as:

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

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

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

4.4.1 Green Sea Turtle

Exposure Analysis

Five DPSs of green sea turtles occur in the Action Area for this consultation. Although the nesting distribution of these DPSs is discrete, their distribution overlaps in the open ocean and in some coastal areas. As a result, genetic analyses from tissue samples collected from green turtles captured in the fishery are required to determine which DPS of green turtle an individual turtle represents and, in some cases, the genetic data do not allow us to assign a particular tissue sample to one green turtle DPS. Thus, assigning green turtles at time of capture to a particular DPS cannot be done with reasonable certainty as genetic analyses of tissue samples cannot occur boatside. The exposure and response analyses that follow treat green turtles as a group, we consider the potential impacts to the five individual DPSs in the Risk Assessment section.

Data have been collected by observers in the ASLL since 2006; however, observer coverage was less than 10% from 2006 to 2009, after which it was around 20%. Therefore, we focus our assessment of expected level of impact to the years 2010 to 2019, but use the full range of data from 2006 to 2019 for demographic assessments of genetic composition, length and mortality.

Green sea turtles have been captured in the ASLL fishery in 9 of the 10 years between 2010 and 2019. During this 10-year time interval fishery observers reported that 37 green sea turtles were captured in the ASLL fishery. When these reports are adjusted to account for the percentage of observer coverage, about 214 green sea turtles are likely to have been captured in the fishery in the 10-year period from 2010 to 2019 (Table 29; McCracken 2019a, 2020a; McCracken and Cooper 2022).

Table 29. Number of green sea turtles observed and estimated captured from 2006 to 2019 (McCracken 2019a, 2020a; McCracken and Cooper 2022).

Year	% Observer Coverage	Observed	Estimated
2006	-	3	-
2007	-	1	-
2008	-	1	-
2009	-	3	-
2010	25.0%	8	50
2011	33.0%	9	32
2012	19.8%	0	0
2013	19.4%	3	19
2014	19.4%	1	17
2015	22.0%	0	0
2016	19.4%	4	17
2017	20.0%	4	22
2018	17.5%	4	20
2019	15.7%	4	26
Grand Total		45	214

Demographic Patterns of Exposure

Based on genetic analyses of 31 tissue samples collected from green turtles captured in the ASLL fishery (Dutton, personal communication, 2018), the green turtles captured in the fishery represent the following DPSs of green turtle that have been listed as endangered or threatened: (1) Central West Pacific; (2) Central South Pacific; (3) East Pacific; (4) East Indian-West Pacific; (5) Southwest Pacific green turtles. NMFS conducted mixed stock analyses of 31 genetic samples of green turtles captured in the ASLL fishery between 2006 and 2017 (P. Dutton, PIFSC, Jun 29, 2018). However, the proportions assigned to each green sea turtle DPS have a high level of uncertainty with large confidence intervals due to the low sample size. Because the certainty of the genetic data, in terms of assigning proportions of captures to green sea turtle DPSs is so low, we cannot be reasonably certain that all captures will not come from one green turtle DPS. While genetic evidence may be collected at the time of an animals capture, analysis of those samples cannot be completed immediately. Therefore, we focus our Risk Analysis on the possibility that all green sea turtle captures may come from any one green turtle DPS by analyzing as if 100% of the captures comes from each. While this is a conservative assumption, the best scientific and commercial data available do not allow us to make more reliable estimates. Furthermore, it is impossible to know the source DPS for any individual green sea turtle at time of capture and we are recognizing that any captured green sea turtle may be from any one of the five affected DPSs.

Sizes of green sea turtles captured by the ASLL fishery ranged from 24 to 60.5 cm SCL (Figure 31). The minimum size of nesting green sea turtles in Hawaii is 75 cm SCL (Balazs et al. 2015), therefore green sea turtles interacting with the fishery are juveniles. Most captured turtles were between 36 and 45 cm SCL.

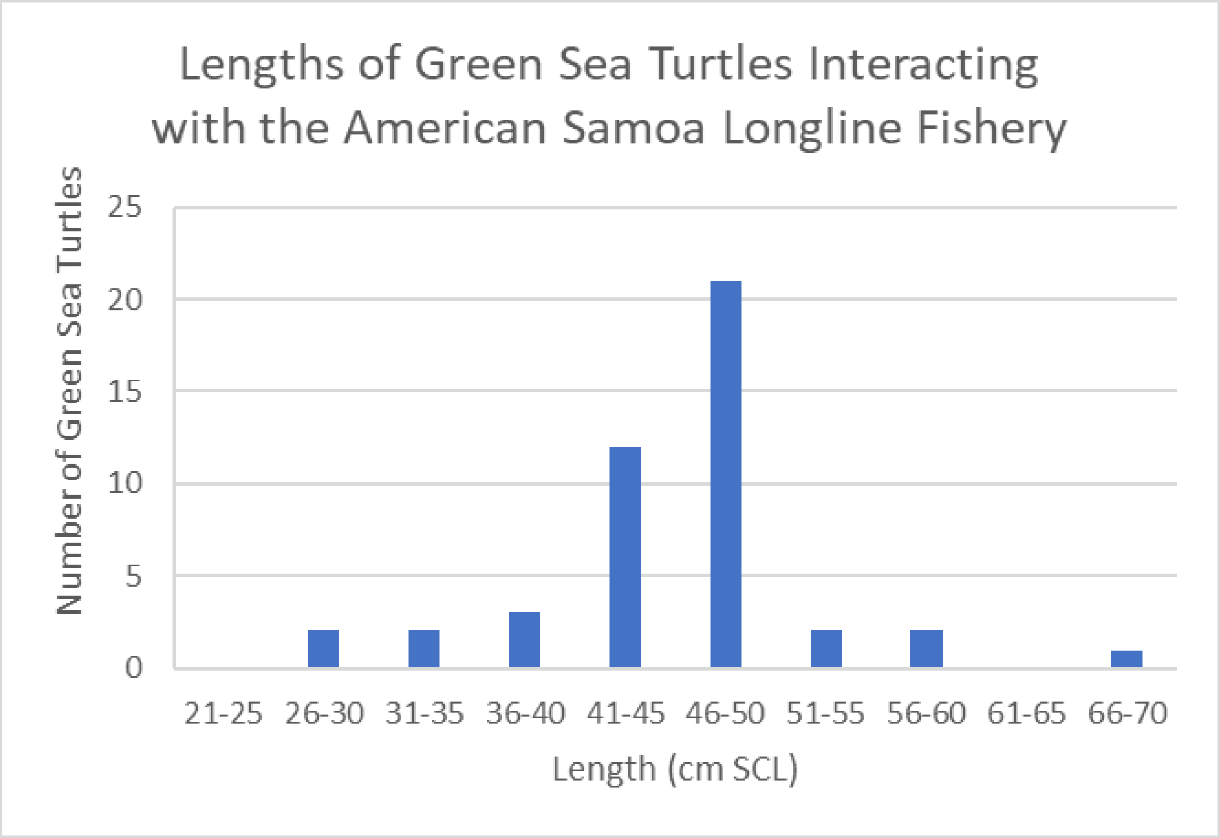


Figure 31. Lengths of observed green sea turtles interacting with the ASLL fishery from 2006 to 2019 (n=45).

Spatial and Temporal Patterns of Exposure

We plotted 43 green sea turtle interactions that occurred in the ASLL fishery between 2006 and 2019 (2nd quarter) in the geographic information system ArcGIS. Due to the small area of the American Samoa EEZ and the number of turtle interactions observed since 2006, we were able to aggregate these positions in ArcMap using a kernel density tool within the confines of NOAA Administrative Order 216-100 on Protection of Confidential Fisheries Statistics.

The ArcMap kernel density tool was used to calculate a density which is based on point features using interaction and haul data. This function allows for the creation of a heat map, which illustrates the relative number of interactions or effort in an area corresponding to those particular values. According to ESRI, the company that created and administers the ArcGIS program, 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 more fluid edges. As the 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 than those areas with orange or reds. The key was adjusted to illustrate low versus high density values.

As evident in Figure 32 below, the majority of observed interactions with green sea turtles occurs in the northern half of the EEZ followed by observed interactions in the southwest corner. Although vessels in the ASLL can fish in international waters and in the EEZ of neighboring countries (i.e., Cook Islands) under an access agreement, the majority of these vessels choose to fish within the American Samoa EEZ. Not surprisingly, the area of observed interactions with green sea turtles resembles the observed effort as demonstrated in Figure 33. However, the number of observed green sea turtle interactions appears to be relatively higher in the southwest corner when compared to fishing effort (Figure 34).

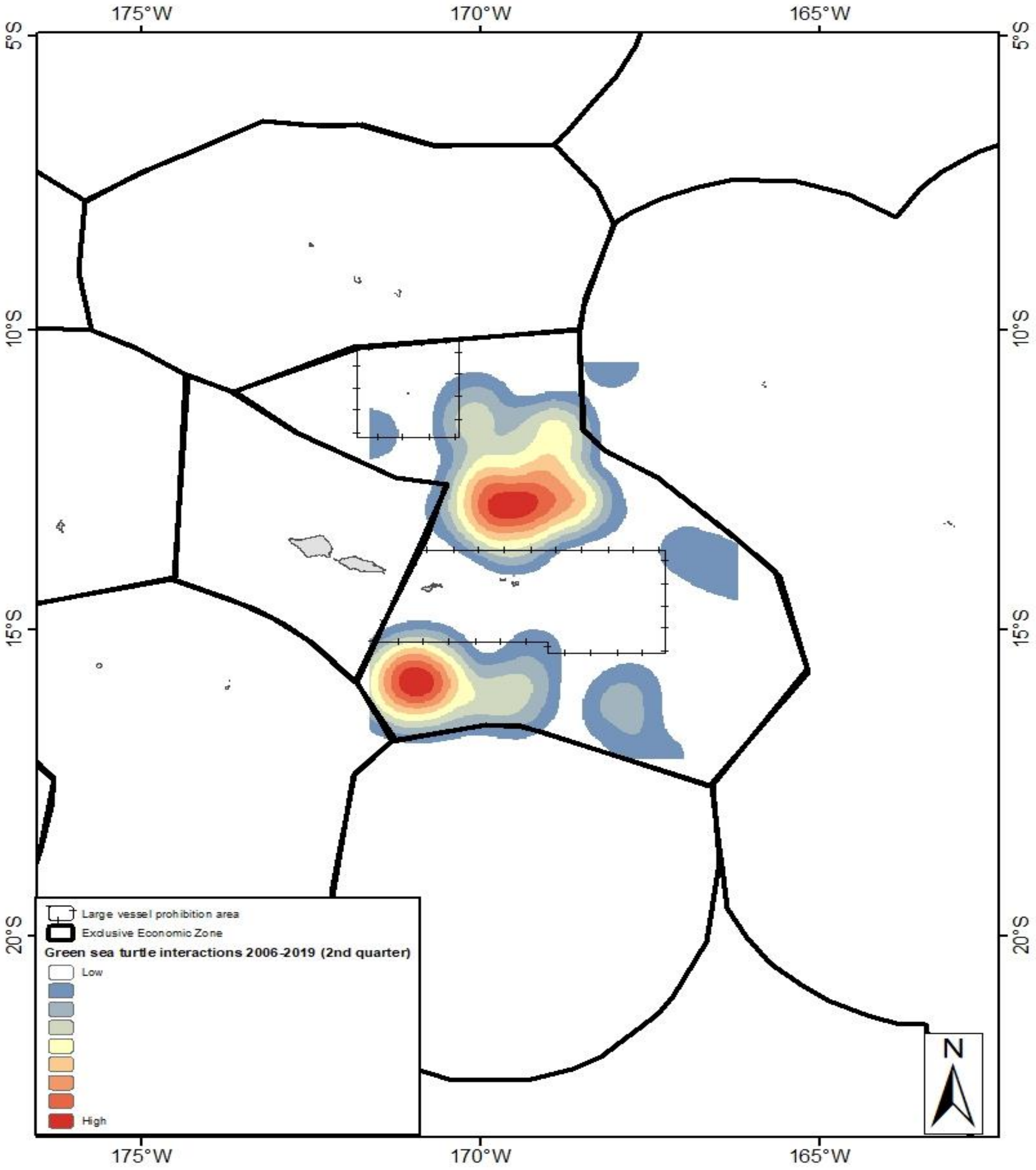


Figure 32. Geographical representation of observed aggregate green sea turtle interactions with the ASLL fishery between 2006 and 2019 (2nd quarter).

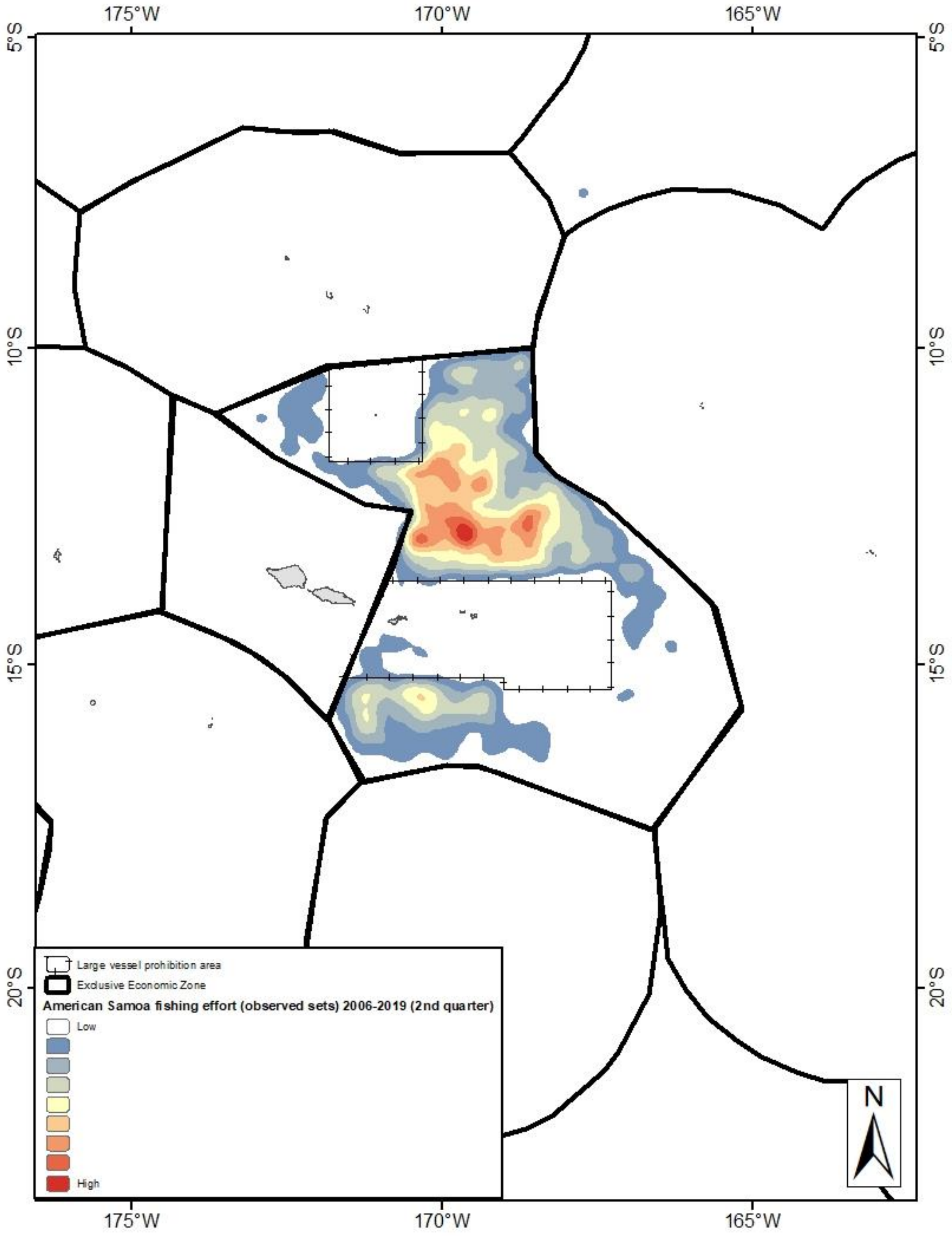


Figure 33. Geographical representation of aggregated observed fishing effort by the number of sets in the ASLL fishery between 2006 and 2019 (2^{nd} quarter).

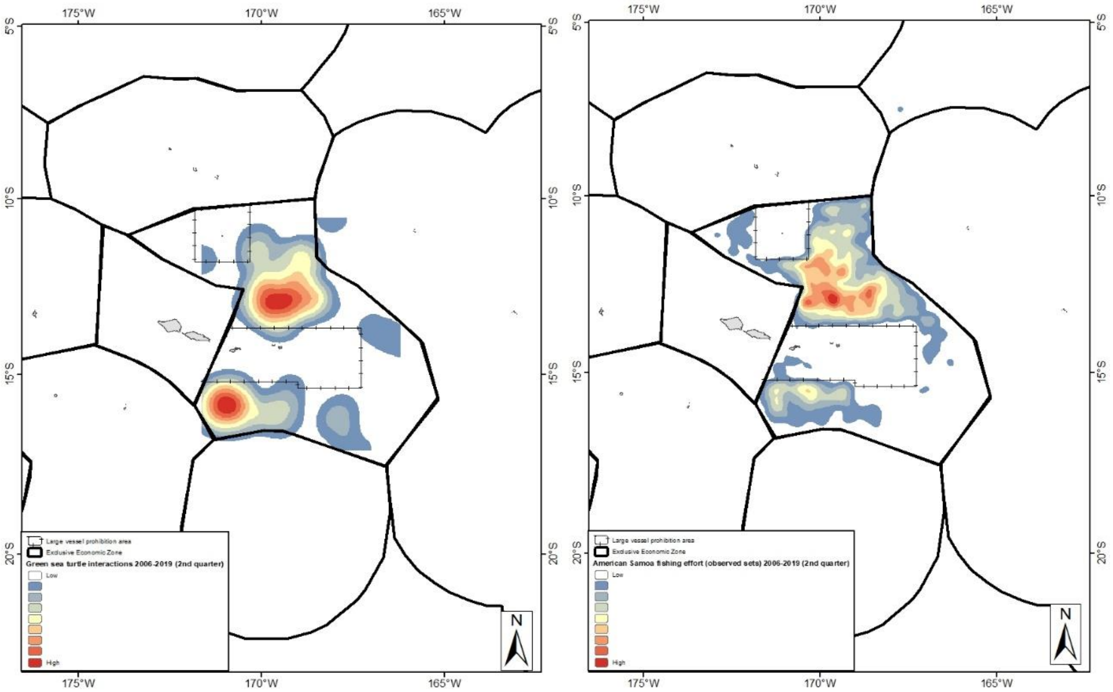


Figure 34. Side-by-side comparison of green sea turtle interactions (left) versus fishing effort (right) from 2006 to 2019 (2nd quarter).

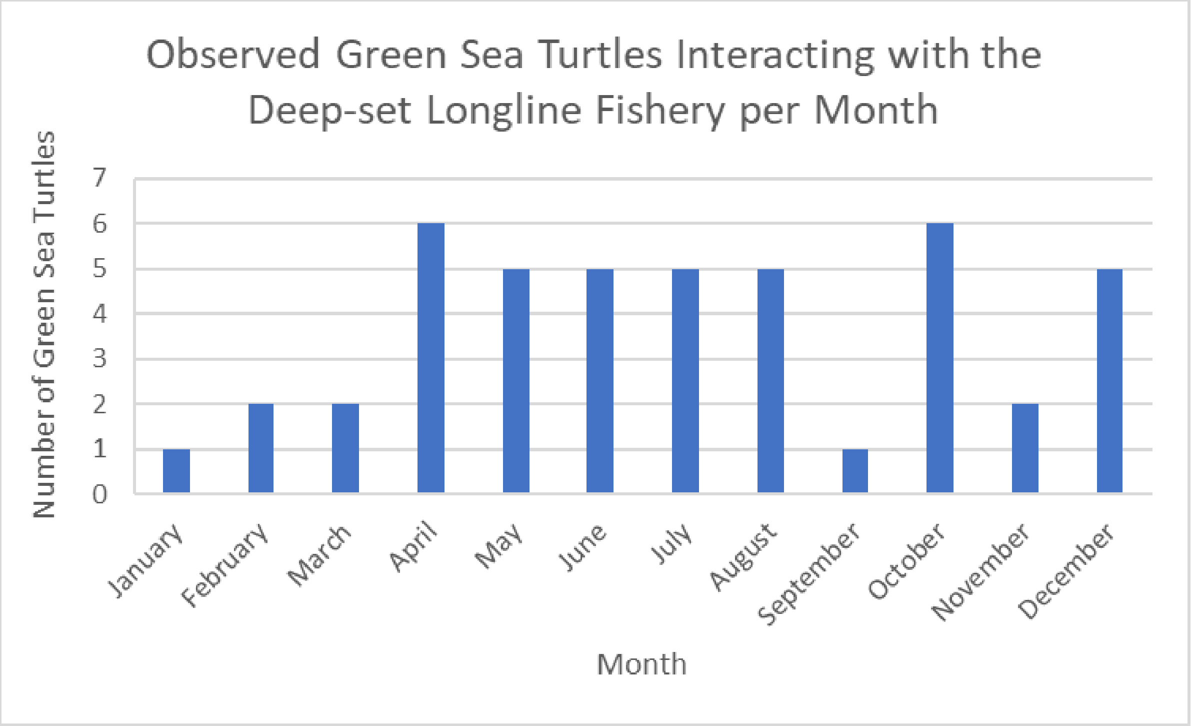


Figure 35. Numbers of observed green sea turtles captured per month in the ASLL fishery from 2006 to 2019 (n=45).

Green sea turtles are captured throughout the year, with the highest observed captures in April and October (Figure 35). Annual observed green sea turtle interactions were the highest in 2010 (8 observed) and 2011 (9 observed). Most recently, four captures were observed in each year from 2016 to 2019 with total estimated captures between 17 and 26 for each of those years based on the model of McCracken (2019a; Figure 36).

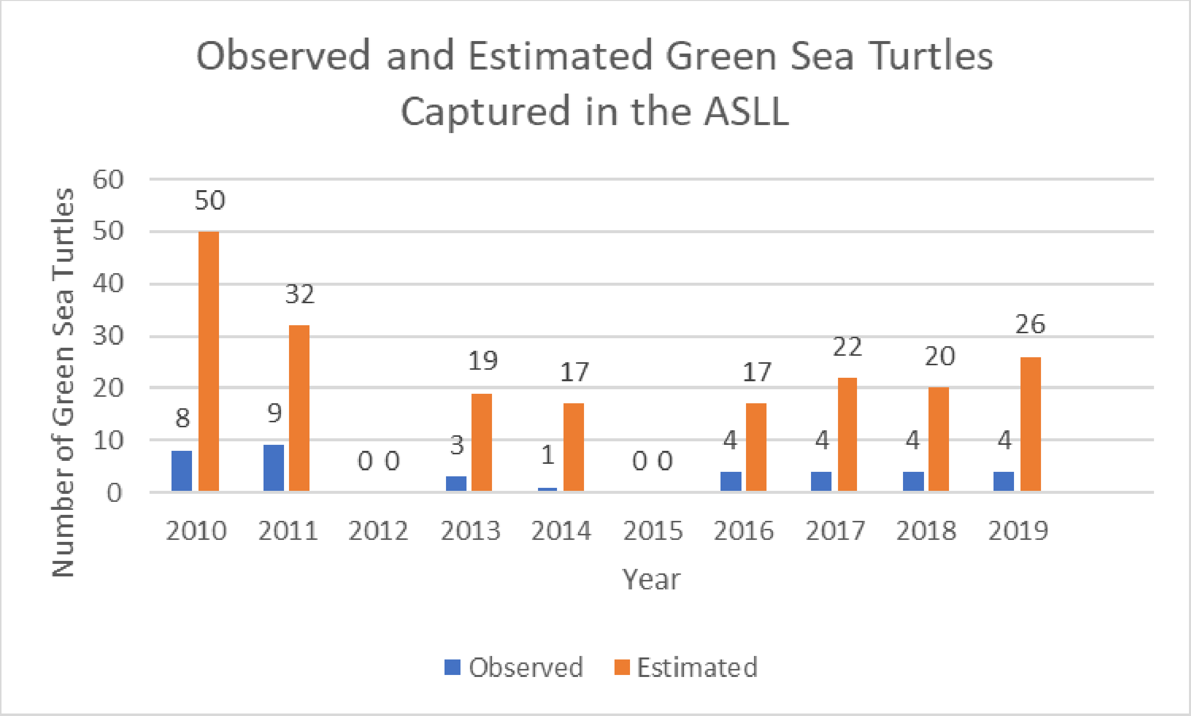


Figure 36. Green sea turtle interactions per year in the ASLL fishery between 2006 and 2019 for observed interactions (blue bars) and estimated interactions (orange bars; McCracken 2019a; 2020a, McCracken and Cooper 2022).

Predicted Future Exposure to the Fishery

Estimates of the number of green turtles that are likely to be captured in the ASLL fishery each year are presented in Table 30 (estimates are from McCracken 2019a). We interpret the data in the table as follows: on average, 14 green turtles are likely to be captured each year in the ASLL fishery; however, the 95th percentile indicates that annual captures meeting or exceeding 45 green sea turtles may be occur in approximately 5% of years. Because observer coverage for this fishery is not 100%, only a portion of these turtle captures would be reported by observers. We also report the maximum 5-yr. running average and the maximum cumulative number of captures over 5 years (Table 30). There is no 95th percentile metric associated with the running average.

Table 30. Anticipated mean, 95th percentile (from McCracken 2019a; 2012-2017), maximum 5-year running average and maximum cumulative 5-year (2012-2019) exposures of green sea turtles with fishing gear in the ASLL fishery.

Period	Mean	95th percentile
1	14	45
Maximum 5-Yr. Running Average	19.2	N/A
Maximum 5-Yr. Running Sum	96	N/A

Response Analysis

Between 2006 and 2019 there were a total of 45 interactions with green sea turtles in the ASLL fishery (see Figure 36 and Table 31), 43 of which were dead. The corresponding at vessel mortality rate for green sea turtle interactions in this fishery is 96% (95% CI: 0.86, 0.99). Since the majority of green sea turtles (43 of 45) were dead upon retrieval, we added the at-vessel mortality coefficient to derive the future mortality estimates. Using the mortality coefficients (Ryder et al. 2006), we expect the two green sea turtles externally hooked and released with no gear attached to have likely survived (5% post-interaction mortality rate based on Ryder et al. [2006]; Table 31). However, these turtles likely suffered from effects ranging from high stress immediately following post release to more severe effects such as injuries that may have impacted their feeding, migration, or even breeding if their injuries did not result in mortality.

Table 31. Cumulative number of green sea turtle mortalities by injury category for the ASLL fishery, 2006 through 2020. Numbers are based on the total number of interactions and injury mortality rates of Ryder et al. (2006). The 95% confidence interval for the fishery mortality rate was calculated using the Wilson Score Method without continuity correction (Newcombe 1998).

	Number of green sea turtle mortalities			
Injury Category	**Line < 1/2 SCL**	**Line ≥ 1/2 SCL**	**No Gear**	**Grand Total**
Entangled	-	-	-	-
External	-	-	2	2
Insertion Not Visible	-	-	-	-
Insertion Visible	-	-	-	-
Jaw	-	-	-	-
At vessel mortality	-	-	-	43

Grand Total	0	0	-	45
Fishery mortality rate (95% CI)			96% (0.86, 0.99)	

We applied the same injury category mortality coefficients to the anticipated exposure numbers for green sea turtles to derive the future mortality estimates. Table 32 provides the mortality estimates for green sea turtles for the annual mean and 95th percentile estimates. The predicted level of annual interactions is a mean of 14 (95th percentile: 45) green sea turtles (see Table 30).

We would also expect sublethal responses to the exposure of hooking and entanglement, which range from being temporary in nature such as elevated stress to more significant injuries that affect feeding and or breeding but do not result in a mortality of the animal as described earlier. If we anticipate that up to 43 green sea turtles could die in a 1-year period then we expect that the turtles that survive have sublethal responses to the interaction that range from stress responses immediately after release to problems that affect feeding, migration patterns, or even breeding if they sustain more severe injuries.

Table 32. Anticipated mortalities of green sea turtles based on predicted exposure values for the anticipated annual mean, 95th percentile (from McCracken 2019a; 2012-2017), maximum 5-year running average and maximum cumulative 5-year (2012-2019) exposures in the ASLL fishery.

Period	Mean Mortality estimate	95th Percentile mortality estimate
Annual	13	43
Maximum 5-Yr. Running Average	18.4	N/A
Maximum 5-Yr. Running Sum	92	N/A

Risk Analyses

The analyses we presented thus far allowed us to estimate the total number of green turtles we expect to be captured and killed in the ASLL fishery. Because these five green turtles are listed separately and are genetically, ecologically, and demographically independent, the analyses that follow treat these five DPSs separately.

NMFS predicts that the ASLL fishery will interact with an annual mean of 14 (95th percentile: 45) green sea turtles over a one-year time period. The maximum 5-yr. running average is 19.2 green sea turtles and we anticipate that up to 96 green sea turtles may be captured over 5 years. Some years may be higher or lower than the annual average, but the annual number of captures will rarely exceed the 95th percentile of 45 and the cumulative captures over 5-years is not expected to exceed 96 turtles. Over a 40-year time horizon, if the number of individual turtles captured remains consistent with historical numbers we would anticipate a cumulative total of 768 captures and we would anticipate that 737 of those turtles would die as a result of their interactions. As noted previously, because the genetic data do not allow us to reliably predict how many individuals will be captured from each DPS, we conduct our analysis as if all green turtle captures are from each of the affected DPSs.

East Pacific

As noted in the Status of Listed Resources section, we estimate that there were 3,580,207 East Pacific green sea turtles greater than one year old in 2015 and that the trend for this DPS is likely increasing. As described in the Exposure Analysis, all of the individuals that were measured would be considered juveniles. Assuming the total abundance for East Pacific green sea turtles is 3,580,207 individuals and assuming 14 (95th percentile: 45) individuals from this DPS may interact with the ASLL fishery, this would equal 0.00039% (95th percentile: 0.0013%) of the DPS total population. The mortality of 13 (95th percentile: 43) individual East Pacific green sea turtles assuming total abundance equals 3,580,207 represents 0.00036% (95th percentile: 0.0012%) of the total population. For the maximum 5-year running average, 19.2 captures and 18 mortalities per year represents 0.00054% of the DPS' population captured and 0.00050% of the population killed.

The maximum 5-year running sum of 96 East Pacific green sea turtles suggests that over 10 years we would anticipate 192 total captures, of these, 184 would be expected to die. This level of interaction would represent 0.0054% of the population being impacted and 0.0051% of the population being killed. As this population is increasing in size (Seminoff et al. 2015), these values will decrease at 10, 20, 30, and 40 years into the future as we anticipate higher population sizes at those time points.

Our assessment illustrates that the ASLL fishery interacts with (incidentally captures) a very small portion of the East Pacific green sea turtle. At projected abundance levels (3,580,207 individuals), incidentally capturing and killing a maximum 5-year running average of up to 18 East Pacific green sea turtles per year constitutes death of 0.00050% of the population. We consider the implications of this effect on East Pacific green sea turtles in the Integration and Synthesis section of this opinion.

Central South Pacific

As noted in the Status of Listed Resources section, there is evidence that this DPS is declining but there are no data to quantify the rate of decline (Seminoff et al. 2015). We estimate the total population size in 2015 to be 631,745 individuals greater than one year old. In the Environmental Baseline section, we present CPUE trend data for green sea turtles captured in the WCPO longline fisheries (Figure 26). Based on this analysis, the CPUE of green sea turtles is declining at a rate of 5.8% per year (95% CI: -7.6% to -3.9%; standard deviation: 0.018). In the absence of specific trend data for the Central South Pacific green sea turtle, we use this rate of decline as an estimate for the population trend. Therefore, we estimate the population size in 2022 to be 420,939 individuals greater than one year old.

As described in the Exposure Analysis, all individuals that were measured would be considered juvenile. Assuming total abundance for Central South Pacific green sea turtles in 2020 is 420,939 individuals and assuming 14 (95th percentile: 45) individuals from this DPS may interact with the ASLL fishery, this would equal 0.0033% (95th percentile: 0.0011%) of the DPS total population. The mortality of 13 (95th percentile: 43) individual green sea turtles assuming total abundance equals 420,939 represents 0.0031% (95th percentile: 0.0010%) of the total population. For the maximum 5-year running average, 19.2 captures and 18 mortalities per year represents 0.0046% of the DPS' population impacted and 0.0043% killed.

The maximum 5-year running sum of 96 green sea turtles suggests that over 10 years we would anticipate 192 total captures, of these, 184 would be expected to die. This level of interaction would represent 0.046% of the population being impacted and 0.044% of the population being killed.

As the DPS declines, the population size in 2032 would be 235,682, 131,959 in 2042, 73,884 in 2052, and 41,369 by 2062. When we consider 192 interactions over 10 years (extrapolated from the maximum 5-yr. running average of 96 per year), the mortality resulting from interaction with the fishery would be 0.078% of the total population size estimated for 2032. Between 2032 and 2042 if the number of expected interactions stay the same, the fishery would kill 0.14% of the total population size estimated in 2042, 0.25% of the Central South Pacific green sea turtle estimated in 2052, and 0.44% of the population in 2062 based on a 5.8% per year decline.

Our assessment illustrates that the ASLL fishery interacts with (incidentally captures) a very small portion of the Central South Pacific green sea turtle. At the 2022 projected abundance levels (420,939 individuals), incidentally capturing a maximum 5-year running average of 19.2 and killing an average of 18 Central South Pacific green sea turtles per year constitutes death of 0.0043% of the population. If captures remain the same as the population declines, the fishery would incidentally capture and kill 0.044% of the population annually in 40 years which we consider to be the most conservative reasonable scenario as capture rates are likely to decline if the population declines by 90% over this time period as predicted from a 5.8% per year decline. Without the mortalities caused by the ASLL, from 2022 to 2023, the population's rate of decline would be reduced to 5.795% per year decline; from 2022 to 2062, the population's rate of decline would be reduced to 5.783% per year decline. Given the estimated standard deviation for the population growth rate (0.018), and based on a power analysis for detecting differences in means with the same variance (Cohen 1988; Appendix B), more than 130,000 years of data would be required to detect a statistically significant difference in these values. We consider the implications of this effect on Central South Pacific green sea turtles the Integration and Synthesis section of this opinion.

Central West Pacific

As noted in the Status of Listed Resources section, there is evidence that this DPS is declining but no data to quantify the rate of decline (Seminoff et al. 2015). We estimate the total population size in 2015 to be 1,543,625 individuals greater than one year old. In the Environmental Baseline section, we present CPUE trend data for green sea turtles captured in the WCPO longline fisheries (Figure 26). Based on this analysis the CPUE of green sea turtles is declining at a rate of 5.8% per year (95% CI: -7.6% to -3.9%; standard deviation: 0.018). In the absence of specific trend data for the Central West Pacific green sea turtle, we use this rate of decline as an estimate for the population trend. Therefore, we estimate the population size in 2022 to be 1,028,533 individuals greater than one year old.

As described in the Exposure Analysis, all captured and measured green sea turtles were juveniles. Assuming total abundance for Central West Pacific green sea turtles is 1,028,533 individuals and assuming 14 (95th percentile: 45) individuals from this DPS may interact with the ASLL fishery, this would equal 0.0014% (95th percentile: 0.0044%) of the DPS total population captured and 0.0013% (95th percentile: 0.0042%) of the population killed. For the maximum 5-

year running average, we expect 19.2 captures and 18 mortalities per year which equals 0.0019% of the population captured and 0.0018% of the population killed.

The maximum 5-year running sum of 96 green sea turtle captures suggests that over 10 years we would anticipate 192 total captures, of these, 184 would be expected to die. This level of interaction would represent 0.019% of the population being impacted and 0.018% being killed.

As the DPS declines, the population size in 2032 would be 575,874; 322,430 in 2042; 180,529 in 2052, and 101,078 by 2062. When we consider 192 interactions and 184 mortalities over 10 years (extrapolated from the maximum 5-yr. running average of 2 per year), 0.033% of the total population size estimated for 2032 would be killed. Between 2032 and 2042, if the number of expected interactions stay the same, the fishery would kill 0.057% of the total population size estimated in 2042, 0.0083% of the Central West Pacific green sea turtle estimated in 2052, and 0.18% of the population in 2062 based on a 5.8% per year decline.

Our assessment illustrates that the ASLL fishery interacts with (incidentally captures) a very small portion of the Central West Pacific green sea turtle. At the 2022 projected abundance levels (1,028,533 individuals), incidentally capturing and killing a maximum 5-year running average of 18 Central West Pacific green sea turtles per year constitutes death of 0.0018% of the population. If captures remain the same as the population declines the fishery would incidentally capture and kill 0.018% of the population in 40 years which we consider to be the most conservative reasonable scenario as capture rates are likely to decline if the population declines by 90% as predicted from a 5.8% per year decline. Without the mortalities caused by the ASLL, from 2022 to 2023 the population's rate of decline would be reduced to 5.7982% per year decline; from 2022 to 2062 the population's rate of decline would be reduced to 5.793% per year decline. Given the estimated standard deviation for the population growth rate (0.018), and based on a power analysis for detecting differences in means with the same variance (Cohen 1988; Appendix B), almost 800,000 years of data would be required to detect a statistically significant difference in these values. We consider the implications of this effect on Central West Pacific green sea turtles in the Integration and Synthesis section of this opinion.

East Indian-West Pacific

As noted in the Status of Listed Resources section, we estimate the total population size in 2015 was 18,171,565 individuals greater than one year old and that the population is declining at a rate of 1.11% per year (95% CI: -1.6% to -0.6%; standard deviation: 0.028). Therefore, the estimated population size in 2022 is 16,813,093. As described in the Exposure Analysis, all individuals that were measured were juveniles. Assuming total abundance for East Indian-West Pacific green sea turtles is 16,813,093 individuals in 2022 and assuming 14 (95th percentile: 45) individuals from this DPS may interact with the ASLL fishery, this would equal 0.000083% (95th percentile: 0.000077%) of the DPS total population for both captures and mortalities.

The maximum 5-year running sum of 96 green sea turtles suggests that over 10 years we would anticipate 192 total captures, of these, 184 would be expected to die. This level of interaction would represent 0.0011% of the population being impacted and 0.0011% killed.

As the DPS declines, the population size in 2032 would be 15,046,688; 13,465,864 in 2042; 12,051,123 in 2052, and 10,785,016 by 2062. When we consider 184 mortalities over 10 years (extrapolated from the maximum 5-yr. running sum of 92 per year), the rate of mortality with the fishery would be 0.0012% of the total population size estimated for 2032. Between 2032 and

2042 if the number of expected interactions stay the same, the fishery would interact with 0.0014% of the total population size estimated in 2042, 0.0015% of the East Indian-West Pacific green sea turtle estimated in 2052, and 0.0017% of the population in 2062 based on a 1.11% per year decline.

Our assessment illustrates that the ASLL fishery interacts with (incidentally captures) a very small portion of the East Indian-West Pacific green sea turtle. At the 2022 projected abundance levels (16,813,093 individuals), incidentally capturing and killing a maximum 5-year running average of 18 East Indian-West Pacific green sea turtles per year constitutes death of 0.00011% of the population. If captures remain the same as the population declines, the fishery would incidentally capture and kill 0.0017% of the population in 40 years which we consider to be the most conservative reasonable scenario as capture rates are likely to decline if the population declines by 36% as predicted from a 1.11% per year decline. Without the mortalities caused by the ASLL, from 2022 to 2023 the population's rate of decline would be reduced to 1.10989% per year decline; from 2022 to 2062 the population's rate of decline would be reduced to 1.10986% per year decline. Given the estimated standard deviation for the population growth rate (0.028), and based on a power analysis for detecting differences in means with the same variance (Cohen 1988; Appendix B), more than one million years of data would be required to detect a statistically significant difference in these values. We consider the implications of this effect on East Indian-West Pacific green sea turtles in the Integration and Synthesis section of this opinion.

Southwest Pacific

As noted in the Status of Listed Resources section, we estimate that there were 14,762,154 Southwest Pacific green sea turtles greater than one year old in 2015 and that the trend for this DPS is likely increasing. As described in the Exposure Analysis, all individuals that were measured were juveniles. Assuming total abundance for Southwest Pacific green sea turtles is 14,762,154 individuals and assuming 14 (95th percentile: 45) individuals from this DPS may interact with the ASLL fishery, this would equal 0.000095% (95th percentile: 0.00030%) of the DPS total population. The mortality of 13 (95th percentile: 43) individual green sea turtles assuming total abundance equals 14,762,154 represents 0.000088% (95th percentile: 0.00029%) of the total population. For the maximum 5-year running average, 18 mortalities per year represent 0.00012% of the DPS' population.

The maximum 5-year running sum of 96 green sea turtles suggests that over 10 years we would anticipate 192 total captures, of these, 184 would be expected to die. This level of interaction would represent 0.0013% of the population being impacted and 0.00012% of the population being killed. As this population is increasing in size (Seminoff et al. 2015), these values will decrease at 10, 20, 30, and 40 years into the future as we anticipate higher population sizes at those time points.

Our assessment illustrates that the ASLL fishery interacts with (incidentally captures) a very small portion of the Southwest Pacific green sea turtle. At projected abundance levels (14,762,154 individuals), incidentally capturing and killing a maximum 5-year running average of 18 Southwest Pacific green sea turtles per year constitutes death of 0.00012% of the population. We consider the implications of this effect on Southwest Pacific green sea turtles in the Integration and Synthesis section of this opinion.

4.4.2 Leatherback sea turtle

Exposure Analysis

The first leatherback sea turtle observed capture in the ASLL was in 2011. From 2011 to 2019 there have been a total of 12 observed interactions with an estimated total captures of 74 leatherback sea turtles from 2011 to 2019 (Table 33; McCracken 2019a, 2020).

Table 33. Number of observed and estimated leatherback interactions from 2010 to 2019 based on observer coverage (McCracken 2019a, 2020).

Year	% Observer Coverage	Observed	Estimated
2010	25.0%	0	0
2011	33.0%	2	4
2012	19.8%	1	6
2013	19.4%	2	13
2014	19.4%	0	4[1]
2015	22.0%	3	22
2016	19.4%	1	3
2017	20.0%	1	3
2018	17.5%	1	6
2019	15.7%	1	7
Grand Total		12	75

[1] In her statistical analysis for estimation of captures, McCracken (2019) defines analysis periods as strata which are roughly equivalent to calendar years; however, trips that occur over two calendar years (i.e. December to January) must be assigned to a strata. Therefore, while there may not have been any observed captures within the 2014 calendar year, the strata defined as 2014 did include one or more observed captures resulting in a positive value for estimated captures.

Demographic Patterns of Exposure

Captured leatherback sea turtles larger than 4 ft., or 122 cm SCL, are generally not boarded as they are too large for the dipnet and their length is estimated by the observer looking over the rail and down at the turtle in the water. Therefore, their lengths have a high degree of uncertainty.

The smallest reported nesting adult female in the West Pacific population is 138.5 cm CCL (Lontoh 2014) or approximately 131 cm SCL based on the relationship between SCL and CCL in Tucker and Frazer (1991); 131 cm SCL converts to 4.3 ft. Given the uncertainty associated with the methods used to estimate the size of unlanded leatherback sea turtles, it is possible that a proportion of leatherback sea turtles estimated as 4-ft are adults. Of the 12 observed leatherbacks captured in the ASLL, 11 had measured or estimated carapace lengths and eight of those were juveniles with measured carapace length less than 100 cm SCL (Figure 37; 73%: 95% CI: 43% to 90%). Of the remaining 3, two were estimated as greater than 4 ft. and assumed to be adults while one was estimated at 4 ft. This individual was boarded which suggests it was not greater than 4 ft. and therefore not likely to be an adult. We consider it a subadult. Therefore, 2 of the 11 measured turtles were adult size (18%; 95% CI: 5% to 48%).

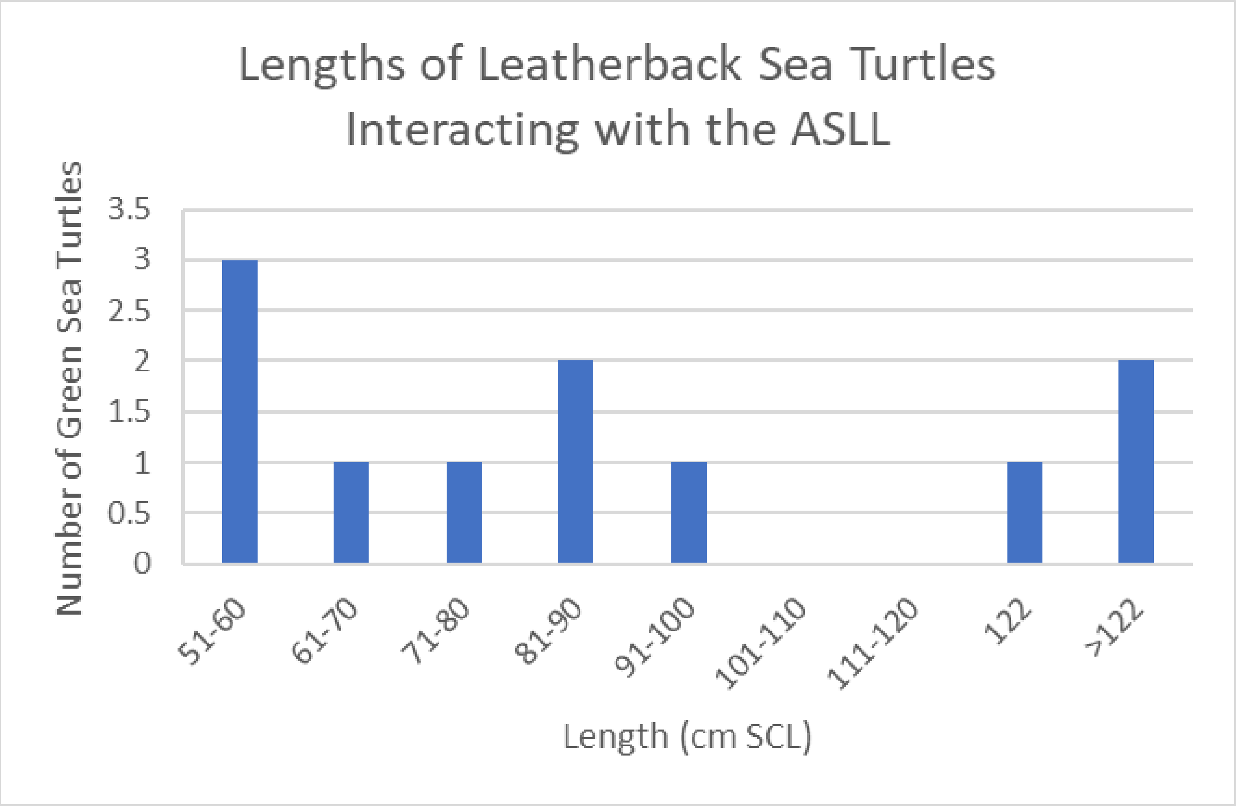


Figure 37. Lengths of observed leatherback sea turtles interacting with the ASLL fishery from 2011 to 2019.

Based on genetic samples from leatherback sea turtles captured in the ASLL fishery, of those sampled, all were confirmed to be from the western Pacific regional population comprised of Papua-Barat, Indonesia, PNG, and the Solomon Islands. (P. Dutton pers. comm. July 5, 2018). Generally, the sex of leatherback sea turtles is difficult to determine because in juveniles, sex cannot be determined by external examination, and adult-sized animals typically cannot be boarded.

Spatial and Temporal Patterns of Exposure

We plotted the 12 observed leatherback interactions that have occurred in the ASLL fishery between 2011 and 2019 in ArcGIS. Spatial remoteness between the data do not allow for viable density mapping products to be produced. Therefore, due to the spatial distance and small number of interactions, a reliable heat map could not be produced using ArcGIS. No spatial patterns are apparent from the data due to the small number of data points. However, it is interesting to note that out of the observed leatherback interactions, 11 out of 12 interactions are in the EEZ and most of these are in the northern quadrant when compared to where fishing effort is occurring.

Temporally, one third of observed leatherback sea turtle interactions occurred during the month of June followed by two interactions in both May and August (Figure 38). When compared to fishing effort in the ASLL fishery from 2011-2019 (Figure 39), June has a low level of effort (sets) when compared to other months. Annual observed and estimated leatherback sea turtle interactions peaked in 2015 with 3 observed and 22 estimated (McCracken 2019a, 2020) captures (Figure 40).

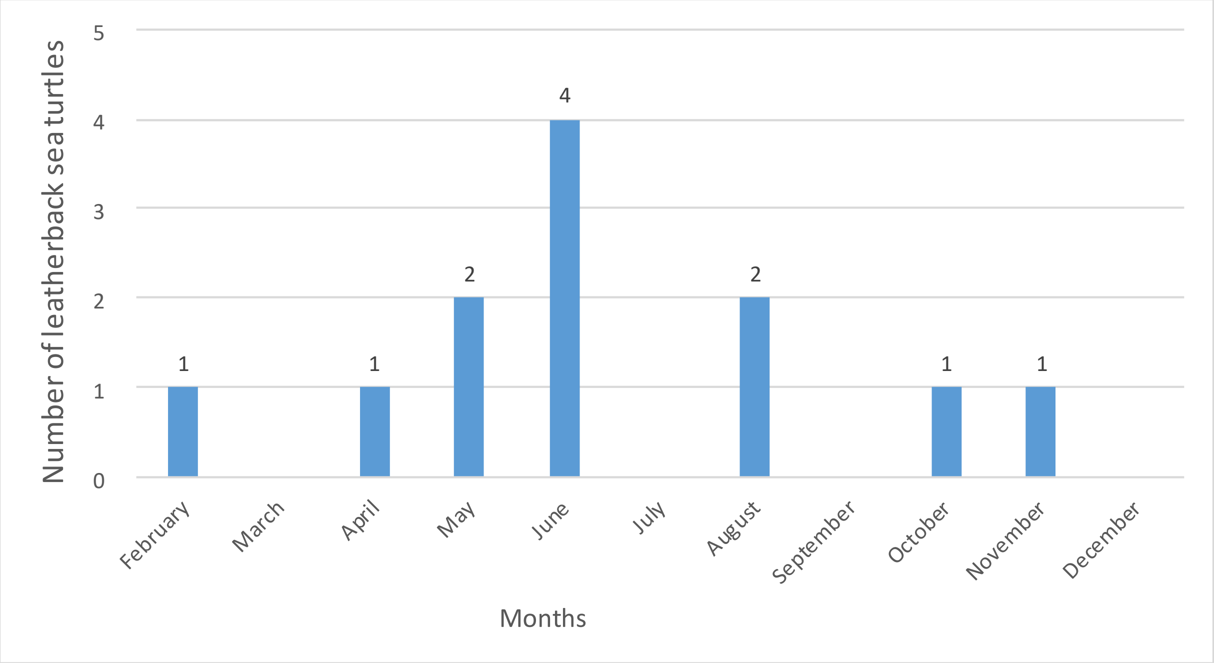


Figure 38. Number of leatherback sea turtle interactions per month in the ASLL fishery from 20011 – 2019.

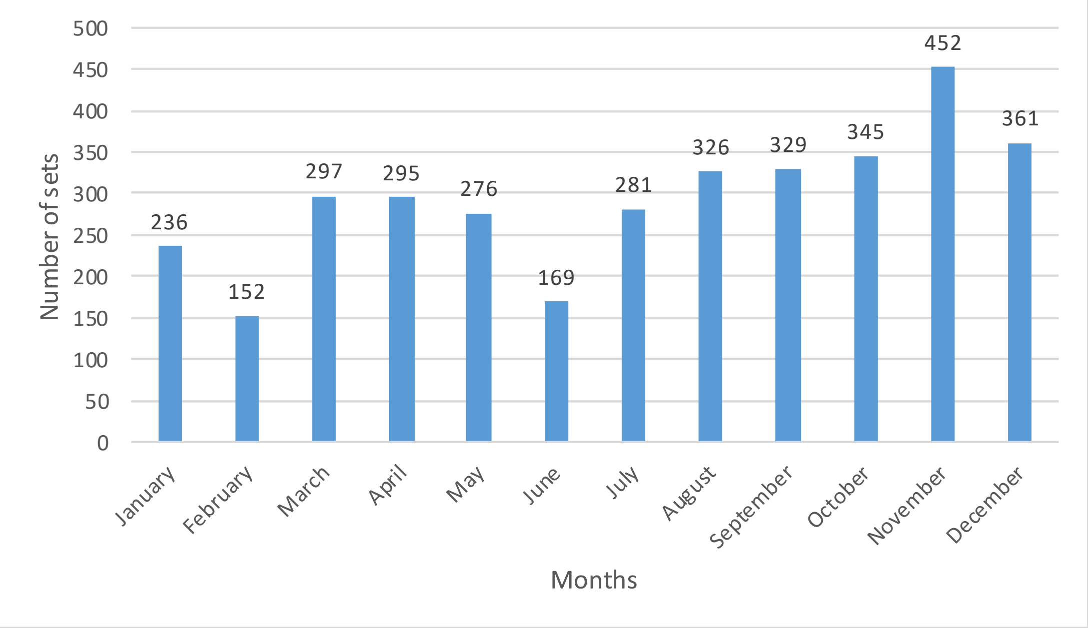


Figure 39. Fishing effort (sets) per month in the ASLL fishery from 2011 – 2019 (NMFS PIFSC logbook data).

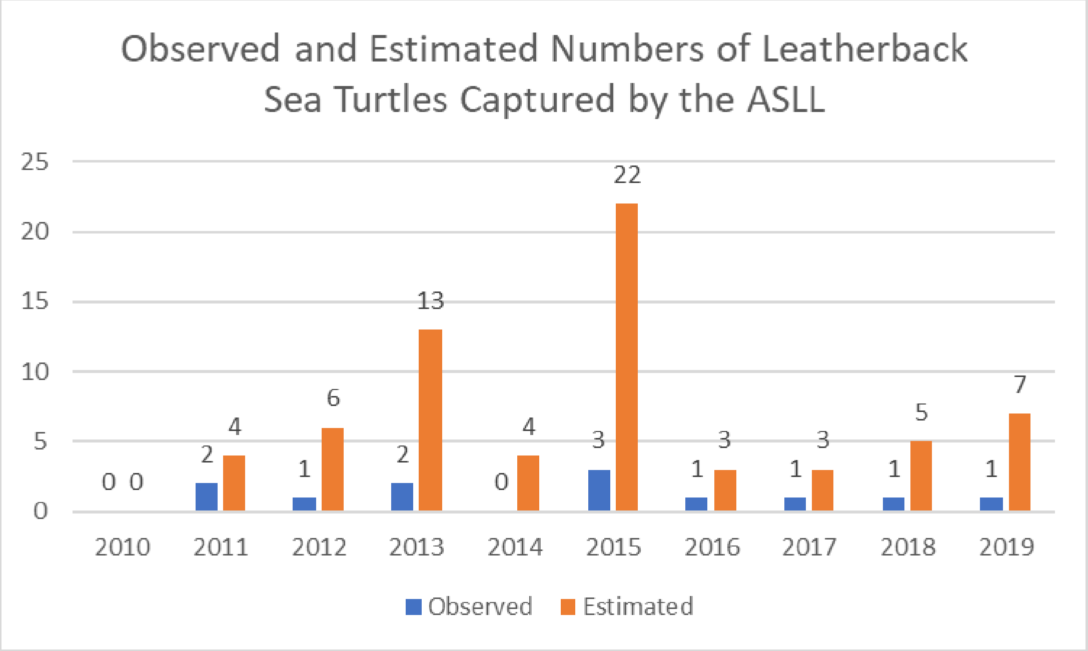


Figure 40. Observed (blue bars) and estimated (orange bars) leatherback sea turtles interacting with the ASLL each year.

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interaction levels (McCracken 2019a), as previously described in the green sea turtle section above. The predictions were based on the recorded bycatch of leatherback sea turtles in the ASLL fishery. Table 34 lists the resulting predictions of future annual (mean and 95th percentile) for leatherback sea turtles in the ASLL fishery (McCracken 2019a). On average, we would expect a mean of 10 (95th Percentile: 30) leatherback turtles to be captured each year in the ASLL fishery, although the number of leatherback turtles captured in any given year could rarely exceed 30. Only a portion of these turtle captures would be reported by observers because the ASLL fishery does not have 100% observer coverage. There is no 95th percentile metric associated with the running average.

Table 34. Anticipated mean, 95th percentile (from McCracken 2019a; 2012-2017), maximum 5-year running average and maximum cumulative 5-year (2012-2018) exposures of leatherback sea turtles with fishing gear in the ASLL fishery.

Period	Mean	95th percentile
1	10	30
Maximum 5-Yr Running Average	9.6	N/A
Maximum 5-Yr Running Sum	48	N/A

Response Analysis

If the patterns found in the observer data from 2011 to 2019 are representative, the majority of leatherback turtles would be hooked rather than entangled. Over this time period, all of the captured leatherbacks were hooked, and one of them was hooked and entangled. Seven of the 12 captures were dead at-vessel, or about 58% (95% CI = 32 to 81%) and all seven dead leatherbacks were small juveniles less than 100 cm SCL. Therefore, the small juveniles have an at-vessel mortality rate of 88% (7/8; 95% CI: 53% to 98%). None of the three larger turtles estimated at 4 ft. or greater were dead at-vessel.

We used Ryder et al. (2006) to assign post-interaction mortality rates to leatherback sea turtles that were alive when they were brought alongside the fishing vessel. Of the five leatherbacks that were alive when brought to the vessel, all were externally hooked, three of them (one subadult, one adult, and one of unknown length) were released with the hook and less than half of their carapace length of line left for a mortality coefficient of 0.15 (Table 28). One juvenile was released with no gear attached for a mortality coefficient of 0.10 and the second adult was released with the hook and an amount of line greater than half the carapace length for a mortality coefficient of 0.30 (Table 28; see Ryder et al. 2006 for an explanation of the mortality coefficients). Estimated total mortality (at-vessel and post release) for all leatherbacks captured by the ASLL is 65% (95% CI: 38% to 85%). Separating juveniles from subadults/adults, small juvenile estimated mortality is 89% (95% CI: 54% to 98%) and adult estimated mortality is 23% (95% CI: 2% to 79%) based on the two observed adult-sized turtles. As the exact classification of the one subadult is unclear, we did not include it in the calculation of small juvenile or adult mortality rates but did include it, along with the one leatherback with unknown length, in the total mortality rate calculation. We applied these mortality rates to the anticipated exposure numbers for leatherback sea turtles to derive the future mortality estimates (Table 35).

Table 35. Number of leatherback sea turtles that are likely to die as a result of being captured in the ASLL each year based on the mean effective mortality rate (65% for all size classes, 23% for adults).

	Anticipated mortalities based on the mean mortality rates			
Rate	**Mean**	**95th Percentile**	**Maximum 5-yr Running Average**	**Maximum 5-yr Running Sum**
All Age Classes				
Annual	7	20	6.2	31
Adults				
Annual	0.4	1.2	0.4	2.0

Risk Analyses

As discussed in Section 1.3.1 Jeopardy Analysis, the data do not allow us to reliably predict the proposed action's effects on West Pacific leatherbacks beyond 20 years. Therefore, we use an analytical timeframe of 20 years here. Leatherback sea turtles in the West Pacific Ocean population declined 83% from the 1980's to 2011 (Tapilatu et al. 2013), and in 2017 the population was composed of an estimated 1,443 mature adults. Based on NMFS' PVA model (Martin et al. 2020a), leatherback sea turtles in the West Pacific Ocean population are declining at about 6% per year (95% CI: -23.8% to 12.2%; standard deviation: 0.894), and the population as indicated by the index beaches is at risk of falling to less than half of its 2017 abundance by 2029 (95% CI, 2022 to 2042) if the mean 6% per year decline continues.

PVA modeled estimates suggest the index beaches of Jamursba Medi and Wermon consisted of about 790 adult female leatherback sea turtles as of 2017 (median sum of total reproductive females, LCI = 666). As noted in the Status of Listed Resources section of this biological opinion, these two nesting beaches represent approximately 75% of the total West Pacific population nesting. Assuming a 73% female sex ratio and based on NMFS' results for median nest counts (Martin et al. 2020a), the total number of adult female leatherback sea turtles in 2017 in the West Pacific Ocean population would be 1,053 (95% CI: 888-1,256)

Preliminary data from the Jamursba Medi and Wermon index beaches indicate that nest numbers were relatively stable from 2017 to 2021 (Lontoh et al. In Prep, see Figure 16) but, as discussed in Section 1.2.2, the data are not yet available in sufficient detail to update the model of Martin et al. (2020a). Hence we acknowledge that there is a great deal of uncertainty associated with the current status of West Pacific leatherback sea turtles, as represented by the two index beaches.

As described in Section 1.5.2, Martin et al. (2020b) present essentially the same PVA as described in Martin et al. (2020a); however, they consider the impact of the ASLL fishery on the West Pacific leatherback sea turtle in Martin et al. (2020b). For both modeling efforts, they ‘removed’ the historic impact of the fishery by adding back to the population observed and estimated leatherback sea turtle mortalities caused by the fishery. An ANE was assigned to each juvenile-sized sea turtle, and the lost adult nester equivalent was added back into the year when it would have first nested and all subsequent years when it would have nested based on remigration intervals and adult survival rates (for this historic impact of the fishery only).

Martin et al. (2020b) then conducted the PVA with and without the fishery by projecting numbers of nesting females into the future, removing adult nester equivalents each year based on probabilistic draws for total annual captures and size and fishery mortality for each modeled capture. The results of this analysis found that the mean time for the population to reach 50% of 2017 abundance was 12.7 years without the fishery and 12.6 years with the fishery, these values are based on 2017, and therefore translate to about 2030 (Table 37). Siders et al. (2023) updated the modeling for Martin et al. (2020b) with a focus on the impacts of the Hawaii DSLL fishery. They updated the population size to 2021 based on the mean decline of -6% (95% CI: -24.1 to 12.2%) per year of Martin et al. (2020a). Siders et al. (2023) found that the mean time for the population to reach 50% of 2017 abundances from the projected population size in 2021 was 7.85 without the impact of the DSLL, or by about 2029.

Similarly, Martin et al. (2020b) found a mean probability of the population reaching 50% of 2017 in 10 years (or by 2027) was 39% with and without the impact of the ASLL (Table 36). Siders et al. (2023) estimate that the probability of the population reaching 50% of 2017 abundances with the population projected to 2021 within 5 years (or by 2026) to be 40% without the impact of the Hawaii DSLL, which compares well with the previously mentioned probability of 39% for the population to reach 50% of 2017 population sizes in 10 years (or by 2027) with and without the fishery (Martin et al. 2020b).

The probabilities of reaching thresholds and the time to reach thresholds in Martin et al. (2020b) and Siders et al. (2023) include the conservative assumptions that all fishery mortalities impact only the Jamursba Medi and Wermon index beaches, and that fishery captures will remain the same as the population declines. The second assumption implies that the proportion of the population impacted by the fishery increases over time as the population declines. Therefore, we can consider these results to be the most conservative reasonable scenarios because: 1) leatherbacks interacting with the fishery are not likely to all be from the Jamursba Medi and Wermon index beaches, 2) between 2017 and 2021, the population did not decline at the rate of 6% per year (Lontoh et al. In Prep; Figure 16) and 3) interactions are likely to decline as the population declines. This means that the proportion of the total population impacted by the fishery would be less and we can expect the magnitude of the impact to be less.

Table 36. Probability of reaching population thresholds of 50% and 25% of 2017 nester abundances with and without the ASLL (Martin et al. 2020b). Note that Martin et al. (2020b) include values for a 12.5% threshold but those results are beyond our analytical timeframe of 20 years.

			New Method for Maturity		
Threshold	**Metric**	**Scenario**	**5 yr**	**10 yr**	**25 yr**
50%	**Median**	**No Take**	0.03	0.39	0.97
		Take	0.03	0.39	0.97
		Δ(NT-T)	0	0	0
	Lower 95% CI	**No Take**	0.02	0.38	0.97
		Take	0.03	0.39	0.97
		Δ(NT-T)	-0.01	-0.01	0
	Upper 95% CI	**No Take**	0.03	0.39	0.97
		Take	0.03	0.40	0.97
		Δ(NT-T)	0	0	0
25%	**Median**	**No Take**	0	0	0.61
		Take	0	0	0.62
		Δ(NT-T)	0	0	-0.01
	Lower 95% CI	**No Take**	0	0	0.60
		Take	0	0	0.62
		Δ(NT-T)	0	0	-0.02
	Upper 95% CI	**No Take**	0	0	0.61
		Take	0	0	0.63
		Δ(NT-T)	0	0	-0.02

Table 37. Time to reach population thresholds with and without the ASLL fishery (Martin et al. 2020b). Times are from 2017. Note that Martin et al. (2020b) include values for a 12.5% threshold but those results are beyond our analytical timeframe of 20 years.

Threshold	Metric	Scenario	Mean yr	Median yr	L95% yr	U95% yr
50%	**Median**	**No Take**	12.7	12	5	26
		Take	12.6	12	5	25
		Δ(NT-T)	0.1	0	0	1
25%	**Median**	**No Take**	24.1	23	13	41
		Take	23.8	23	13	40
		Δ(NT-T)	0.3	0	0	0

As described previously, NMFS predicts that the ASLL will interact with a mean of 10 (95th percentile: 30) West Pacific leatherback sea turtles per year. Of these, we anticipate that a mean of 7 (95th percentile: 20) will die each year from their interaction based on the mean interaction mortality rate of 65%. The maximum 5-year running average is 9.6. We estimate that an average of 0.4 adults may be killed by the fishery in a year (note: these are adults, not juveniles extrapolated to adult equivalents as in Martin et al [2020b]). Applying the sex ratio of 0.73, these adults equate to 0.3 adult females or about one adult female mortality (not ANE) every 3.3 years.

We looked at the percent of the population that would be captured or killed by the fishery over the next 20 years assuming average annual numbers of captures in the ASLL remain stable (Table 38). Given the preliminary data of Lontoh et al. (In Prep), the adult population size appeared stable from 2017 to 2021 and we assume that the population size in 2022 was the same as 2017. Starting in 2022, we consider the possibility of the population declining by 6% per year over the next 20 years and the impact of the ASLL if captures remain constant by considering the proportion of the population captured and killed by the fishery. We looked at these proportions for the estimated mean number of adults and the mean of the estimated total population size. For adults, the percent of the 2017 and 2022 population abundances killed by the fishery in any one year represents 0.03% of West Pacific adults and 0.01% of the total population size (Table 38).

By 2042, if the population declines by 6% per year from 2022 while fishery captures remain the same, the fishery would kill 0.09% of the adult population abundance and 0.02% of total population abundance per year (Table 38). We consider the 2032 and 2042 values the most conservative reasonable scenarios as 1) fishery interactions are not likely to stay the same if the population declines by 70% from 2022 to 2042 as predicted by a mean decline of 6% per year, and 2) based on preliminary nesting beach data from 2018 to 2021, the population does not appear to be declining and the population sizes at 2032 and 2042 may be pessimistic. We also considered the scenario where the ASLL mortalities were added back to the population each year and estimated the resulting population growth rates without the impact of the ASLL for the time periods 2022-2023, 2022 to 2032 and 2022-2042 (Table 38).

We explore the impact of the fishery mortalities on time to reach population thresholds using the difference in population growth rates presented in Table 38 and the exponential population growth equation:

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

Where N_0 is the starting population size, N_t is the population size after time duration t, and r is the population growth rate. This equation can be rewritten as:

$$N_t/N_0 = \exp(rt) \text{ or } \ln(N_t/N_0) = rt; \text{ which can be solved for time as:}$$

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

Hence, the time for the West Pacific leatherback adult population to reach 50% of 2022 abundances with the fishery is ln(0.50)/-0.06 or 11.55 years. Removing the impact of the fishery this value becomes ln(0.50)/-0.0597 or 11.61 years for a difference of 0.06 years (Table 39). These values would be reached by 2028 to 2029. This is very comparable to the results of Martin et al. (2020b) who predict median times of 12.7 without the ASLL fishery and 12.6 years with the ASLL fishery from 2017, or by 2029 to 2030 with a similar difference of 0.1 years (Table 37). We conducted these calculations for the mean impact of the fishery from 2022 to 2042 specific in Table 38. Assuming current (2022) population sizes are similar to 2017 then we expect a mean difference of times for the adult population to reach 50%, 25% and 12.5% of 2022 abundances of less than 0.17 yr (or about 2 months) with and without the fishery. Under the most conservative reasonable scenario of the population continuing to decline while fishery captures remain the same, the difference with and without the fishery is a maximum of 0.16 years (about 2 months) for the population to reach 12.5% of 2022 abundances (Table 39).

The impact of these losses on the rate of decline in this population are much smaller than our confidence in the underlying data. As noted in the Status of Listed Resources section, the 95% confidence interval for the population growth rate estimated by Martin et al. (2020a) is -23.8% to 12.2% and the standard deviation is 0.0896. The greatest change in growth rate we computed is an annual reduction of 0.09% of adults in year 20, under the conservative assumption that interactions stay the same despite the population declining by 70%. If the growth rate has improved from an annual mean of -6% as recent evidence suggests, population numbers will be higher and the proposed action's effect on the growth rate will be lower than we calculated.

Even at the mean rate of decline of 6% per year and the most conservative reasonable scenario of fishery captures remaining the same as the population declines, the difference between the scenarios with and without the fishery is an increase in the population trend to -5.95%. Based on a power analysis and assuming the same standard deviation of 0.0896 (Cohen 1988; Appendix B), over 300,000 years of data would be required to detect a significantly significant difference in these values. This difference cannot be statistically detected when the 95% CI around the estimate of a value ranges 36%. Even if the most conservative reasonable scenario happens, it would not be statistically detectable. Therefore, we are reasonably certain the consequences of the proposed action will not be appreciable biologically to the numbers, reproduction, or distribution of the West Pacific leatherback sea turtle population.

We acknowledge the uncertainty in the status of this population and the reliance of the status on continued nesting beach monitoring and protection programs. If necessary, we will reassess our analysis of the impacts when updated nesting beach data are available. We consider the implications of the effect of the fishery on leatherback sea turtles in the Integration and Synthesis section of this opinion.

Table 38. Impact of ASLL incidental captures and mortalities on West Pacific leatherback sea turtles assuming captures remain constant over time. Capture numbers are based on the 5-year running average and sum (Table 34). Population growth rates with ASLL mortalities calculated as baseline population growth rate (-6% per year) minus the percent of abundance killed by the fishery each year.

Population Segment and Statistic	Abundance	Captures	Mortalities	% of Abundance Captured	% of Abundance Killed	Population Growth Rate With ASLL Mortalities (% per year; Martin et al. 2020b)	Population Growth Rate with ASLL Mortalities Removed (% per year)
2017							
Mean Adults	1442	1.7	0.4	0.12	0.03	-6.0	-5.97
Mean Total Population	99,921	9.6	6.2	0.01	0.01	-6.0	-5.99
2022 (assuming stable population since 2017)							

Population Segment and Statistic	Abundance	Captures	Mortalities	% of Abundance Captured	% of Abundance Killed	Population Growth Rate With ASLL Mortalities (% per year; Martin et al. 2020b)	Population Growth Rate with ASLL Mortalities Removed (% per year)
Mean Adults	1442	1.7	0.4	0.12	0.03	-6.0	-5.97
Mean Total Population	99,921	9.6	6.2	0.01	0.01	-6.0	-5.99
2032 (assuming 6% per year decline from 2022)							
Mean Adults	792	1.7	0.4	0.22	0.05	-6.0	-5.96
Mean Total Population	54,838	9.6	6.2	0.02	0.01	-6.0	-5.99

Population Segment and Statistic	Abundance	Captures	Mortalities	% of Abundance Captured	% of Abundance Killed	Population Growth Rate With ASLL Mortalities (% per year; Martin et al. 2020b)	Population Growth Rate with ASLL Mortalities Removed (% per year)
2042 (assuming 6% per year decline from 2022)							
Mean Adults	434	1.7	0.4	0.40	0.09	-6.0	-5.95
Mean Total Population	30,096	9.6	6.2	0.03	0.02	-6.0	-5.99

Table 39. Mean times for the adult portion of the West Pacific leatherback to reach 2017 population thresholds with and without the ASLL fishery based on population growth rates in Table 38. Zeros indicate that the threshold will have been reached by that year, based on mean time to threshold.

Threshold	**50% of 2017 Population Size = 721**	**25% of 2017 Population Size = 361**	**12.5% of 2017 Population Size = 180**
Time from 2017 to threshold with ASLL (yr.)	11.55	23.10	34.66
2017 Time to threshold without ASLL (yr.)	11.61	23.22	34.83
2017 Difference (yr.)	0.06	0.11	0.17
Time from 2022 to threshold with ASLL (yr.)	11.55	23.10	34.66
Time from 2022 to threshold without ASLL (yr.)	11.61	23.22	34.83
2022 Difference (yr.)	0.06	0.11	0.17
Time from 2032 to threshold with ASLL (yr.)	1.55	13.10	24.66
Time from 2032 to threshold without ASLL (yr.)	1.56	13.19	24.82
2032 Difference (yr.)	0.01	0.09	0.16
Time from 2042 to threshold without ASLL (yr.)	0	3.10	14.66

Threshold	50% of 2017 Population Size = 721	25% of 2017 Population Size = 361	12.5% of 2017 Population Size = 180
Time from 2042 to threshold with ASLL (yr.)	0	3.13	14.79
2042 Difference (yr.)	0	0.03	0.13

4.4.3 Olive Ridley Sea Turtle

Exposure Analysis

The first olive ridley was observed being captured by the ASLL in 2011. From 2011 to 2019 there have been a total of 16 observed interactions with an estimated total captures of 80 olive ridley sea turtles from 2011 to 2019 (Table 40; McCracken 2019a, 2020).

If we apply the results of genetic analyses conducted by NMFS, about 6 of the observed captures would have represented olive ridley sea turtles from both the endangered population and the threatened East Pacific population (or 32 when adjusted for observer coverage) and 10 would have been from the West Pacific population (or 48 when adjusted for observer coverage).

Table 40. Number of observed and estimated olive ridley interactions from 2010 to 2019 based on observer coverage (McCracken 2019a, 2020).

Year	% Observer Coverage	Observed	Estimated
2010	25.0%	0	-
2011	33.0%	1	4
2012	19.8%	1	6
2013	19.4%	1	4
2014	19.4%	2	5
2015	22.0%	1	6
2016	19.4%	3	12

Year	% Observer Coverage	Observed	Estimated
2017	20.0%	2	12
2018	17.5%	2	11
2019	15.7%	3	20
Grand Total		16	80

Demographic Patterns of Exposure

Based on mitochondrial and nuclear DNA analyses of tissue samples collected from 5 olive ridley turtles captured in the ASLL fishery from 2012 to 2017, 40% of these turtles appear to be from the eastern Pacific population that occurs in Mexico, Costa Rica, and other areas of Central America (Dutton pers. comm. June 29, 2018). The remaining 60% are part of the Western-Indo Pacific (Dutton pers. comm. June 29, 2018).

Sizes of olive ridley sea turtles captured by the ASLL fishery ranged from 37 to 63 cm SCL (Figure 41). The minimum size of nesting olive ridley sea turtles is 53 cm SCL (Zug et al. 2006), therefore approximately 60% of olive ridley sea turtles interacting with the fishery that were measured may be adults.

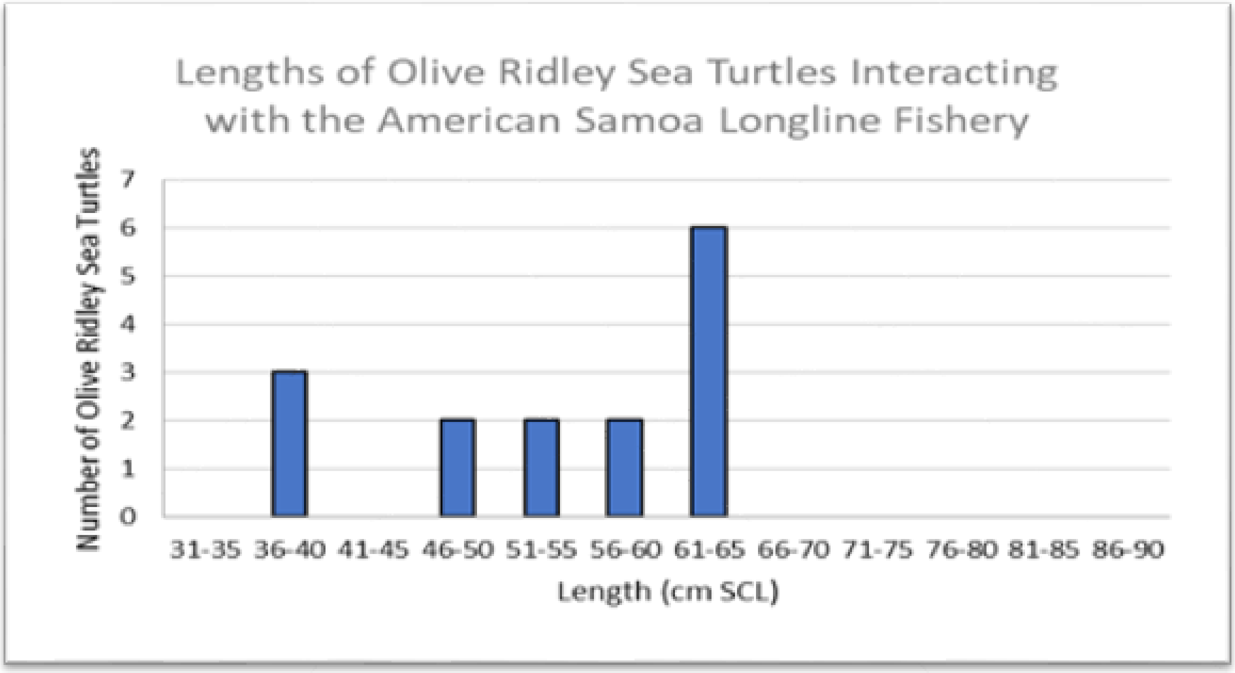


Figure 41. Lengths of observed olive ridley sea turtles interacting with the ASLL fishery from 2011 to 2019.

Spatial and Temporal Patterns of Exposure

We plotted the 16 observed olive ridley interactions that have occurred in the ASLL fishery between 2011 and 2019 (2nd quarter) in ArcGIS (note, only data up to the second quarter of 2019 were available at the time of this analysis). Spatial remoteness between the data do not allow for viable density mapping products to be produced. Therefore due to the spatial distance and small number of interactions, a reliable heat map could not be produced using ArcGIS. No spatial patterns are apparent from the data due to the small number of data points. However, it should be noted that out of the observed olive ridley interactions, all but one interaction is in the EEZ. Temporally, over 60% of observed olive ridley sea turtle interactions occurred in August and October (Figure 42). This time frame coincides with moderately high observed fishing effort in the ASLL fishery from 2011-2019 (2nd Quarter).

Most olive ridley captures occur from August to December (81% of observed captures; Figure 42). Annual observed and estimated olive ridley sea turtle interactions peaked in 2019 with 3 observed and 20 estimated (McCracken 2019a) captures (Figure 43).

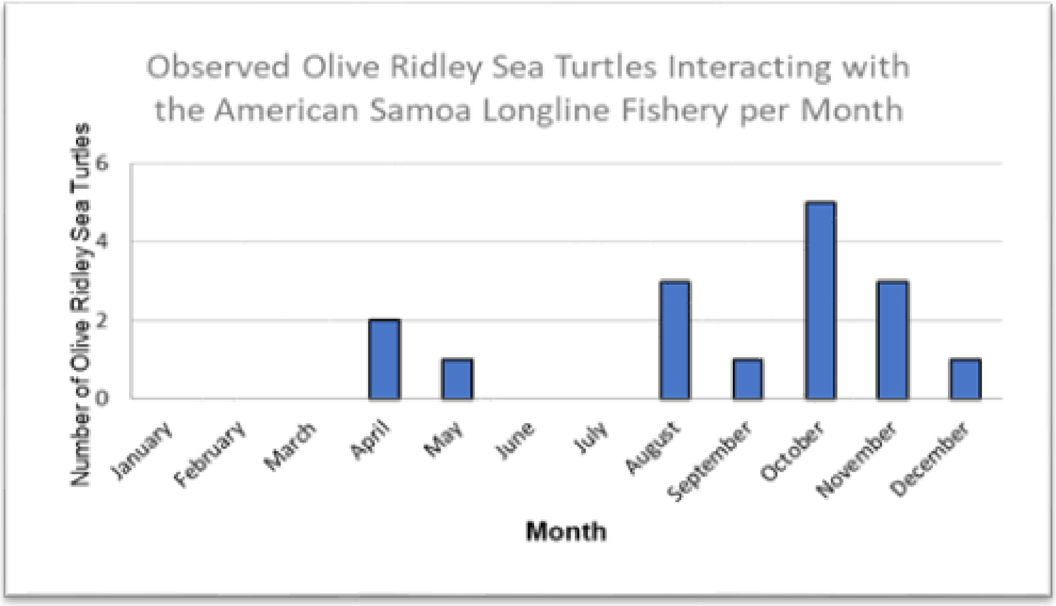


Figure 42. Number of olive ridley turtle interactions per month in the ASLL fishery from 2011 to 2019.

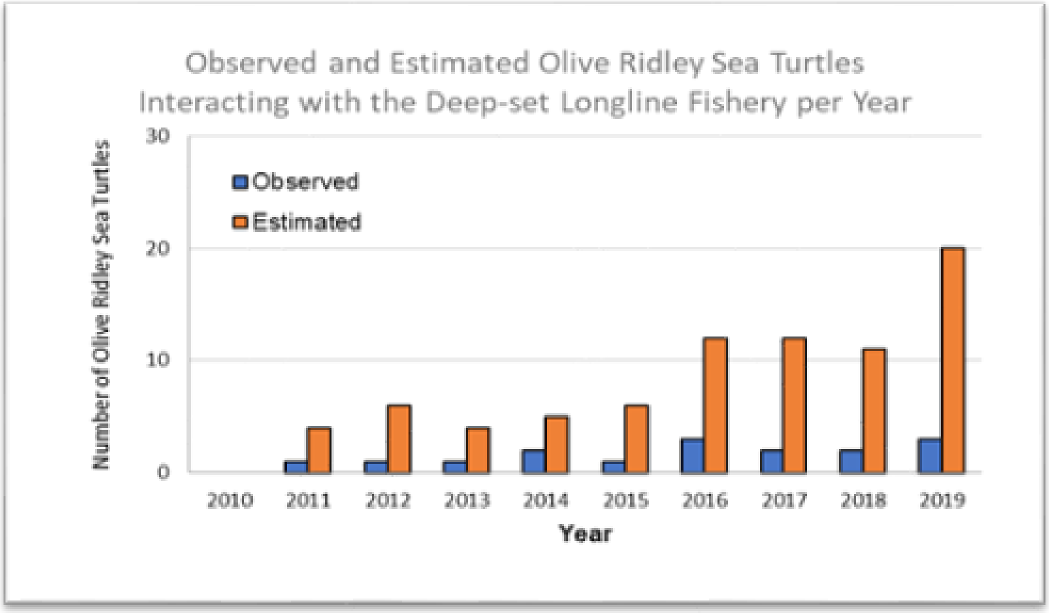


Figure 43. Observed (blue bars) and estimated (orange bars) olive ridley sea turtles interacting with the ASLL each year.

Predicted Future Exposure to the Fishery

Estimates of the number of olive ridley turtles that are likely to be captured in the ASLL fishery each year are presented in Figure 43 (estimates are from McCracken 2019a). On average, we would expect 11 (95th percentile: 27) olive ridley turtles to be captured each year in the ASLL fishery. As with the turtles discussed previously, only a portion of these turtle captures would be reported by observers because the ASLL fishery has only 20% observer coverage (though this number dipped to 2.13% in 2020 due to restrictions associated with the COVID 19 pandemic). We also report the maximum 5-yr. running average and the cumulative number of captures over 5 years (Table 41).

Based on the genetic data, we would expect 4 of the 11 olive ridley turtles that might be captured in the DSLL fishery in an average year to represent East Pacific populations, both the endangered and threatened populations. The remaining 7 would be from the West Pacific nesting population. At the 95th percentile of 27, we would expect 11 of those turtles to be from the East Pacific populations and 16 from the West Pacific population.

Table 41. Anticipated mean, 95th percentile (from McCracken 2019a; 2012-2017), maximum 5-year running average and maximum cumulative 5-year running sum (2012-2019) of olive ridley sea turtle captures in the ASLL fishery.

Period	**Mean**	**95th percentile**
1	11	27
Maximum 5-Yr. Running Average	12.2	N/A
Maximum 5-Yr. Running Sum	61	N/A

Response Analysis

Between 2006 and 2019 there were a total of 16 interactions with olive ridley sea turtles in the ASLL fishery and of these, 11 were dead when brought to the vessel. The corresponding at-vessel mortality rate for olive ridley sea turtle interactions in this fishery is 68.8% (95% CI: 44.4%, 85.8%). We used the post-interaction mortality coefficients of Ryder et al. (2006) to estimate the mortality rates of the 5 turtles that were released alive (Table 42) and we found an average post-interaction mortality rate of 12% for olive ridleys released alive in the ASLL. Combining at-vessel and post-interaction mortality rates, the total mortality rate for olive ridleys interacting with this fishery is 72.5% (95% CI: 48.0% to 88.3%).

Table 42. Number of olive ridley sea turtles interacting with the ASLL fishery between 2010 and 2019 by their injury and release condition. Numbers in parentheses are the corresponding post-interaction mortality rates from Ryder et al. (2006).

	Number of olive ridley sea turtles (Mortality Rate)			
Injury Category	**Line < 1/2 SCL**	**Line ≥ 1/2 SCL**	**No Gear**	**Grand Total**
Entangled	-	-	-	-
External	-	-	1 (0.05)	1 (0.05)
Insertion Not Visible	-	-	-	-
Insertion Visible	-	-	1 (0.25)	1 (0.25)
Jaw	-	-	3 (0.10)	3 (0.10)
Grand Total – Live Releases	-	-	5	5 (0.12)
Dead At Vessel				11 (1.0)
Grand Total		-		16 (0.725)
Fishery mortality rate (95% CI)			0.725 (0.480, 0.883)	

We applied the same injury category mortality coefficients to the anticipated exposure numbers for olive ridley sea turtles to derive the future mortality estimates. The predicted level of interactions with olive ridley sea turtles is a mean of 11 (95th percentile: 27) for one year (see Table 43) and of these we anticipate 8 (95th percentile: 20) to die. Of the 61 interactions expected to occur in the next 5-year period, 44 of these turtles would be expected to die (Table 43).

Table 43. Anticipated mortalities of olive ridley sea turtles based on predicted exposure values for the anticipated annual mean, 95th percentile (from McCracken 2019a; 2012-2017), maximum 5-year running average and maximum cumulative 5-year (2012-2018) exposures of olive ridley sea turtles with fishing gear.

Period	Mortality estimate for the mean	Upper mortality estimate
Annual	8	20
Maximum 5-Yr. Running Average	8.8	N/A
Maximum 5-Yr. Running Sum	44	N/A

Risk Analysis

About 3 of the 8 olive ridley turtles that are likely to die in an average year would originate from either the endangered or the threatened East Pacific population with 5 originating from the West Pacific population. At the 95th percentile of 27 captures, about 8 of the 20 olive ridley turtles that are likely to die would originate from either the endangered or the threatened East Pacific population with another 12 originating from the West Pacific population. For the maximum 5-year running sum, out of the 44 olive ridley sea turtles expected to die, about 18 would originate from either the endangered or the threatened East Pacific population, with 26 originating from the West Pacific population. Over a 40 year time horizon, if captures remain consistent with historical numbers we would anticipate a cumulative total of 488 captures with 195 captures and 142 mortalities for olive ridley sea turtles originating from the East Pacific populations (both endangered and threatened) and 293 captures with 212 mortalities for the West Pacific population.

East Pacific Populations (Threatened and Endangered)

As noted in the Status of Listed Resources section, the endangered Mexico population of olive ridley sea turtles is comprised of more than 1,000,000 nesting females (Abreu-Grobois and Plotkin 2008; Valverde et al. 2012; NMFS and FWS 2014). The portion of the threatened population that nests in the East Pacific similarly numbers at more than 1,000,000 nesting females on beaches from Costa Rica and Nicaragua (Chaves et al. 2005 as cited in NMFS and FWS 2014; Valverde et al. 2012; NMFS and FWS 2014).

Assuming total abundance of olive ridley sea turtles for the endangered population is a minimum of one million individuals, and we apply all captures from the East Pacific to the endangered population, annual mortalities of 3 (95th percentile: 8) olive ridleys would correspond to 0.0003% to 0.0008% of the total population. Based on the maximum 5-yr. running average, we anticipate an average of 4.9 East Pacific olive ridley sea turtles to be captured in a year with 3.5 mortalities. These numbers would represent 0.00049% of the population captured and 0.00036% of the population killed. When we consider 49 interactions and 36 mortalities over the next 10 years, the rate of interaction with the fishery would be 0.0049% and 0.0036% of the total population for the endangered Mexico populations. As this population is at least stable (NMFS and FWS 2014), these percentages of the population being impacted will not increase over time. Therefore, captures between 30 and 40 years into the future will impact no more than 0.0049% (captures) and 0.0036% (mortalities) of the total population over 10 years, or the mortality of 0.0004% annually based on the 5-yr. maximum running average.

Similarly, the threatened portion of the East Pacific populations is considered to number more than 1 million individuals with a population trend that is at least stable. Therefore, the numbers we provided above for the endangered portion of the population apply to the threatened portion of the East Pacific population. We anticipate that captures between 30 and 40 years into the future will impact no more than 0.0049% (captures) and 0.0036% (mortalities) of the total population, or the mortality of 0.0004% annually based on the 5-yr. maximum running average.

Our assessment illustrates that the ASLL fishery interacts with (incidentally captures) a very small portion of East Pacific olive ridley sea turtles from either the threatened or endangered populations. At projected abundance levels (1,000,000 individuals for both the threatened and endangered East Pacific populations), incidentally capturing (killing) an average of 4.9 (4) East Pacific olive ridley sea turtle from either population per year constitutes death of 0.00040% of the population. We consider the implications of this effect on East Pacific olive ridley sea turtles in the Integration and Synthesis section of this opinion.

West Pacific Population

As noted in the Status of Listed Resources section, in the Western Pacific, the largest remaining breeding aggregations are in Northern Australia where it's estimated that at least 5,000 females nest annually (Commonwealth of Australia 2017). Considering remigration intervals, we estimate this population at 10,000 adult females, assuming a 50% sex ratio, this would equate to 20,000 adults. When added to a well-studied nesting aggregation in Alas Purwo National Park, East Java, Indonesia, which is currently increasing; suggests a minimum adult abundance of 24,400 for West Pacific olive ridley sea turtle populations. We estimate the West Pacific population to have a minimum of 316,883 juveniles and adults based on an adult population size of 24,400 with 7.7% of the population occurring in that stage (Maulany et al. 2017).

Based on the maximum 5-yr. running average, we anticipate an average of 7.3 West Pacific olive ridley sea turtles to be captured in a year with 5 mortalities. These numbers would represent 0.0022% of the abundance captured and 0.0016% killed. Based on the maximum 5-yr. running sum, we anticipate 73 captures over 10 years, with 53 mortalities. These numbers would represent 0.023% of the abundance captured and 0.017% killed over 10 years. As West Pacific abundance is at least stable, we expect these percentages to not increase over our 40-year analytical timeframe. We consider the implications of this effect on West Pacific olive ridley sea turtles in the Integration and Synthesis section of this opinion.

4.4.4 Hawksbill sea turtle

Exposure Analysis

Since observers started monitoring this fishery in 2006, there were no observed interactions with hawksbill sea turtles until 2016. Since then, there have been a total of 3 observed interactions with an estimated total captures of 12 hawksbill sea turtles from 2016 to 2019 (Table 44; McCracken 2020).

Table 44. Number of observed and estimated sea turtle interactions in the ASLL fishery between 2010 and 2019. The estimated interactions are from McCracken (2020).

Year	% Observer Coverage	Observed	Estimated
2010	25.0%	0	-
2011	33.0%	0	0
2012	19.8%	0	0
2013	19.4%	0	0
2014	19.4%	0	0
2015	22.0%	0	0
2016	19.4%	1	4
2017	20.0%	0	3
2018	17.5%	2	5
2019	15.7%	0	0
Grand Total		3	12

Demographic Patterns of Exposure

Hawksbill sea turtles encountered in the ASLL fishery are small and considered juveniles. Compared to green and loggerhead sea turtles, hawksbill sea turtles may exhibit faster growth rates and earlier maturation ages. Analysis using skeletochronology estimated that Hawaii hawksbills reach sexual maturity at 17–22 years, at a size of 78.6 cm SCL (Snover et al. 2013). However, using bomb ^{14}C dating technique, Van Houtan et al. (2016a) estimated age at sexual maturity of hawksbills in Hawaii to be 29 years (range 23–36). The three observed hawksbill sea turtles captured in the ASLL measured 32, 39 and 40 cm SCL.

As described in the Status of Listed Resources section, recent research using mtDNA markers has shown several distinct hawksbill nesting stocks in the West, Central, and East Pacific (Gaos et al. 2016, 2018; Vargas et al. 2016; PIFSC unpublished data). These studies have also revealed an exceptionally large proportion of shared mtDNA haplotypes among nesting populations in the Pacific Ocean, which complicates BMSA and limits robust assessments of nesting stock contributions (Gaos et al. 2016). The confounding issues posed by shared mtDNA nesting haplotypes, large BMSA confidence intervals and limited sample sizes suggest that continued research is needed to characterize the stock structure and natal origins of hawksbills at foraging habitats in the Pacific Ocean.

Spatial and Temporal Patterns of Exposure

We analyzed the location and dates of the observed hawksbill sea turtle interactions that occurred since 2016 to see if any spatial or temporal patterns emerged. Due to the spatial distance and small number of interactions, a reliable heat map could not be produced using ArcGIS. All three interactions occurred in close proximity to the LVPA; one hawksbill was captured north of Tutuila and two hawksbill sea turtles were captured south of Tutuila.

Temporally, the three observed hawksbill sea turtle interactions occurred during three different months: October (2016), May, and June (2018). In 2016, there were 20 active vessels in the fishery and the observed fishing effort (19.4% coverage) was highest during the month of October (69 sets). In 2018, there were 12 active vessels in the fishery and the observed fishing effort (17.5% coverage) was highest during the month of December (74 sets). The two hawksbill sea turtle interactions occurred in May and June when observed fishing effort was 38% and 23% lower in October and December. In May and June, only two observed vessels were fishing and observed effort was primarily in the southern end of the EEZ.

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interaction levels as previously described in the green sea turtle section above. The predictions were based on the recorded bycatch of hawksbill sea turtles in the ASLL fishery. As observing a hawksbill sea turtle bycatch event is extremely rare, the number of expected captures is estimated using the Bayesian model for extremely rare events (McCracken 2019a). Because a hawksbill bycatch event was not observed until 2016, including all the y-values of 0 from 2010 to 2015 resulted in estimated means and 95th percentiles that seemed too low considering that there have been three observed bycatch events in recent years (McCracken 2019a). Therefore, McCracken (2019a) estimated the expected number captures only using the time period from 2016–2017.

Table 45 includes the resulting predictions of future annual (mean and 95th percentile) captures for hawksbill sea turtles in the ASLL fishery (McCracken 2019a). We also report the anticipated mean number of annual captures associated with the 5-yr. running average and the cumulative number of anticipated captures over 5 years. There is no 95th percentile metric associated with the running average. As the maximum 5-yr. running average is less than the estimated annual mean number of captures, we focus our risk assessment on the annual mean number of captures of 6 and use a cumulative 5-year sum of 30 (6*5).

Table 45. Anticipated mean, 95th percentile (from McCracken 2019a; 2016-2017), maximum 5-year running average and maximum cumulative 5-year (2012-2019) exposures of hawksbill sea turtles with fishing gear in the ASLL fishery.

Period	**Mean**	**95th percentile**
Annual	6	14
Maximum 5-Yr. Running Average	2.4	N/A
Maximum 5-Yr. Running Sum	12	N/A

Response Analysis

Between 2006 and 2019, there were a total of 3 interactions with hawksbill sea turtles in the ASLL fishery. The corresponding at vessel mortality rate for hawksbill sea turtle interactions in this fishery is 100%. Based on the at-vessel mortality rate of 100% and the low number of interactions, we expect all hawksbill sea turtles that interact with the ASLL fishery to perish. Consequently, based on the 100% at-vessel and fishing mortality rate, the number of hawksbill sea turtles anticipated to interact with the ASLL fishery reflects the annual mean, annual 95th percentile, maximum 5-year running average and maximum 5-year running sum mortality estimates (Table 46).

Table 46. Anticipated mortalities of hawksbill sea turtles 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 (2012-2018) with fishing gear in the ASLL fishery.

Period	**Mortality estimate for the mean**	**Upper mortality estimate**
1	6	14
Maximum 5-Yr. Running Average	2.4	N/A
Maximum 5-Yr. Running Sum	12	N/A

Risk Analysis

Based on the three hawksbill sea turtles that interacted with the ASLL fishery from 2016 to June of 2019 as described in the Effects Analysis section, NMFS predicts that the ASLL fishery will interact with an annual mean of 6 (95th percentile: 14) hawksbill sea turtles per year. The maximum 5-yr. running average is 2.4 hawksbill sea turtles; however, as noted above, hawksbill sea turtles have only recently been observed interacting with this fishery and the 5 yr. running average may be an underestimate of future anticipated interactions. Therefore we use the 2016 to 2017 estimated mean of 6 as the maximum 5-yr. average and anticipate that up to 30 hawksbill sea turtles may be captured over 5 years. That is, over any 5-year period, we expect the fishery to capture, on average, 6 hawksbill sea turtles per year. Some years may be higher or lower than that value but a 5-year running average will not exceed that value and therefore the cumulative 5-year captures will not exceed 30 turtles and up to 60 turtles over 10 years. Over 40 years, assuming captures remain consistent with recent capture rates, up to 240 hawksbill sea turtles will be captured and based on historical captures, all would be expected to die as a result of their interaction with the fishery.

As noted above, the ASLL fishery may interact with 6 (95th percentile: 14) hawksbill sea turtles in a given year. As noted in the Status of Listed Resources section, we estimate that the Oceania population size in 2013 was 2,592,331 individuals greater than one year old and that the population is declining at a rate of 4.9% per year. Based on this, we estimate that the population size in 2022 is 1,667,887 individuals greater than one year old. Assuming total abundance for hawksbill sea turtles is 1,667,887 individuals, the ASLL fishery may interact with 0.00036% (95th percentile 0.00084%) of the total population in a given year. The mortality of 6 (95th percentile: 14) individual hawksbill sea turtles represents 0.00036% (95th percentile 0.00084%) of the total population. The 5-yr. running average of 6 turtles per year would result in the mortality of 0.00036% of the population annually, or up to 0.0018% of the population over five years and 0.0036% over 10 years.

As the population declines, the population size would be 1,021,791 in 2032; 625,976 in 2042, 383,490 in 2052, and 234,937 by 2062. When we consider 60 interactions over 10 years (extrapolated from the maximum 5-yr. running average of 6 per year), the rate of interaction with the fishery would be 0.0059% of the total population size estimated for 2031. Between 2032 and 2042 if the number of expected interactions stay the same, the fishery would interact with 0.0096% of the total population size estimated in 2042, 0.016% in 2052 and 0.026% of the Oceania population of hawksbill sea turtles estimated in 40 years (2062) based on a 4.9% per year decline. As we anticipate that 100% of the hawksbill sea turtles that interact with the ASLL fishery to die, these values represent both the proportions of the population expected to be impacted by the fishery and to die as a result of the fishery. In terms of annual impacts, in 2062, when the population has declined by 86%, the fishery will kill 0.0026% of the population, suggesting that without the fishery, the population rate of decline would be 0.04897%, which is not likely to be discernable from 0.049%.

Our assessment illustrates that the ASLL fishery interacts with (incidentally captures) a very small portion of the Oceania population of hawksbill sea turtles. At the 2022 projected abundance levels (1,667,887 individuals), incidentally capturing and killing an average of 6 hawksbill sea turtles per year constitutes death of 0.00036% of the population. If captures remain the same as the population declines, the fishery would incidentally capture and kill 0.0026% of the population in 40 years, which we consider to be the most conservative reasonable scenario as capture rates are likely to decline if the population declines by 86% over this time period as predicted from a 4.9% per year decline. If we added these mortalities back to the population, from 2022 to 2023 the population growth rate would increase to a 4.8997% per year decline; from 2022 to 2062 the population growth rate would increase to a 4.8989% per year decline. Given the estimated standard deviation for the population growth rate (0.018), and based on a power analysis for detecting differences in means with the same variance (Cohen 1988), more than one million years of data would be required to detect a statistically significant difference in these values. We consider the implications of this effect on Oceania hawksbill sea turtles in the Integration and Synthesis section of this opinion.

4.4.5 Oceanic Whitetip Shark

Exposure Analysis

Between 2006 and 2019, 1,412 confirmed oceanic whitetip sharks were observed caught in the ASLL fishery (Table 47; 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 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%). Due to uncertainty in applying observer data to full captures, we apply the most conservative reasonable estimate of 9.5% (upper 95% CI) of unidentified sharks to estimate the annual number of unidentified sharks that were likely oceanic whitetip sharks

Table 47. 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 (2019a, 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 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 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 judgment 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 44). If interaction rates are considered by dividing the number of observed 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 44, blue bars).

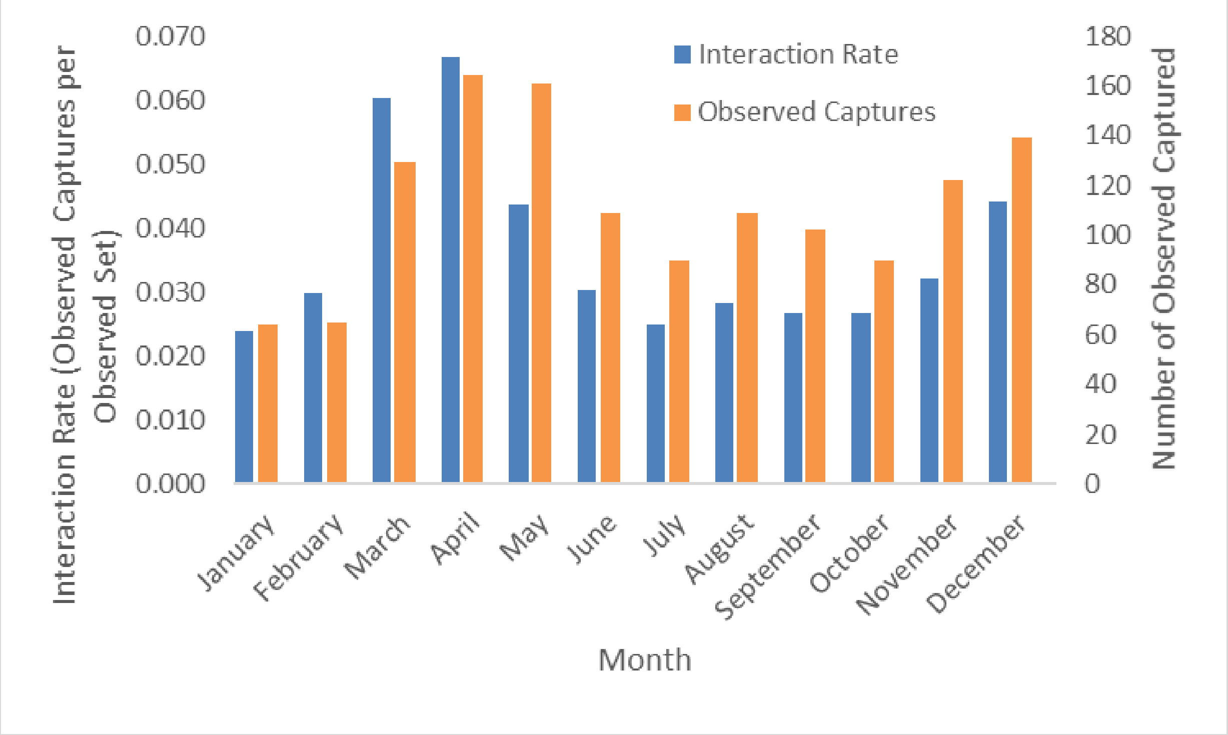


Figure 44. Total number of observed oceanic whitetip shark interactions (*n*= 1,344; orange bars) and interaction rates (observed shark captures divided by observed sets; blue bars) by month in the ASLL fishery, 2006 through 2019 (2nd quarter).

The highest estimated annual 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 captures), and 2019 (870 estimated captures; Figure 45). Overall, there is no trend in observed or estimated captures per year.

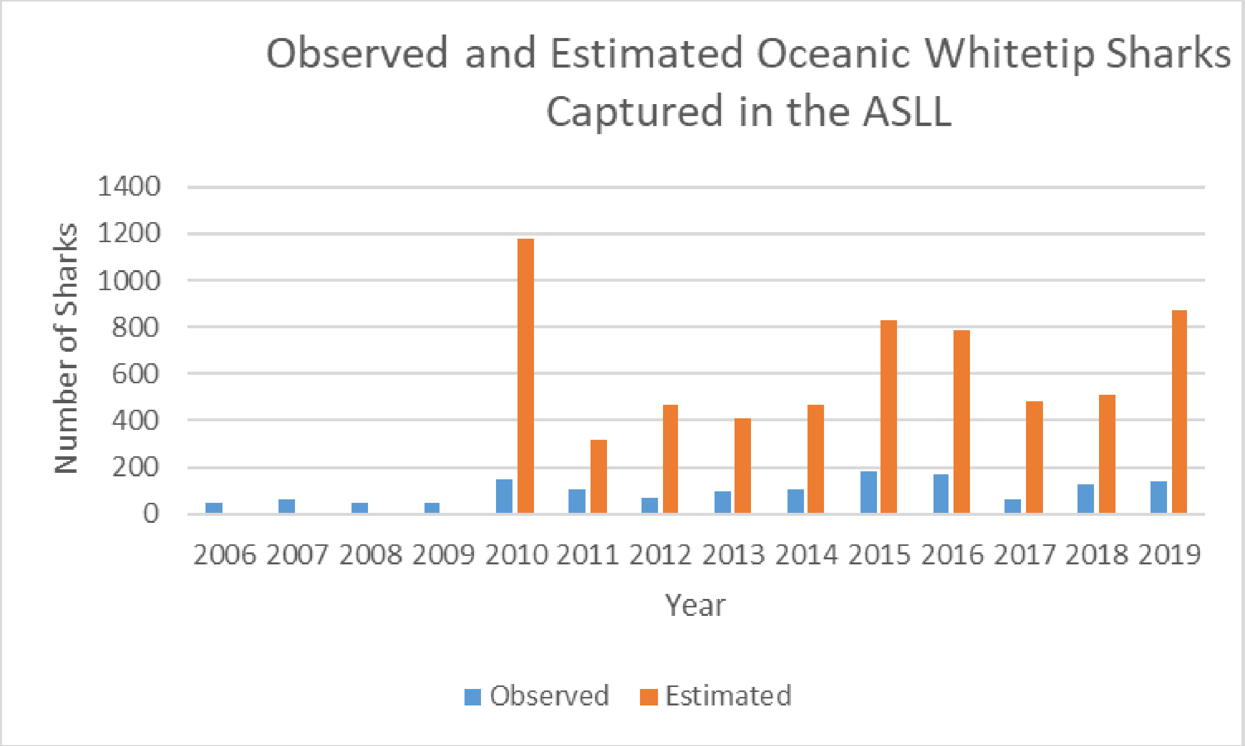


Figure 45. Observed (blue bars) and estimated (orange bars) oceanic whitetip shark interactions in the ASLL fishery, 2006-2019. Estimated numbers are from McCracken (2019a, 2020).

Using haul back locations, we mapped all sets where interactions occurred using ArcGIS (Figure 46) 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 47). The number of interactions is slightly higher in density in the southwest corner of the EEZ; however, this reflects 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.

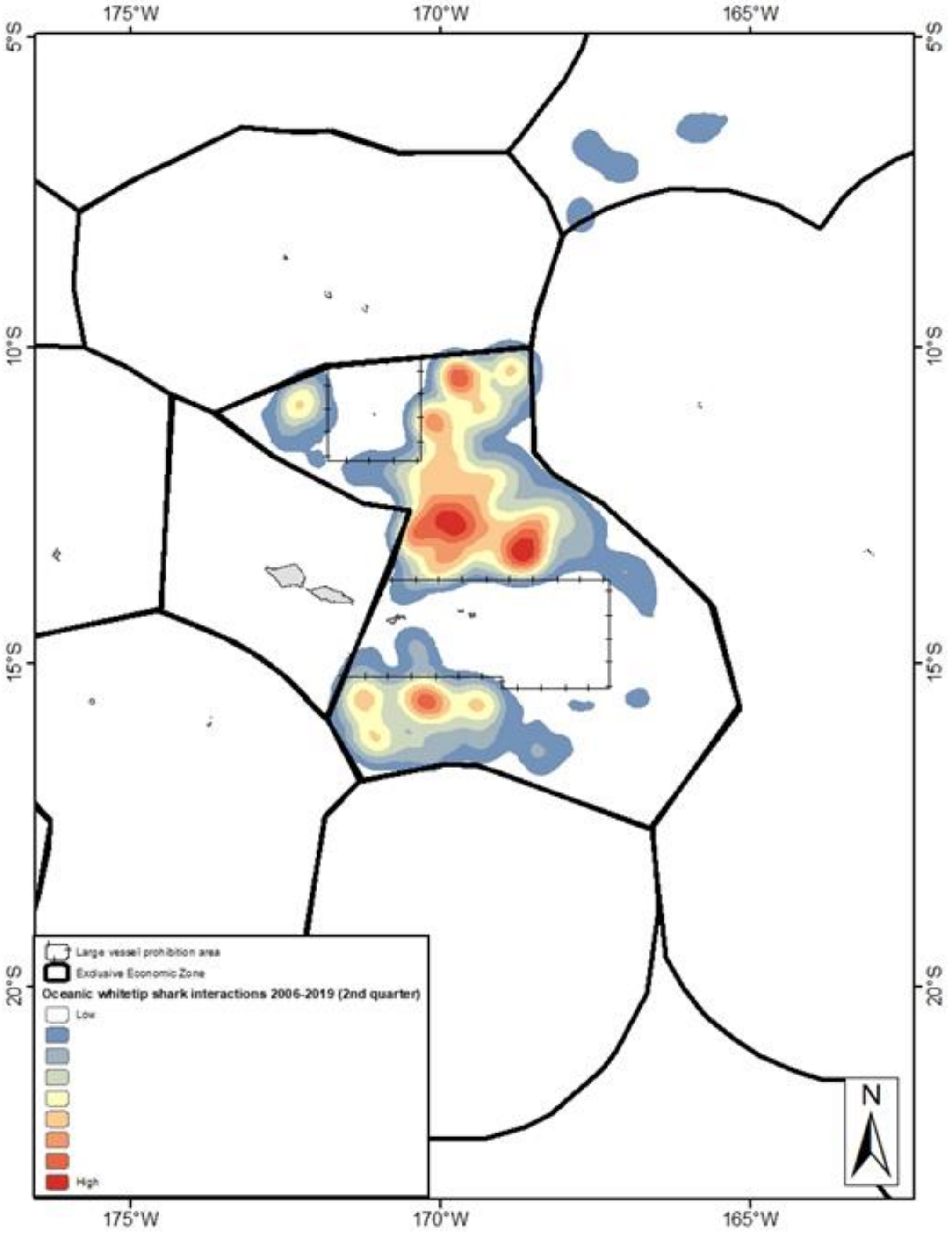


Figure 46. Geographical representation of aggregate oceanic whitetip shark interactions with the ASLL fishery between 2006 and 2019 (2nd quarter).

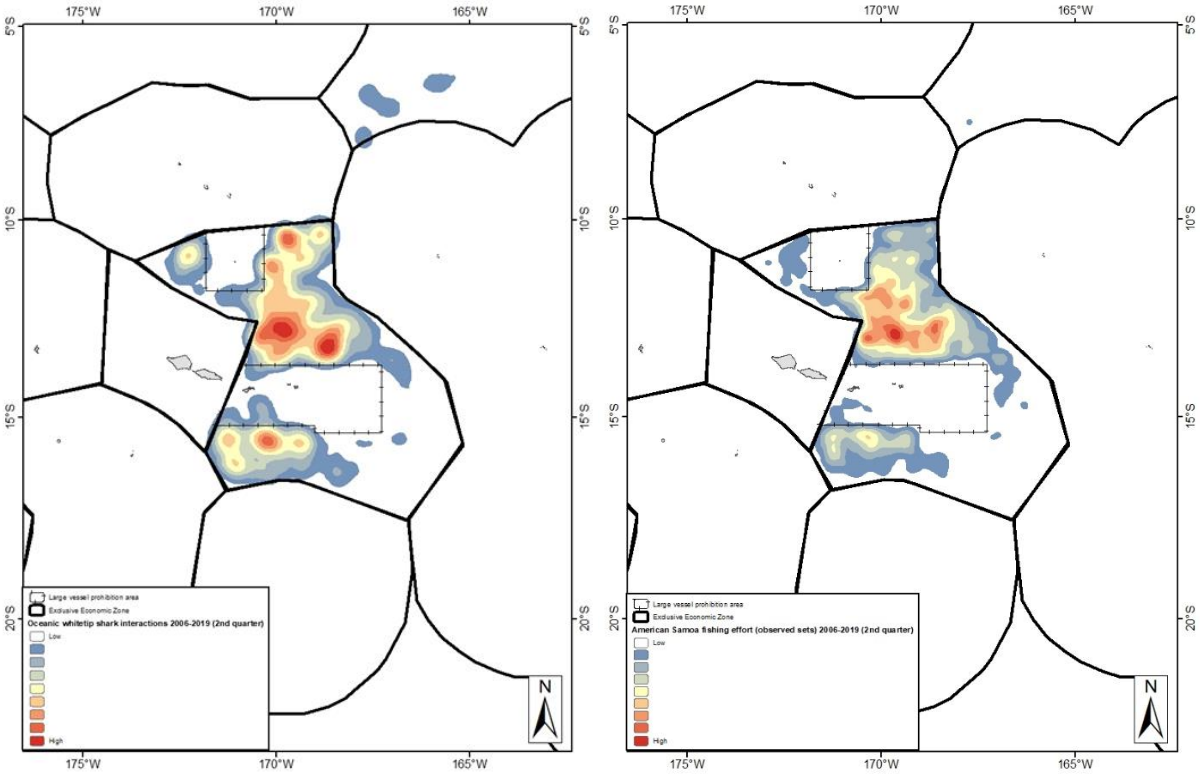


Figure 47. Side-by-side comparison of oceanic whitetip shark interactions (left; *n* =1,344 observed) versus fishing effort (right) from 2006 to 2019 (2nd quarter).

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interaction levels (McCracken 2019a; Table 48) 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 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 captures.

Table 48. 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.

Period	**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, while we focus on the mortality and injury that leads to latent mortality of oceanic whitetip sharks, we have accounted for sublethal responses in our analysis to the best of our ability using the best available information.

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 two that were 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 (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 49; 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 49. 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 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. However, at this time, we are unsure exactly how many sharks may bite off the line prior to be observed at the vessel. To the best of our ability, we have accounted for these delayed mortalities and other sublethal effects in our analysis from those individuals which are observed.

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; Table 49) 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 50. 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 48), NMFS predicts that 322 oceanic whitetip sharks on average may die from their incidental capture in one year (Table 50). If annual incidental captures reach the 95th percentile, which could happen any given year but would not likely be exceeded often, then we would expect 1,110 oceanic whitetip sharks to be incidentally captured in the ASLL fishery (see Table 48) resulting in the death of 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 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 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 start 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 (Table 51, Table 52). 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 51, Table 52). 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 approximately 1,292,023 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 51, Table 52). Our estimates assume that the estimated population trends in the WCPO is indicative of the trend for the entire Pacific population.

Table 51. 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-Yr. Running Average	2022	772,196	1,286,995	0.09	0.05	0.05	0.03
Max. 5-Yr. Running Average	2032	762,222	1,270,372	0.09	0.06	0.05	0.03
10-yr. based on 5-yr. running average	2032	762,222	1,270,372	0.92	0.55	0.48	0.29
Max. 5-Yr. Running Average	2042	752,377	1,253,965	0.09	0.06	0.05	0.03
10-yr. Mean Projected 20 years	2042	752,377	1,253,965	0.94	0.56	0.49	0.29
Max. 5-Yr. Running Average	2052	742,660	1,237,769	0.09	0.06	0.05	0.03
10-yr. Mean projected 30 years	2052	742,660	1,237,769	0.95	0.57	0.49	0.30
Max. 5-Yr. Running Average	2062	733,067	1,221,782	0.10	0.06	0.05	0.03
10-yr. 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 52. 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-Yr. Running Average	2022	783,632	1,306,052	0.09	0.05	0.05	0.03
Max. 5-Yr. Running Average	2032	812,356	1,35\3,925	0.09	0.05	0.05	0.03
10-yr. based on 5-yr. running average	2032	812,356	1,353,925	0.87	0.52	0.45	0.27
Max. 5-Yr. Running Average	2042	842,134	1,403,557	0.08	0.05	0.04	0.03
10-yr. Mean Projected 20 years	2042	842,134	1,403,557	0.84	0.50	0.43	0.26
Max. 5-Yr. Running Average	2052	873,002	1,455,005	0.08	0.05	0.04	0.03
10-yr. Mean projected 30 years	2052	873,002	1,455,005	0.81	0.48	0.42	0.25
Max. 5-Yr. Running Average	2062	905,002	1,508,338	0.08	0.05	0.04	0.02
10-yr. 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 oceanic whitetip sharks, as a species, in the Integration and Synthesis section of this opinion.

4.4.6 Indo-west Pacific Scalloped Hammerhead Shark

Exposure Analysis

Between 2006 and 2019, 15 Indo-west Pacific scalloped hammerhead sharks were observed caught in the ASLL fishery (NMFS Observer Program unpublished data; Table 53). 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 hammerhead shark is a smooth or scalloped hammerhead. In this situation, observers are instructed to use the unidentified hammerhead classification.

When interactions are adjusted to account for the percentage of observer coverage and for the proportion of unidentified sharks that were probably Indo-west Pacific scalloped hammerhead sharks, approximately 56 Indo-west Pacific scalloped hammerhead sharks are likely to have been captured in the fishery over the 10-year period from 2010 to 2019 (Table 54). Lastly, the number of unidentified sharks prorated for Indo-West Pacific scalloped hammerhead sharks was calculated using the Wilson Score Method without continuity correction (Newcombe 1998) with a mean of 59% (95% CI: 49% to 68%). Due to uncertainty in applying observer data to full captures, we apply the most conservative reasonable estimate and use the upper 95% CI of 68% to estimate that 18 of the 27 unidentified hammerhead sharks were potentially scalloped hammerhead sharks for a total estimate of 74 (56+[27*68/100] = 74.4 rounded to 74) Indo-West Pacific scalloped hammerhead sharks captured between 2010 and 2019.

Table 53. Number of interactions with hammerhead shark species in the ASLL fishery including the caught and release condition.

Year	IWP scalloped hammerhead shark	Dead	Alive	Unidentified hammerhead shark	Dead	Alive	Smooth hammerhead shark	Dead	Alive
2006	1	1	0	0	0	0	1	0	1
2007	1	0	1	0	0	0	0	0	0
2008	0	0	0	0	0	0	0	0	0
2009	0	0	0	0	0	0	0	0	0
2010	4	1	3	1	0	1	1	1	0
2011	2	1	1	0	0	0	3	2	1
2012	0	0	0	1	0	1	0	0	0
2013	0	0	0	0	0	0	1	0	1
2014	1	0	1	0	0	0	1	1	0
2015	1	1	0	1	0	1	0	0	0
2016	1	0	1	2	1	1	0	0	0
2017	1	0	1	0	0	0	0	0	0
2018	3	0	3	0	0	0	0	0	0
2019	0	0	0	0	0	0	2	2	0
Totals	13	3	10	5	1	4	8	6	2

Table 54. The observed and estimated number of interactions of Indo-west Pacific scalloped hammerhead, unidentified hammerhead, and smooth hammerhead sharks (McCracken 2019a, 2020). These estimates only include 2010-2019. Highlighted cells indicate no estimate was available, therefore, observed interactions were multiplied by the expansion factor to calculate the estimate. For example, for 2011, 100/33% (observer coverage) = 3, so (3 smooth hammerheads) x 3 = 9.

Year	**IWP Scalloped**		**Unid. Hammerhead**		**Smooth Hammerhead**		**Observer Coverage (%)**	**Expansion Factor**
	Obs.	**Est.**	**Obs.**	**Est.**	**Obs.**	**Est.**		
2010	4	17	1	5	1	5	-	-
2011	2	7	0	0	3	9	33%	3
2012	0	0	1	6	0	0	-	-
2013	0	0	0	0	1	5	19.4%	5.2
2014	1	6	0	0	1	6	-	-
2015	1	3	1	7	0	0	-	-
2016	1	8	2	9	0	0	-	-
2017	1	7	0	0	0	0	-	-
2018	3	8	0	0	0	5		
2019	0	0	0	0	2	9		
Grand Total	13	56	5	27	8	39	-	-

Demographic Patterns of Exposure

In the ASLL 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, safer and less stressful for the animal, and for the crew's safety. Therefore, observers estimate lengths based on their best professional judgment and with known measurements around the rail of the vessel. Of the 15 observed interactions of Indo-west Pacific scalloped hammerhead sharks between 2006 and 2019, length estimates were provided for seven individuals and gender was identified for three. The average estimated length was 213 cm (7.0 ft., range 5 ft. to 9 ft.) whereas the largest length estimated was 274 cm and this shark was a male.

When feasible, observers collect actual length measurements. Currently the protocol is to measure every third fish, regardless of species (NMFS 2015a). Two individuals were measured and identified as females. The fork lengths were 98 cm and 104 cm respectively. As described in the Status of Listed Resources, the age at maturity for scalloped hammerhead sharks varies geographically and they are generally considered mature about 200-250 cm total length (females) while males reach maturity at smaller sizes (range 128 – 200 cm). These lengths correspond to ages from 3.8 to 15.2 years. Therefore, the two measured female sharks are considered to be immature whereas the male shark with the estimated length of 213 cm is considered to be mature.

Spatial and Temporal Patterns of Exposure

All observed interaction data with Indo-west Pacific scalloped hammerhead sharks from the ASLL fishery were reviewed from 2006 through 2019. Comprehensively, most interactions occur in July and August, followed by March (Figure 48).

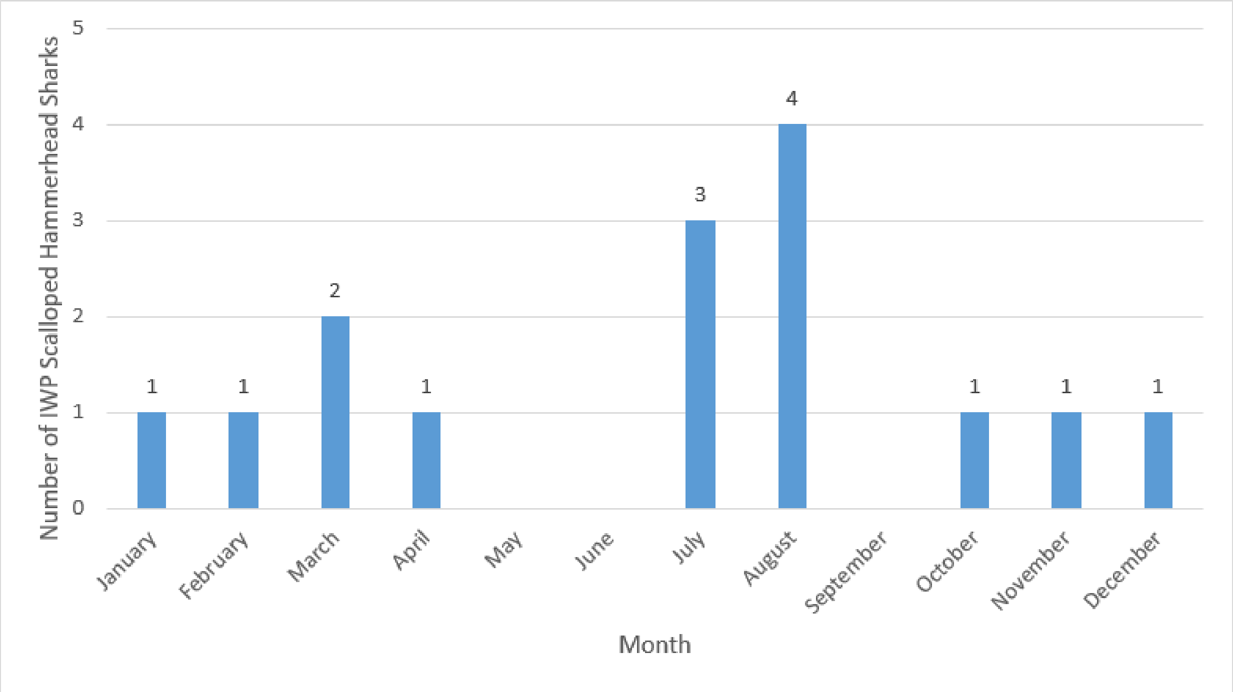


Figure 48. Total number of Indo-west Pacific scalloped hammerhead shark interactions (n=15) by month in the ASLL fishery, 2006 through 2019.

We plotted the 15 observed Indo-west Pacific scalloped hammerhead shark interactions that have occurred in the ASLL fishery between 2006 and 2019 (2nd quarter) in ArcGIS. No spatial patterns are apparent from the data due to the small number of data points. Spatial remoteness between the data do not allow for viable density mapping products to be produced.

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interaction levels (McCracken 2019a) using Bayesian inference techniques as described in the *Approach to the Assessment* section of this biological opinion. The predictions were based on the recorded bycatch of Indo-west Pacific scalloped hammerhead sharks, and unidentified hammerhead sharks in the ASLL fishery. McCracken (2019a) estimates a mean of 6 (95th percentile 19) unidentified hammerhead sharks to be captured in the ASLL annual. As described previously we anticipate 68% of unidentified hammerhead sharks to be scalloped hammerhead sharks, and therefore a mean of 4 (95th percentile 13) of the unidentified hammerhead sharks are anticipated to be scalloped hammerhead sharks. Table 55 lists the resulting anticipated exposure levels of future annual interactions (mean and 95th percentile) and the anticipated exposure levels when unidentified hammerhead sharks are considered as well (in parentheses; McCracken 2019a). We also report the anticipated mean number of annual mortalities associated with the maximum 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 or running sum.

NMFS predicts that the ASLL fishery will capture a total of 64 Indo-west Pacific scalloped hammerhead sharks over the next 10 years or 94 inclusive of unidentified hammerhead sharks. Over a 40-year time horizon, if captures remain consistent with historical numbers, we would anticipate a cumulative total of 256 captures or 376 inclusive of unidentified hammerhead sharks.

Table 55. Anticipated future exposure levels for Indo-west Pacific scalloped hammerheads with prorated unidentified records for the ASLL fishery including the anticipated mean, 95th percentile (from McCracken 2019a; 2012-2017), and maximum 5-year running average and maximum cumulative 5-year (2012-2019). Numbers in parentheses are inclusive of unidentified sharks assigned to for Indo-west Pacific scalloped hammerheads.

Period	Mean	95th percentile
1	5 (9)	15 (28)
Maximum 5-Yr. Running Average	6.4 (9.0)	N/A
Maximum 5-Yr. Running Sum	32 (45)	N/A

We believe the proportion of unobserved interactions is very small and the short-term (survival) and long-term (fitness) effects are minimal. For this biological opinion, we have done everything possible to account for unobserved interactions and include their effects in our analyses (e.g. rounding up to the nearest whole number of mortalities, assuming all serious injuries result in mortality, etc.).

Response Analysis

Between 2010-2019 we estimate the ASLL fishery had 74 interactions with Indo-west Pacific scalloped hammerhead sharks inclusive of unidentified hammerhead sharks. We calculated overall mortality rates using a known at-vessel mortality rate based on NMFS observer data from 2006 to 2019 in the ASLL fishery and a post release mortality metric for obligate ram-ventilation elasmobranchs captured on longline gear (Dapp et al. 2016). Most confirmed Indo-west Pacific hammerhead sharks were released alive (73%) and no sharks were recorded as kept. Average at-vessel mortality of Indo-west Pacific hammerhead sharks is 26.7% in the ASLL fishery. However, the publicly available data compiled by Dapp et al. (2016), estimate 37.6% at-vessel mortality based on the gear type (longline) and the respiratory mode of the animals (i.e., obligate ram-ventilation). We will use 26.7% because it is a direct calculation using data from this fishery on this DPS.

At this time, metrics related to post release mortality have not been determined for Indo-west Pacific scalloped hammerhead sharks in any fishery. We therefore reviewed the scientific literature for scalloped hammerhead sharks and potential surrogate species in the fisheries that best exemplify the same operational and handling methods practiced in the ASLL fishery. A structured literature search performed by Dapp et al. (2016) assessed how different gear types and respiratory modes affect at-vessel and post-release mortality of 83 elasmobranch species. Consequently, Dapp et al. (2016) estimate that individuals of many scalloped hammerhead species released alive, will die given the average post-release morality percentage (19.5%) for obligate ram-ventilators. However, it should be noted that this estimate is derived from a sample size of less than five sharks, 17% of the longline studies came from benthic longline sets (i.e., different gear), and confidence intervals were not calculated. At this time, this is the best available scientific and commercial data available in the literature as we did not find other applicable post release mortality values in our systematic review. Therefore, the post release mortality rate of 19.5% will be used in our analyses for this species.

Combining the at-vessel (26.7%) and the post-interaction (19.5%) mortality rates, the total mortality rate is 41.0% (0.267+0.195[1-0.267]; using the same formula as noted in the oceanic whitetip shark section above). When we apply the total mortality rates to the number of Indo-West Pacific scalloped hammerhead sharks that are likely to be captured in the ASLL fishery each year (Table 56), the number of Indo-West Pacific scalloped hammerhead shark mortalities we would expect are presented in Table 56. We note, as the estimated rate (i.e., 41%) for all observed identified and unidentified hammerhead shark records falls within the range presented here, we did not add another column to this table to illustrate those specific metrics.

As previously discussed, NMFS predicts that the ASLL fishery will capture a total of 90 Indo-West Pacific scalloped hammerhead sharks over the next 10 years and of those, 37 are expected to die. Over a 40-year time horizon, if captures remain consistent with historical numbers, we would anticipate a cumulative total of 360 captures with 148 mortalities.

Table 56. The estimated number of Indo-West Pacific scalloped hammerhead sharks that are likely to die as a result of being captured in the ASLL per year assuming a total mortality of 41.0%. 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. This uses the known 26.7% at-vessel mortality rate (calculated from observer data) and an estimated post-release mortality of 19.5% (Dapp et al. 2015). Numbers in parentheses are inclusive of unidentified hammerhead sharks.

	Estimated Indo West Pacific scalloped hammerhead shark Mortalities in the ASLL	
Rate	**Mean with Estimated Total Mortality of 41.0%**	**95th Percentile with Estimated Total Mortality of 41.0%**
Annual	2 (4)	6 (12)
5-Yr. Running Annual Average	2.6 (3.7)	N/A
5-Yr. Cumulative Sum	13 (18)	N/A

Risk Analysis

Scalloped hammerhead sharks are commonly caught in artisanal fisheries, recreational fisheries, gillnet fisheries, and purse seine fisheries, but are not considered to be vulnerable to longline fisheries (Miller et al. 2014). Nevertheless, based on historical captures we estimate that the fishery would capture a maximum 5-year running average of 9.0 sharks with an average of 3.7 of them dying each year. Over 5 years, the maximum 5-year running sum is 45 captures with 18 of them dying as a result of their capture. Over a 40-year time horizon, if captures remain consistent with historical numbers we would anticipate a cumulative total of 360 captures and we would anticipate that 148 would die as a result of their interactions. Although Indo-West Pacific scalloped hammerhead sharks occur across a large geographic area, we do not know if this DPS forms discrete populations and, if so, which of these populations is likely to be exposed to the ASLL fishery. The information available suggests that these scalloped hammerhead sharks represent a single population but there are no abundance estimates specific to this DPS (NMFS 2014, 2018b). The global abundance of adult scalloped hammerhead sharks based on effective population size is 280,000 (Miller et al. 2014)

As discussed previously in the Status of Listed Resources, Pacoureau et al. (2020) indicates a 67% decline from 1970 to 2018 equating to a 2.31% decline per year for the global population and we estimated a 2.28% decline for the Indo-West Pacific scalloped hammerhead shark based on ocean basin trends reported in Pacoureau et al. (2020). We simulated populations declining at a rate of 2.28% per year for a range of adult abundances from 28,000 (i.e. 10% of the global abundance) to 280,000 (i.e. 100% of the global abundance). For each starting population size, we projected the populations out 40 years at 2.28% per year declines. We then removed 3.6 mortalities that might be attributable to the ASLL each year and recalculated the population trajectories. Regardless of the starting population size, adding back the mortalities did not change the population trajectories, the new trajectories were -2.28% per year. We consider the implications of this effect on Indo-West Pacific scalloped hammerhead sharks in the Integration and Synthesis section of this opinion.

4.4.7 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 like leatherback sea turtles, 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 57 below. Between 2006-2009, 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 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 captures of protected species through 2019 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, *Manta birostris*. Therefore, the estimate of giant manta ray interactions may include the reef manta ray.

Table 57. 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.**
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	-	-
2019	0	0	4	24	-	-
Grand Total	12	64	11	64	11	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 58). 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 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 59). 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 58. The proportion of estimated 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 59. The proportion of estimated 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 (2nd quarter) 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 60) 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.

Response Analysis

While we use the 12 observed giant manta rays captured from 2010 to 2019 to estimated anticipated captures, we include the one observed giant manta ray in 2009 to increase the sample size for our estimates of mortality associated with capture in the ASLL. 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%.

Table 60. 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.

Period	Mean	95th percentile
1	8.7 (11)	28 (33)
Maximum 5-Yr. Running Average	8.4 (11.4)	N/A
Maximum 5-Yr. Running Sum	42 (57)	N/A

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 breathe, 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; Table 61). 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 61. Cumulative number of giant manta ray, *Mobulidae* spp., and unidentified rays captured in the ASLL fishery from 2006 through 2019 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 62. 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	**Mortality estimate for the mean**	**Mortality estimate for the 95th Percentile**
1	3 (3)	8 (10)
Maximum 5-Yr. Running Average	2.4 (3.3)	N/A
Maximum 5-Yr. Cumulative Sum	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 62). 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 456 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 2021a), and the ASLL and DSLL supplements (NMFS 2022b, 2022c) 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 63). Six of these were identified in the analysis conducted for the United States WCPO purse seine fishery biological opinion (NMFS 2021a), 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 Resources 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 2021a) 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 49). 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 (2020a) 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.

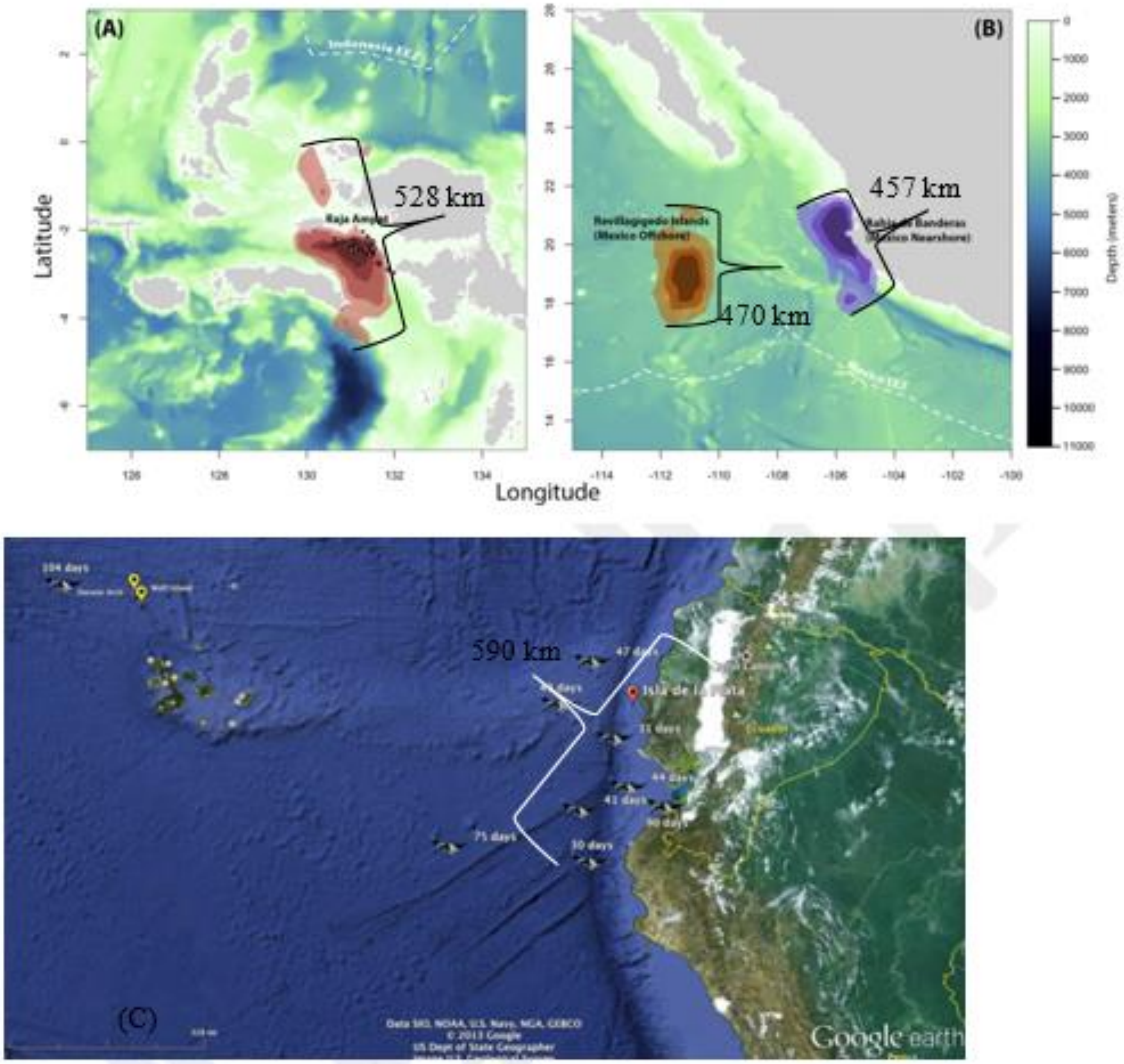


Figure 49. Extents of giant manta ray subpopulations based on tracking tagged individuals. Black brackets (white brackets in C) highlight the longest dimension of the area of occupancy with the length noted in kilometers (km). Locations: A) Raja Ampat, Indonesia (Stewart et al. 2016a); B) Revillagigedo 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 2021a), 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 13 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 63; Figure 50).

For the Tuvalu and Kiribati-Line Islands subpopulations in Table 63, no observed captures occurred within 500 km of the center point of these two subpopulations, and 1 (Tuvalu) or 2 (Kiribati-Line Islands) occurred 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 63. The locations of these 5 subpopulations are depicted in Figure 50 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 63, Figure 50).

Table 63. 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 2021a). 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 (7)	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

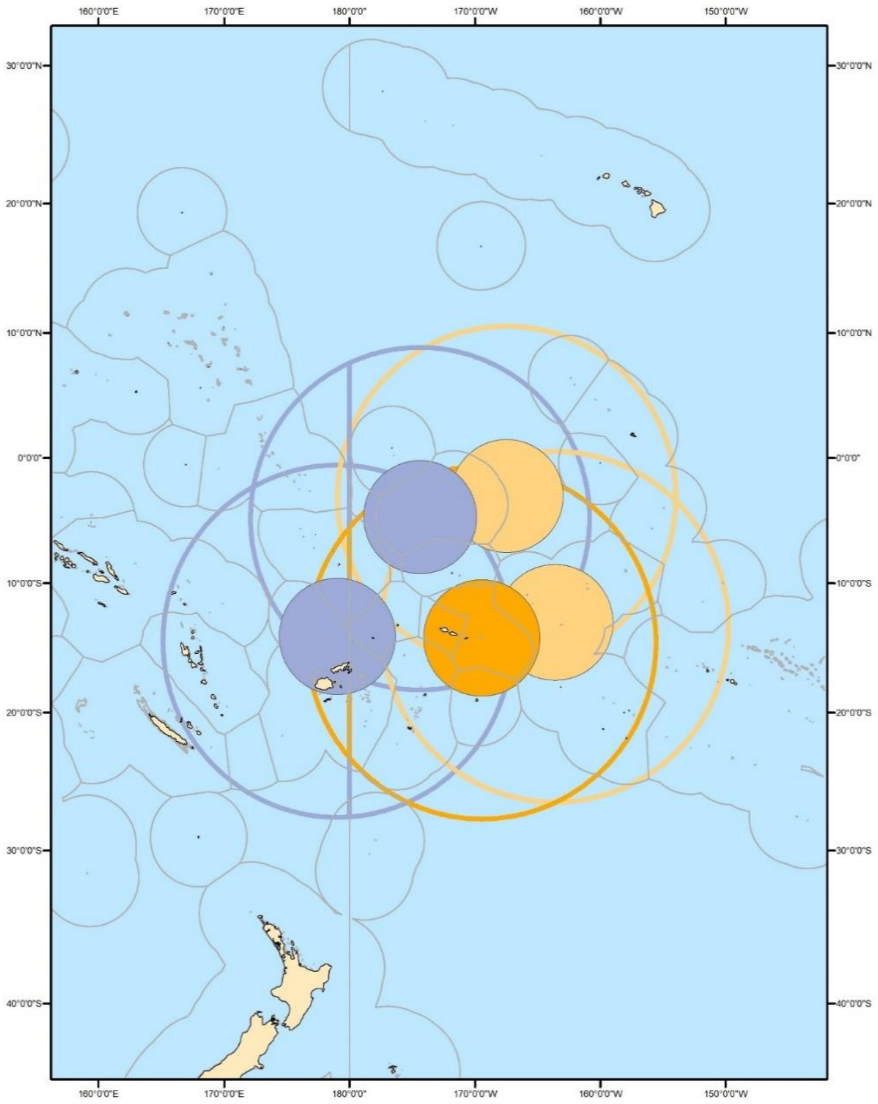


Figure 50. Map of fishing area for the ASLL fishery showing the five purported giant manta ray subpopulations potentially impacted by the ASLL fishery. Dark orange circle represents the 500 km boundary of the purported American Samoa subpopulation, the dark orange open circle represents the 1,500 km 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 (2021a) concluded that the r_{max} values published by Rambahiniarison et al. (2018) likely represent the best r_{max} value available for that consultation ($r_{max} = 0.019$). NMFS (2021a) concluded that the r_{max} value published by Ward-Paige et al. (2013; $r_{max} = 0.042$) was also reasonable, but there are some concerns over the methodology because the density dependent assumptions used to estimate survival to maturity result in somewhat illogical r_{max} values, whereby the lowest adult mortality rate and highest fecundity would combine to result in a low r_{max} value of 0.012 contrasted with the mean estimate of 0.116 estimated by Dulvy et al. (2014) using the same values.

Lastly, NMFS (2021a) determined that the r_{max} estimated by Dulvy et al. (2014) was 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[7]. For this analysis, we agree with the findings of NMFS (2021a), follow the same methodology, 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 Resources, we assume the viable giant manta ray subpopulations are likely to be larger in abundance, potentially greater than 1,000 individuals, and we focus our analysis on these population sizes but include the smaller population sizes since they may also represent exploited populations that may be at higher risk of extirpation. Within 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 60) multiplied by the mean estimated post-interaction mortality rate of 0.29. We consider the maximum 5-yr. running average as the

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[7] 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.

outcome that is reasonably likely to occur, as this level of interaction has been estimated (based on observed records) to have occurred. Exploration of the upper 95th percentile values represents the most conservative reasonable 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 (Table 64):

> 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 64. 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 65). For the other four subpopulations potentially impacted by the fishery, all continue to increase at all initial population abundances (Table 66 and Table 67). 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 65) 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 66). The remaining subpopulations had increasing population trajectories at all initial abundances and values of r_{max} (Table 67).

The data available are not sufficient to answer the question: "how many giant manta ray subpopulations 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 subpopulations 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 65. 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 (95th percentile). 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										

Initial Abundance	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
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										
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 66. 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

Initial Abundance	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**
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 67. 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

Initial Abundance	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**
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 listed species globally and within the Action Area given the effects of activities such as commercial fisheries, direct harvests and modification or degradation of habitat caused by marine debris and climate change. The pre-existing condition of these species 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 continuing the ASLL fishery poses to Central South Pacific green sea turtles, Central West Pacific green sea turtles, East Indian-West Pacific green sea turtles, East Pacific green sea turtles, Southwest Pacific green sea turtles, leatherback sea turtles, the Mexico breeding population of olive ridley sea turtles and threatened populations of olive ridley sea turtles, Indo-West Pacific scalloped hammerhead sharks, oceanic whitetip sharks, giant manta rays 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 East Pacific Green Sea Turtle

As described in the Status of Listed Resources, the East Pacific green sea turtle is listed as threatened and nesting has been steadily increasing at the primary nesting sites in Michoacan, Mexico, and in the Galapagos Islands since the 1990s (Delgado and Nichols 2005; Senko et al. 2011). Nesting trends at Colola have continued to increase since 2000 with the overall East Pacific green turtle population also increasing at other nesting beaches in the Galapagos and Costa Rica (NMFS and FWS 2007a; Wallace et al. 2010a). Based on nesting beach data, the current adult female nester population for Colola, Michoacan is 11,588 females, which makes this the largest nesting aggregation in the threatened East Pacific green sea turtle, comprising nearly 58% of the total adult female population. The total number of adult females for the entire East Pacific green sea turtle is estimated at 20,112 nesting females (Seminoff et al. 2015) and based on this we estimate there are 3,580,207 (range 2,473,546 to 5,814,512) total individuals greater than one year old.

East Pacific green sea turtles are exposed to a variety of natural threats both at their nesting beaches and in the open ocean. The beaches where turtles nest and the nest themselves are threatened by hurricanes, tropical storms, storm surges, sand accretion, and rainfall associated with storms. The largest threat on nesting beaches to the East Pacific green sea turtle is reduced availability of habitat due to heavy armament and subsequent erosion, and coastal development. Incidental capture in artisanal and commercial fisheries is a significant threat to the survival of green turtles throughout the eastern Pacific Ocean. Significant bycatch has been reported in artisanal gill net and longline shark and mahi mahi fisheries operating out of Peru (Kelez et al. 2003; Alfaro-Shigueto et al. 2006) and, to a lesser extent, Chile (Donoso and Dutton 2010).

As described in the Environmental Baseline, past and present fisheries interactions have been, and continue to be, a threat to East Pacific green turtles within the Action Area. Bycatch of green sea turtles in this DPS occurs in many fisheries throughout the geographic oceanic range of this species. While mitigation and minimization measures have reduced fisheries sea turtle bycatch in the United States in recent years, large numbers of East Pacific green sea turtles are still routinely captured in international, federal and state commercial fisheries that target other species. In addition, warming sea surface temperatures may lead to potential fitness consequences in sea turtles resulting from altered seasonality and duration of nesting (Pike et al. 2006). Sea turtles may also expand their range as temperature-dependent distribution limits change (McMahon and Hays 2006; Poloczanska et al. 2009a). Further, sea turtles have temperature-dependent sex determination, and many populations produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Newson et al. 2009; Patricio et al. 2017; Jensen et al. 2018). Increases in both the frequency and severity of storms as a result of climate change may lead to further loss of nesting habitat.

Environmental changes associated with climate change are occurring within the Action Area and are expected to continue into the future. Marine populations that are already at risk due to other threats are particularly vulnerable to the direct and indirect effects of climate change.

As described in the *Exposure Analysis*, hooking and entanglement are the most significant stressors that green sea turtles are exposed to directly from the action. Since April of 2006, 45 green sea turtles have been observed caught in the ASLL fishery, and all but two were dead. The two green sea turtles that were released alive were released with no gear attached.

Because the genetic data do not allow us to reliably predict how many individuals will be captured from each green turtle DPS, we analyzed as if all are East Pacific. The action results in the capture of 19.2 and mortality of 18 East Pacific green sea turtles per year based on the maximum 5-year running average. At projected abundance levels (3,580,207 individuals); this constitutes death of 0.0043% of the population per year. As this DPS has been experiencing increasing trends, this level of impact will not change the DPS' overall trajectory.

Therefore, we are reasonably certain the population trend will continue to be positive with the continued authorization of the ASLL fishery as currently managed. Because East Pacific green sea turtle abundance will continue to grow with the proposed action, we are reasonably certain it will not cause material changes having biological consequences to the DPS' numbers, reproduction, or distribution. In accordance with Section 1.3.1 (Jeopardy Analyses) above, NMFS does not anticipate the ASLL fishery will reduce appreciably the likelihood of the survival or recovery of East Pacific green sea turtles in the wild by reducing their reproduction, numbers, or distribution.

5.2 Central South Pacific Green Sea Turtle

As described in the Status of Listed Resources, the Central South Pacific green sea turtle is listed as endangered and population trends are poorly understood. Based on available data, we estimate the DPS has 2,677 adult females (Seminoff et al. 2015) and based on this we estimate a total population size of 631,745 (range 323,514 to 1,669,441) individuals greater than one year old as of 2013. No sites have long-term monitoring programs, and no single site has had standardized surveys for even five continuous years to allow for trend estimation. Based on historic declines and ongoing threats, this DPS is anticipated to be declining (Seminoff et al. 2015). As a proxy, we use the declining trend in CPUE for green sea turtles captured in the WCPO longline fisheries (-5.8% per year) as an estimate of the DPS' trend (Figure 26). At this rate of decline, we estimate the population size in 2022 to be 420,939.

Central South Pacific green sea turtles are exposed to a variety of natural threats both at their nesting beaches and in the open ocean. Depredation may have been a factor that contributed to the historical decline of this population. The best available data suggest that current nest and hatchling predation on several Central South Pacific green sea turtle nesting beaches and in water habitats is a potential threat to this population.

Directed harvest of turtles and their eggs is an ongoing problem in the Central South Pacific in American Samoa, Fiji, Kiribati (e.g., Phoenix Islands), Tuvalu, Tokelau, and the Cook Islands (Balazs 1983; Tuato'o-Bartley et al. 1993; Weaver 1996a, 1996b; NMFS and FWS 1998a; Obura and Stone 2002; Alefaio et al. 2006). Commercial harvest (a major threat), as well as subsistence and ceremonial harvest, are all contributing factors. Other anthropogenic threats include incidental capture in artisanal and commercial fisheries.

As described in the Environmental Baseline, past and present fisheries interactions may have been, and may continue to be, a threat to Central South Pacific green sea turtles within the Action Area. Bycatch of green sea turtles occurs in many fisheries throughout the geographic oceanic range of this DPS. While mitigation and minimization measures have reduced fisheries sea turtle bycatch in the United States in recent years, large numbers of Central South Pacific green sea turtles may still be captured in international and United States commercial fisheries that target other species. These activities are reasonably likely to continue, and may increase over time because of increased human population and increased human consumption of fish products.

In addition, warming sea surface temperatures may lead to potential fitness consequences in sea turtles resulting from altered seasonality and duration of nesting (Pike et al. 2006). Sea turtles may also expand their range as temperature-dependent distribution limits change (McMahon and Hays 2006; Poloczanska et al. 2009a). Further, sea turtles have temperature-dependent sex determination, and many populations produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Newson et al. 2009; Patricio et al. 2017; Jensen et al. 2018).

Environmental changes associated with climate change are occurring within the Action Area and are expected to continue into the future. Marine populations that are already at risk due to other threats are particularly vulnerable to the direct and indirect effects of climate change.

As described in the *Exposure Analysis*, hooking and entanglement are the most significant stressors that green sea turtles are exposed to directly from the action. Since April of 2006, 45 green sea turtles have been observed caught in the ASLL fishery, and all but two were dead. The two green sea turtles that were released alive were released with no gear attached.

Because the genetic data do not allow us to reliably predict how many individuals will be captured from each green turtle DPS, we analyzed as if all are Central South Pacific.

The action results in the capture of 19.2 and mortality of 18 Central South Pacific green sea turtles per year based on the maximum 5-year running average. At the 2022 projected abundance levels (420,939 individuals), the level of mortality constitutes death of 0.0043% of the population. If captures remain the same as the population declines, the fishery would incidentally capture and kill 0.044% of the population in 40 years, which we consider to be the most conservative reasonable scenario, as 1) capture rates are likely to decline if the population declines by 90% over this time period as predicted from a 5.8% per year decline and 2) this assumes all captures are Central South Pacific green turtles. Assuming the population declines include the historic impact of the ASLL, removing the fishery would result in population declines of between 5.7929 to 5.798% with the lower rate of decline representative of the population growth rate over 40 years. These values are well within the 95% CI for our estimate of the rate of the population decline (-7.6% to -3.9%).

A power analysis indicates more than 150,000 years of data would be required to detect this difference and therefore the difference is not statistically detectable given our predictive capabilities. Thus, the trajectory of the population is not substantially changed by the removal of mortalities from the ASLL fishery. Because Central South Pacific green sea turtle abundance will not be substantially impacted by the proposed action, we are reasonably certain it will not cause material changes having biological consequences to the DPS' numbers, reproduction, or distribution. In accordance with Section 1.3.1 (Jeopardy Analyses) above, NMFS does not anticipate the ASLL fishery will reduce appreciably the likelihood of the survival or recovery of Central South Pacific green sea turtles in the wild by reducing their reproduction, numbers, or distribution.

5.3 Southwest Pacific Green Sea Turtle

As described in the Status of Listed Resources, the Southwest Pacific green sea turtle is listed as threatened and nesting occurs in many islands throughout their range, but only two nesting areas (Raine Island and Heron Island) have long-term (>15 years) annual indices of nesting abundance. Total adult female abundance for this DPS is estimated at 83,058, based on this we estimate a total population size of 14,762,154 (range 10,137,513 to 23,139,940) individuals greater than one year old. The DPS is anticipated to be increasing in abundance (Seminoff et al. 2015).

Southwest Pacific green turtles are exposed to a variety of natural threats both at their nesting beaches and in the open ocean. This DPS of green turtles are vulnerable to harvest throughout Australia and neighboring countries such as New Caledonia, Fiji, Vanuatu, Papua New Guinea, and Indonesia (Limpus 2009). Nesting turtles are also vulnerable to the Queensland East Coast Trawl Fisheries and the Torres Strait Prawn Fishery, and to the extent they forage west of Torres Strait, they are also vulnerable to the northern prawn fishery. Total mortality of green turtles in fisheries bycatch is not known because there is not reliable reporting of threatened species bycatch in Australian commercial fisheries (Limpus 2009).

As described in the Environmental Baseline, past and present fisheries interactions may have been, and may continue to be, a threat to Southwest Pacific green sea turtles within the Action Area. Bycatch of green sea turtles occurs in many fisheries throughout the geographic oceanic range of this DPS. While mitigation and minimization measures have reduced fisheries sea turtle bycatch in the United States in recent years, large numbers of Southwest Pacific green sea turtles may still be captured in international and United States commercial fisheries that target other species. These activities are reasonably likely to continue, and may increase over time due to the effects of increased human population and increased human consumption of fish products.

In addition, warming sea surface temperatures may lead to potential fitness consequences in sea turtles resulting from altered seasonality and duration of nesting (Pike et al. 2006). Sea turtles may also expand their range as temperature-dependent distribution limits change (McMahon and Hays 2006; Poloczanska et al. 2009a). Further, sea turtles have temperature-dependent sex determination, and many populations produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Newson et al. 2009; Patricio et al. 2017; Jensen et al. 2018). Increases in both the frequency and severity of storms as a result of climate change may lead to further loss of nesting habitat.

Environmental changes associated with climate change are occurring within the Action Area and are expected to continue into the future. Marine populations that are already at risk due to other threats are particularly vulnerable to the direct and indirect effects of climate change.

As described in the *Exposure Analysis*, hooking and entanglement are the most significant stressors that green sea turtles are exposed to directly from the action. Since April of 2006, 45 green sea turtles have been observed caught in the ASLL fishery and all but two were dead. The two green sea turtles that were released alive were released with no gear attached.

Because the genetic data do not allow us to reliably predict how many individuals will be captured from each green turtle DPS, we analyzed as if all are Southwest Pacific. The action results in the capture of 19.2 and mortality of 18 Southwest Pacific green sea turtles per year based on the maximum 5-year running average. At projected abundance levels (14,762,154 individuals), this constitutes death of 0.0018% of the population per year. As this DPS has been experiencing increasing trends, this level of impact will not change the DPS' overall trajectory.

Therefore, we are reasonably certain the population trend will continue to be positive with the continued authorization of the ASLL fishery as currently managed. Because Southwest Pacific green sea turtle abundance will continue to grow with the proposed action, we are reasonably certain it will not cause material changes having biological consequences to the DPS' numbers, reproduction, or distribution. In accordance with Section 1.3.1 (*Jeopardy Analyses*) above, NMFS does not anticipate the ASLL fishery will reduce appreciably the likelihood of the survival or recovery of Southwest Pacific green sea turtles in the wild by reducing their reproduction, numbers, or distribution.

5.4 Central West Pacific Green Sea Turtle

As described in the Status of Listed Resources, the Central West Pacific green sea turtle is listed as endangered and there is insufficient long-term and standardized monitoring information to adequately describe abundance and the population trend. The limited available information suggests a nesting population decrease in some areas like the Marshall Islands, or unknown trends in other areas such as Palau, Papua New Guinea, the Marianas, Solomon Islands, or the FSM (Maison et al. 2010). Currently, NMFS estimates the Central West Pacific has approximately 51 nesting sites with a total of 6,518 nesting females (Seminoff et al. 2015) and based on this we estimate a total population size of 1,543,625 (range 821,772 to 3,592,842) individuals greater than one year old. Based on historic declines and ongoing threats, this DPS is anticipated to be declining (Seminoff et al. 2015). As a proxy, we use the declining trend in CPUE for green sea turtles captured in the WCPO longline fisheries (-5.8% per year) as an estimate of the DPS' trend (Figure 26). At this rate of decline, we estimate the population size in 2022 to be 1,028,533.

Central West Pacific green sea turtles are exposed to a variety of natural threats both at their nesting beaches and in the open ocean. Directed harvest of turtles and their eggs is an ongoing problem in the Central West Pacific in the CNMI, FSM, Guam, Kiribati (Gilbert Islands chain), Papua, Papua New Guinea, Republic of the Marshall Islands, and Palau (Eckert 1993; Hitipeuw and Maturbongs 2002; Philip 2002; Humber et al. 2014). In addition to the collection of eggs from nesting beaches, the killing of nesting females continues to threaten the stability of green turtle populations. Other anthropogenic threats include incidental capture in artisanal and commercial fisheries.

As described in the Environmental Baseline, past and present fisheries interactions may have been, and may continue to be, a threat to Central West Pacific green sea turtles within the Action Area. Bycatch of green sea turtles from this population occurs in many fisheries throughout the geographic oceanic range of this DPS. While mitigation and minimization measures have reduced fisheries sea turtle bycatch in the United States in recent years, large numbers of Central West Pacific green sea turtles may still be captured in international and United States commercial fisheries that target other species. These activities are reasonably likely to continue, and may increase over time due to the effects of increased human population and increased human consumption of fish products.

In addition, warming sea surface temperatures may lead to potential fitness consequences in sea turtles resulting from altered seasonality and duration of nesting (Pike et al. 2006). Sea turtles may also expand their range as temperature-dependent distribution limits change (McMahon and Hays 2006; Poloczanska et al. 2009a). Further, sea turtles have temperature-dependent sex determination, and many populations produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Newson et al. 2009; Patricio et al. 2017; Jensen et al. 2018).

Environmental changes associated with climate change are occurring within the Action Area and are expected to continue into the future. Marine populations that are already at risk due to other threats are particularly vulnerable to the direct and indirect effects of climate change.

As described in the Exposure Analysis, hooking and entanglement are the most significant stressors that green sea turtles are exposed to directly from the action. Since April of 2006, 45 green sea turtles have been observed caught in the ASLL fishery and all but two were dead. The two green sea turtles that were released alive were released with no gear attached.

Because the genetic data do not allow us to reliably predict how many individuals will be captured from each green turtle DPS, we analyzed as if all are Central West Pacific. The action results in the capture of 19.2 and mortality of 18 Central West Pacific green sea turtles per year based on the maximum 5-year running average. At the 2022 projected abundance levels (1,028,533 individuals), this level of mortality constitutes death of 0.0018% of the population. If captures remain the same as the population declines the fishery would incidentally capture and kill 0.018% of the population in 40 years, which we consider to be the most conservative reasonable scenario as 1) capture rates are likely to decline if the population declines by 90% over this time period as predicted from a 5.8% per year decline and 2) not all captures are expected to be from the Central West Pacific. Assuming the population declines include the historic impact of the ASLL, removing the fishery would result in population declines of between 5.7929 to 5.7982% with the lower rate of decline representative of the population growth rate over 40 years. These values are well within the 95% CI for our estimate of the rate of the population decline (-7.6% to -3.9%).

A power analysis indicates more than 1 million years of data would be required to detect this difference and therefore the difference is not statistically detectable given our predictive capabilities. Thus, the trajectory of the population is not substantially changed by the removal of mortalities from the ASLL fishery. Because Central West Pacific green sea turtle abundance will not be substantially impacted by the proposed action, we are reasonably certain it will not cause material changes having biological consequences to the DPS' numbers, reproduction, or distribution. In accordance with Section 1.3.1 (Jeopardy Analyses) above, NMFS does not anticipate the ASLL fishery will reduce appreciably the likelihood of the survival or recovery of Central West Pacific green sea turtles in the wild by reducing their reproduction, numbers, or distribution.

5.5 East Indian-West Pacific Green Sea Turtle

As described in the Status of Listed Resources, the East Indian-West Pacific green sea turtle is listed as threatened and there are four sites for which 15 or more years of recent data are available for annual nester abundance: Sabah Turtle Islands in Malaysia; Royal Navy Center in

Khram Island, Thailand; Redang in Terrengganu, Myanmar; and Thameela Islands, Myanmar. Only Sabah Turtle Islands represent a sizable nesting population, estimated at 7,011 in 2011 (Pilcher 2010b). The PVA indicates that the nesters from Sabah Turtle Islands in Malaysia, with an estimated 7,000 nesters, will likely continue to increase, while the nesters from the Royal Navy Center in Khram Island, Thailand (estimated 297 nesters), Redang in Terrengganu, Myanmar (estimated 278 nesters), and Thameela Islands, Myanmar (estimated 109 nesters) will likely continue to decline (Seminoff et al. 2015). The total abundance for this population is estimated at 77,009 adult females (Seminoff et al. 2015) and based on this we estimate total population size of 18,171,565 (range 9,227,181 to 43,725,766) individuals greater than one year old. Based on the analysis of Mazaris et al. (2017), we estimate this DPS is declining at a rate of 1.11% per year. At the mean rate of decline, we estimate the population size in 2022 to be 16,813,093.

East Indian-West Pacific green sea turtles are exposed to a variety of natural threats both at their nesting beaches and in the open ocean. The best available data suggest that current nest and hatchling predation on the East Indian-West Pacific green sea turtle is prevalent. Depredation of nests by feral animals is also widespread throughout the range of the species.

As described in the Environmental Baseline, past and present fisheries interactions may have been, and may continue to be, a threat to East Indian-West Pacific green sea turtle within the Action Area. Bycatch of green sea turtles occurs in many fisheries throughout the geographic oceanic range of this DPS. While mitigation and minimization measures have reduced fisheries sea turtle bycatch in the United States in recent years, large numbers of East Indian-West Pacific green sea turtles may still be captured in international and United States commercial fisheries that target other species. These activities are reasonably likely to continue, and may increase over time due to the effects of increased human population and increased human consumption of fish products.

In addition, warming sea surface temperatures may lead to potential fitness consequences in sea turtles resulting from altered seasonality and duration of nesting (Pike et al. 2006). Sea turtles may also expand their range as temperature-dependent distribution limits change (McMahon and Hays 2006; Poloczanska et al. 2009a). Further, sea turtles have temperature-dependent sex determination, and many populations produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Newson et al. 2009; Patricio et al. 2017; Jensen et al. 2018). Increases in both the frequency and severity of storms as a result of climate change may lead to further loss of nesting habitat.

Environmental changes associated with climate change are occurring within the Action Area and are expected to continue into the future. Marine populations that are already at risk due to other threats are particularly vulnerable to the direct and indirect effects of climate change.

As described in the *Exposure Analysis*, hooking and entanglement are the most significant stressors that green sea turtles are exposed to directly from the action. Since April of 2006, 45 green sea turtles have been observed caught in the ASLL fishery and all but two were dead. The two green sea turtles that were released alive were released with no gear attached.

Because the genetic data do not allow us to reliably predict how many individuals will be captured from each green turtle DPS, we analyzed as if all are East Indian-West Pacific. The action results in the capture of 19.2 and mortality of 18 East Indian-West Pacific green sea turtles per year based on the maximum 5-year running average. At the 2022 projected abundance levels (16,813,093 individuals), this level of mortality constitutes death of 0.00011% of the population. If captures remain the same as the population declines the fishery would incidentally capture and kill 0.00017% of the population in 40 years, which we consider to be the most conservative reasonable scenario as 1) capture rates are likely to decline if the population declines by 36% over this time period as predicted from a 1.1% per year decline and 2) not all captures are expected to be from the East Indian-West Pacific. Assuming the population declines include the historic impact of the ASLL, removing the fishery would result in population declines of between 1.10986 to 1.10989% with the lower rate of decline representative of the population growth rate over 40 years. These values are well within the 95% CI for our estimate of the rate of the population decline (-1.6% to -0.6%).

A power analysis indicates more than 1 million years of data would be required to detect this difference and therefore the difference is not statistically detectable given our predictive capabilities. Because East Indian-West Pacific green sea turtle abundance will not be substantially impacted by the proposed action, we are reasonably certain it will not cause material changes having biological consequences to the DPS' numbers, reproduction, or distribution. In accordance with Section 1.3.1 (Jeopardy Analyses) above, NMFS does not anticipate the ASLL fishery will reduce appreciably the likelihood of the survival or recovery of East Indian-West Pacific green sea turtles in the wild by reducing their reproduction, numbers, or distribution.

5.6 Leatherback Sea Turtle

As discussed in the Status of Listed Resources section of this biological opinion, leatherback sea turtles are globally listed as endangered. The species is composed of seven populations, and the proposed action adversely affects the West Pacific population.

As described in the Exposure Analysis, hooking and entanglement are the most significant stressors that leatherback sea turtles are exposed to directly from the proposed action. On average, we expect a mean of 10 leatherback sea turtles to be captured each year in the ASLL fishery, although the number captured in any given year may rarely exceed 30 (Table 34). Assuming the patterns found in the observer data from 2010 to 2019 are representative, the majority of leatherback turtles would be hooked rather than entangled (100% of observed leatherbacks have been hooked).

When at-vessel mortality and post-release mortality are combined, the effective mortality rate for leatherback turtles captured in this fishery is 65% (95% CI = 38 to 85%). At this mortality rate, if 10 leatherbacks are caught in any given year, we would expect about 7 of them to die as a result of being captured. If 30 leatherback turtles are captured, we would expect about 20 to die as a result of being captured. Separating juveniles from adults, no adults were dead at-vessel while 89% (95% CI = 53 to 98%) of juveniles were dead at vessel; this assessment assumes these percentages are applicable to future captures.

We add the ASLL effects to an Environmental Baseline of interactions and mortalities from other fisheries, including international and United States fisheries in the Action Area. The fisheries that occur in the Action Area and their effects are reasonably likely to continue and may increase over time due to the effects of increased human population, and increased human consumption of fish products. Between 2013 and 2020, an estimated 40 to 244 leatherbacks were killed annually in WCPO longline fisheries, which is inclusive of the U.S. fisheries. These data includes mortalities outside of the action area, but those mortalities impact the condition of the species within the Action Area.

Climate change may be affecting the species already but will likely increase in the future. Global average sea levels are expected to continue to rise by at least several inches in the next 15 years, and by one to four feet by 2100 (Wuebbles et al. 2017). This could affect migrations and feeding patterns by changing ocean circulation. Leatherback sea turtles were predicted to gain core habitat area by Hazen et al. (2012). Such range shifts could affect foraging success and sea turtle reproductive periodicity (Kashner et al. 2011; Pike 2014). Increased sand temperatures can also cause decreased egg survival and increase the proportion of female hatchlings, skewing sex ratios and affecting the reproductive capacity of the populations.

As described in the Status of Listed Resources, the West Pacific leatherback population experienced a precipitous declines with populations declining by 83% from the 1980's to 2011 (Tiwari et al. 2013). Based on data up to 2017, Martin et al. (2020a) estimated that the adult female portion of the population would continue to decline at a mean of 6% per year (95% CI: -23.8% to 12.2%) and was comprised of 1,053 adult females in 2017. Lontoh et al. (In Prep) indicate that nest numbers were relatively stable from 2018 to 2021 and we assume the population abundance in 2022 was equivalent to 2017.

The mean number of leatherback sea turtles from the West Pacific population that are likely to be captured in the ASLL fishery in any given year is 10 (95th percentile: 30) with 7 (95th percentile: 20) mortalities assuming mean mortality rates. Based on the maximum 5-year running average of 9.6 leatherbacks captured in the ASLL fishery annually, we expect 6.2 would die from their capture. Based on the maximum 5-year running sum of 48 leatherback turtles that would be captured in the ASLL fishery, we expect 96 leatherback turtles would be captured over 10 years. Out of 96 leatherback turtles captured in ten years, we expect 62 of those to die. Over a 20-year time horizon, if captures remain consistent with historical numbers, we would anticipate a cumulative total of 192 captures, of those 125 would die as a result of their interactions. We anticipate that approximately 18% of leatherback sea turtles interacting with the fishery are adults and that 23% of those will die as a result of their interaction. We estimate that an average of 0.4 adults may be killed by the fishery in a year. Applying the sex ratio of 0.73, these adults equate to 0.3 adult females. Over 10 years, we anticipate that the fishery will interact with 17 adult leatherback turtles resulting in the mortality of 4 adults, 3 of which would be females.

The primary nesting beaches for the West Pacific leatherback population are Jamursba Medi and Wermon beaches in Indonesia, representing 50 to 75% of the population's nesting (NMFS and FWS 2020a). Conservation efforts at these beaches have increased nest and nesting female protection and hatchling production (Pakiding et al. 2020), and it is possible these conservation efforts have alleviated the precipitous decline at these locations. We recognize that it is imperative that these efforts continue as well as new efforts initiated at other nesting beaches to fully understand the status of this population and protect and enhance its productivity. For the purpose of this analysis, we acknowledge the uncertainty regarding the current status of this population as detailed in the Status of Listed Resources section and noted by Martin et al. (2020a) given the 95% credible interval for their estimate of the population's growth rate. We present information on the impact of the ASLL if the 6% per year decline continues from 2022, which we consider to be the most conservative reasonable scenario.

Based on NMFS' PVA model (Martin et al. 2020a, Siders et al. 2023), leatherback sea turtles in the West Pacific population are declining at about 6% per year (95% CI: -23.8% to 12.2%), and the population as indicated by the index beaches is at risk of falling to less than half of its 2017 abundance in 7.3 years (95% CI, 1 to 19 years) from 2021, or by about 2028, range 2022 to 2040. Based on Martin et al. (2020a), we estimate that there were approximately 1,054 adult females in the West Pacific population in 2017 or 1,442 adults based on a 73% female biased sex ratio. We assume this abundance is representative of 2022 abundances based on Lontoh et al. (In Prep) and further assume that the mean decline of 6% per year will occur after 2022.

At our estimated 2022 abundance, the capturing and killing of 0.4 adult West Pacific leatherbacks each year constitutes the death of 0.03% of the adult population. If captures in the ASLL remain the same as the population declines, the fishery would incidentally capture and kill 0.09% of the adult population in 20 years, which we consider the most conservative reasonable scenario as capture rates are likely to decline if the population declines by 70% over this time period as predicted from a 6% per year decline. The population declines include the historic impact of the ASLL. Removing the fishery would result in population declines in the adult portion of the population between 5.95% and 5.97% with the lower rate of decline representative of the most conservative reasonable scenario and fishery captures remaining the same as the population declines.

These anticipated population declines without the impact of the fishery are well within our uncertainty regarding the current trend for this population (95% CI: -23.8% to 12.2%) and a power analysis indicates that over 500,000 years of data would be required to detect a statistically significant difference between these numbers. Therefore, we do not have the ability to detect this difference over a 20-year timeframe. Furthermore, a population declining at 5.95% per year would reach 12.5% of the 2022 population abundance about 0.13 years (or about 1.6 months; 14.66 yr. compared to 14.79 yr.) later than a population declining at 6% per year. Over shorter time periods, the impact of the ASLL has a lower impact on population growth rate because keeping ASLL captures the same as the population declines results in fishery captures having a greater proportional impact as the population continues to decline. A population declining at a rate of 5.97% per year will reach 50% of 2017 abundance levels 0.06 years (less than one month; 11.55 yr. compared to 11.61 yr.) later than a population declining at a rate of 6% per year.

In summary, the West Pacific leatherback population declined by 78.3% from the 1980's to 2011 (Tapilatu et al. 2013) due to a combination of factors that include adult female harvest, low hatchling production due to harvest, predation, inundation and beach erosion (Tapilatu and Tiwari 2007, Tapilatu et al. 2013, Tapilatu 2014), and fishery bycatch (Lewison et al. 2014). The primary index beaches of Jamursba Medi and Wermon have been the focus of intense conservation efforts since the early 2000's and nest protection and relocation efforts have increased hatchling production from 21,966 per year between 2005 to 2013 to a minimum of 32,000 to 50,000 per year from 2017 to 2019. These efforts have likely slowed the decline of this species and potentially at least stabilized trends, although continuation of this potential stabilization will rely on continued efforts at these index beaches and the identification and protection of additional nesting beaches. The recent potential improvement in nesting beach trends have occurred in conjunction with the continued operation of the ASLL capturing a mean of 10 West Pacific leatherbacks per year, suggesting the fishery is not impeding the potential for this population to recover. At 2017/2022 abundance levels, the ASLL removed 0.03% of the adult population, which we described as not having an appreciable impact on population growth rates. Thus, we do not anticipate that the trajectory of the West Pacific population will be appreciably changed by interactions with the ASLL. Because West Pacific leatherback sea turtle abundance will not be appreciably impacted by the proposed action, we are reasonably certain it will not cause material changes having biological consequences to the population's numbers, reproduction, or distribution.

We roll the effects on the West Pacific leatherback population up to the species level, in accordance with Section 1.2.2 (Jeopardy Analyses) above. When we add the effects of the action and cumulative effects to the environmental baseline and in light of the status of the species, we find the proposed action will not reduce appreciably the likelihood of both the survival and recovery of the leatherback sea turtles species in the wild by reducing their reproduction, numbers, or distribution. Our task in a section 7 consultation is to consider whether the nature and magnitude of the proposed action's effects, when considered together with the species status and all other threats acting on it, are consequential enough to appreciably reduce the species' likelihood of survival and recovery. Our analysis suggests that the proposed action's effects, which include interacting with a mean of 10 leatherback sea turtles and killing 7 leatherback sea turtles in any given year, would have inconsequential impacts on the species' overall reproduction, numbers and distribution in the wild. That is, our no-jeopardy determination merely concludes that the action's impacts, when considered together with the leatherback's baseline status and all other threats acting upon the species, is not the cause of some new peril or material change that exacerbates the species' decline.

Although we conclude the ASLL fishery does not jeopardize leatherback sea turtles, we identify measures in Sections 7.3 and 7.4 necessary or appropriate to minimize the impacts from takings in the fishery.

5.7 Olive Ridley Sea Turtle

As discussed in the Status of Listed Resources section, the threatened and endangered populations of the Eastern Pacific and the threatened populations from the West Pacific occur in the Action Area.

Where densities are high enough, nesting takes place in synchronized aggregations known as arribadas. The largest known arribadas in the eastern Pacific are off the coast of Costa Rica (~475,000 - 650,000 females estimated nesting annually) and in southern Mexico (~1,000,000 nesting females per year at La Escobilla, in Oaxaca, Mexico [Abreu-Grobois and Plotkin 2008; Valverde et al. 2012; NMFS and FWS 2014]). The minimum size of the endangered Mexico population is 1,000,000 individuals.

At the nesting site in Ostional, Costa Rica, the average arribada size in the main nesting beach increased from 75,000 turtles in 1980 to 125,000 turtles in 2003 (Chaves et al. 2005 as cited in NMFS and FWS 2014). The minimum size of the threatened population in the East Pacific is 1,000,000 individuals. Overall, both the endangered Mexico population and the threatened populations in the East Pacific are at least stable with the larger nesting sites showing either stable or increasing trends.

In the western Pacific, the largest remaining breeding aggregations are in Northern Australia where it's estimated that at least 5,000 females nest annually (Commonwealth of Australia 2017). Considering remigration intervals, we estimate this aggregation at 10,000 adult females. Assuming a 50% sex ratio, this would equate to 20,000 adults. When added to a well-studied nesting aggregation in Alas Purwo National Park, East Java, Indonesia, which is currently increasing (Kurniawan and Gitayana 2020), suggests a minimum adult abundance of 24,400 for Western Pacific olive ridley sea turtles. Based on this we estimate a total abundance size of 316,883 individuals greater than one year old. Trends of the Western Pacific populations are uncertain although they appear to be stable or increasing on some beaches, we assume stable trends in our analysis.

Reviewing threats from the Status of Listed Resources, major anthropogenic threats to both threatened and endangered populations include impacts to nesting beaches resulting from development, direct harvest of eggs and adults, and fishing bycatch (NMFS and FWS 2014). Fisheries operating near arribadas can take tens of thousands of adults as they congregate for breeding (NMFS and FWS 2014).

As described in the Environmental Baseline, effects from fisheries bycatch are the most significant threat for the species. There were 762 olive ridley sea turtles reported captured by 16 countries in the western and central Pacific Ocean from 1989 to 2015. Of these, 206 were a result of United States fishery interactions. When extrapolated from 5% observer coverage the estimate for all fishery interactions results in 15,240 olive ridley sea turtles caught in the western and central Pacific Ocean from 1989-2015. These activities are reasonably likely to continue, and may increase over time due to the effects of increased human population, increased human consumption of fish products, and increased effort by fleets to catch fish.

No significant climate change-related impacts to olive ridley turtle populations have been observed to date. However, climate change-related impacts will likely influence biological trajectories in the future on a century scale (Paremsan and Yohe 2003). However, olive ridleys are migratory and are expected to adapt to changing conditions, which may minimize the impacts of climate change (Plotkin 2010 in NMFS and FWS 2014).

As discussed in the Exposure section, hooking is the most significant stressor that olive ridley sea turtles are exposed to directly from the action. The first olive ridley was not observed being captured until 2011. Since 2011 there have been a total of 16 observed interactions that have resulted in a hooking in which five olive ridley sea turtles were released alive (Table 42). Based on genetic samples from olive ridley sea turtles interacting with the ASLL fishery, NMFS estimates that approximately 40% of individuals are from the East Pacific (representing both the endangered and threatened populations) and 60% are from the Western Pacific threatened populations (NMFS 2015a).

Eastern Pacific Populations. As noted in the Status of Listed Resources section, the endangered Mexico population of olive ridley sea turtles is comprised of more than 1,000,000 nesting females (Abreu-Grobois and Plotkin 2008; Valverde et al. 2012; NMFS and FWS 2014). The minimum size of the threatened population in the East Pacific similarly numbers at more than

1,000,000 nesting females on beaches from Costa Rica and Nicaragua (Chaves et al. 2005 as cited in NMFS and FWS 2014; Valverde et al. 2012; NMFS and FWS 2014).

We anticipate that over 10 years, up to 49 olive ridley sea turtles from East Pacific beaches will be captured and of those 36 will be killed. Over 40-years, assuming captures remain consistent with historical capture rates, up to 195 olive ridley sea turtles from East Pacific beaches will be captured, and of those 142 will be killed. Even when we considered the potential effects of these mortalities on adult abundance, which would overestimate their effect, these mortalities are not likely to appreciably affect the abundance of adult females on these rookeries in any given year or accumulated over 40 years.

Western Pacific Populations: The action results in the capture of an average of seven Western Pacific olive ridley sea turtles with the mortality of five of those. The maximum 5-year running average is 7.3 turtles with five mortalities. At the estimated abundance level (316,883 individuals), incidentally capturing an average of 7.3. Western Pacific olive ridley sea turtles per year with five mortalities constitutes death of 0.0016% of the abundance per year. As this abundance is likely stable with some beaches experiencing increasing trends with the historic effect of the ASLL, we do not expect the continued operation of the fishery to change the populations' overall trajectory.

Therefore, we are reasonably certain the abundance trend will continue to be at least stable with the continued authorization of the ASLL fishery as currently managed. Because Western Pacific olive ridley sea turtle abundances are not declining 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 summary, we have concluded that the ASLL fishery is not likely to cause appreciable change in either the endangered or threatened populations of olive ridley sea turtle. Therefore, we do not expect the action to change the number of populations that comprise the species, the spatial distribution of those populations, or their expected patterns of growth and decline over time. As a result, in accordance with Section 1.3.1 (Jeopardy Analyses) above, NMFS does not anticipate the ASLL fishery will reduce appreciably the likelihood of the survival or recovery of either the endangered or threatened populations of olive ridley sea turtles' in the wild by reducing their reproduction, numbers, or distribution.

5.8 Hawksbill Sea Turtle

As described in the Status of Listed Resources, the hawksbill sea turtle is listed as endangered and population trends are poorly understood. Based on available data, we estimate between 5,400 and 6,160 nesters occur in Oceania (NMFS and FWS 2013a). The nesting trend at the largest known nesting aggregation in Oceania, the Milman Islands, Great Barrier Reef, Australia is declining at a rate of 4.9% per year (Limpus 2009, Mazaris et al. 2017) and we consider this the best available information on trends for Oceania hawksbill sea turtles.

We estimate the total population of Oceania hawksbill sea turtles greater than one year old in Oceania in 2013 at 2,592,331 sea turtles. Based on the declining trend of 4.9% per year, we estimate the 2022 population size to be 1,667,887.

As described in the Status of Listed Resources, hawksbill sea turtles are exposed to a variety of anthropogenic threats both at their nesting beaches and in the open ocean. A main threat to hawksbill sea turtles globally has been the direct exploitation and harvest of turtles for their carapace ('tortoiseshell') (Frazier 2003; Pita and Broderick 2005; Kinch 2007; Mortimer and Donnelly 2008; Hamilton et al. 2015; Miller et al. 2019). Additional anthropogenic threats include harvest of eggs as well as juveniles and adults for meat (Allen 2007; Limpus and Miller 2008; NMFS and FWS 2013a) and predation by non-native predators (pigs, ungulates, rats, feral dogs and cats; NMFS and FWS 2013a, 2018). Additional threats include loss of nesting beach habitat and bycatch in artisanal and industrial fisheries.

As described in the Environmental Baseline, past and present fisheries interactions may have been, and may continue to be, a threat to hawksbill sea turtles within the Action Area. Bycatch of hawksbill sea turtles occurs in many fisheries throughout the geographic oceanic range of this species. While mitigation and minimization measures have reduced fisheries sea turtle bycatch in United States fisheries in recent years, large numbers of hawksbill sea turtles may still be captured in international and United States commercial fisheries that target other species. These activities are reasonably likely to continue, and may increase over time due to the effects of increased human population and increased human consumption of fish products.

In addition, warming sea surface temperatures may lead to potential fitness consequences in sea turtles resulting from altered seasonality and duration of nesting (Pike et al. 2006). Sea turtles may also expand their range as temperature-dependent distribution limits change (McMahon and Hays 2006; Poloczanska et al. 2009a). Further, sea turtles have temperature-dependent sex determination, and many populations produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Newson et al. 2009; Patricio et al. 2017; Jensen et al. 2018).

Other climatic aspects, such as extreme weather events, precipitation, ocean acidification and sea level rise also have potential to affect sea turtle populations. Changes in global climatic patterns will likely have profound effects on the coastlines of every continent, thus directly affecting sea turtle nesting habitat (Wilkinson and Souter 2008). Lastly, studies examining the spatio-temporal coincidence of sea turtle nesting with hurricanes, cyclones and storms suggest that cyclical loss of nesting beaches, decreased hatching success and hatchling emergence success could occur with greater frequency in the future due to global climate change (Hawkes et al. 2009). Environmental changes associated with climate change are occurring within the Action Area and are expected to continue into the future. Marine populations that are already at risk due to other threats are particularly vulnerable to the direct and indirect effects of climate change.

As described in the *Exposure Analysis*, there had been no interactions with hawksbill sea turtles until 2016. Since then, there have been two more interactions recorded in 2018. In total, three hawksbill sea turtles have been captured in the ASLL fishery and all three individuals were juveniles and died from the interaction.

The action results in the capture and mortality of an average of six hawksbill sea turtle per year. As the maximum 5-year running average was smaller than the mean estimate of six, we use this value in place of the maximum 5-yr. running average. At the 2022 projected abundance levels (1,667,887 individuals), incidentally capturing and killing an average of six hawksbill sea turtles per year constitutes death of 0.00096% of the population. If captures remain the same as the population declines, the fishery would incidentally capture and kill 0.0026% of the population in 40 years which we consider to be the most conservative reasonable scenario as capture rates are likely to decline if the population declines by 85% over this time period as predicted from a 4.9% per year decline. If we added these mortalities back to the population, from 2022 to 2023 the population growth rate would increase to a 4.8997% per year decline; from 2022 to 2062 the population growth rate would increase to a 4.8989% per year decline. We do not have a measurement of uncertainty associated with the population growth rate of -4.9% to be able to conduct a power analysis, but given the similarity between the estimated population growth rates with and without the fishery mortalities and the fact that similar differences for Central West and Central South Pacific green sea turtles were not statistically detectable, the change in the trajectory of the population without the fishery would not be detectable given our predictive capabilities.

Because Oceania hawksbill sea turtle abundance will not be substantially impacted by the proposed action, we are reasonably certain it will not cause material changes having biological consequences to the 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 Oceania hawksbill sea turtles in the wild by reducing their reproduction, numbers, or distribution.

5.9 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. Sublethal effects could include 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. We included these effects in our analysis to the best of our ability using the best information available.

As discussed in the Status of Listed Resources 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). Oceanic whitetip sharks have low fecundities for sharks (between 0 and 15 pups) and a biennial reproductive cycle.

Oceanic whitetip sharks are listed as threatened throughout their range and are classified as overfished and have experienced substantial declines in abundance, total biomass, spawning biomass, and recruitment levels (Rice and Harley 2012; Futerman 2018). The potential impacts from climate change on oceanic whitetip shark habitat are highly uncertain, but given their broad distribution in various habitat types, these species may 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. Without the mortalities from the ASLL, the population trend estimates would range from -0.08% to 0.407% per year for 2022 to 2023 and from -0.082% to 0.403% per year from 2022 to 2062. 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.

5.10 Indo-West Pacific Scalloped Hammerhead Shark

Three identified scalloped hammerhead DPSs occur in the Pacific, two of which are listed under the ESA that are potentially exposed to fishing activities in the Action Area. However, only the

Indo-West Pacific scalloped hammerhead shark is expected to be adversely affected by the fishery (see Appendix A for the discussion on the eastern Pacific scalloped hammerhead shark) and is listed as threatened. As discussed in the Status of Listed Resources, the global population is estimated to be 280,000 adults based on an analysis of effective population size but there are no estimates for the Indo-West Pacific DPS. Overutilization by industrial/commercial fisheries, artisanal fisheries, and illegal fishing are the most serious threats to the persistence of the Indo-West Pacific scalloped hammerhead shark. Indo-West Pacific scalloped hammerhead sharks are both targeted and incidentally captured as bycatch in many fisheries. Contributing to the Indo-West Pacific scalloped hammerhead shark's biological vulnerability is the fact that these sharks are obligate ram ventilators and suffer very high at-vessel fishing mortality in fisheries (Morgan and Burgess 2007; Macbeth et al. 2009; Miller et al. 2014; Dapp et al. 2016).

As described in the Environmental Baseline, effects from international fisheries have resulted in interactions with the Indo-West Pacific scalloped hammerhead sharks in the Action Area. Between 2013 and 2020, the international WCPO longline fisheries are estimated to have captured 4,561 Indo-West Pacific scalloped hammerhead sharks, with 2,004 mortalities. The WCPO purse seine fisheries are estimated to have captured 217 Indo-West Pacific scalloped hammerhead sharks with no observed mortalities. These activities are reasonably likely to continue, and may increase over time due to the effects of increased human population, increased human consumption of fish products, and the international trade of shark fins.

Because the Indo-West Pacific scalloped hammerhead shark range is comprised of open ocean environments occurring over broad geographic ranges, large-scale impacts such as global climate change that affect ocean temperatures, currents, and potentially food chain (trophic) dynamics, climate change may pose a risk to this DPS. Chin et al. (2010) conducted an integrated risk assessment which determined scalloped hammerhead sharks were ranked as having a low overall vulnerability to climate change, with low vulnerability to each of the assessed climate change factors.

As discussed in the *Exposure Analysis* section, estimates of the number of Indo-West Pacific scalloped hammerhead sharks that are likely to be captured in the ASLL fishery each year are presented in Table 55. On average, 9 (95th Percentile: 28) of these sharks are likely to be captured each year in the ASLL fishery. We expect over 5 years, a maximum cumulative sum of 45 Indo West scalloped hammerhead sharks will be caught in the fishery with a 5-year maximum running average of 8.9 sharks.

As discussed in the Response Analysis sections, we estimate an at-vessel mortality rate of 26.7% and a post-interaction mortality rate of 19.5% for a total mortality rate of 41.0%. Based on the maximum 5-yr. running average we would expect an average of 3.6 mortalities each year or 18 mortalities over 5 years.

As discussed previously in the Status of Listed Resources, Pacoureau et al. (2020) indicates a 67% decline from 1970 to 2018 equating to a 2.31% decline per year for the global population and we estimated a 2.28% decline for the Indo-West Pacific scalloped hammerhead shark based on ocean basin trends reported in Pacoureau et al. (2020).

As discussed in the Risk section, as we do not have abundance estimates for this DPS. Thus, we assessed the impact of removing 3.7 mortalities per year from various starting population sizes to assess if the ASLL will have a measurable impact on the DPS' survival and recovery. We found that the rate of decline (2.28% per year) did not change by removing the potential impacts of the ASLL. Therefore, we are reasonably certain the action will not change the number of populations that comprise the DPS or the spatial distribution of those populations. While the proposed action will remove animals from one of the populations, it will not appreciably reduce its expected patterns of growth and decline over time. Based on the available evidence, we do not expect the loss of 3.7 individuals per year to affect the viability of the population that these individuals represent considering the number of interactions expected to occur in this fishery, the corresponding mortality, the abundance, and ability to persist. We do not expect the action to change the number of populations that comprise the DPS, the spatial distribution of those populations, or their expected patterns of growth and decline over time. As a result, NMFS does not anticipate the ASLL fishery will reduce appreciably the likelihood of the survival or recovery of Indo-West Pacific scalloped hammerhead sharks in the wild by reducing their reproduction, numbers, or distribution.

5.11 Giant Manta Ray

As described in the Status of Listed Resources, 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 7); however, estimates of the global population size as well as subpopulation 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 (pers. comm. Joshua Stewart, Manta Trust 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 status of the giant manta ray appears 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 effected by several stressor within the Action Area, including climate change, fisheries, vessel strikes, and marine debris; however both direct harvest and bycatch in fisheries is the dominant stressor in the baseline that affects the species. 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 62).

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. We believe the proportion of unobserved interactions is very small and the short-term (survival) and long-term (fitness) effects are minimal. For this biological opinion, we have done everything possible to account for unobserved interactions and include their effects in our analyses (e.g. rounding up to the nearest whole number of mortalities, assuming all serious injuries result in mortality, etc.).

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 50). 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 64). For the other four subpopulations potentially impacted by the fishery, all continue to increase at all initial population abundances (Table 64). 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 64), 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 64). The remaining subpopulations had increasing population trajectories at all initial abundances and values of r_{max} (Table 64).

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 following species under NMFS' jurisdiction:

> Endangered leatherback sea turtle, threatened green sea turtles in the East Pacific, East Indian-West Pacific, Southwest Pacific, endangered green sea turtles in the Central West Pacific and Central South Pacific, threatened olive ridley sea turtles and olive ridley sea turtles from the endangered Mexico breeding population, endangered hawksbill sea turtles, threatened oceanic whitetip sharks, endangered Indo-West Pacific scalloped hammerhead sharks, and threatened giant manta ray.

7 INCIDENTAL TAKE STATEMENT

Section 9 of the ESA and protective regulations pursuant to Section 4(d) of the ESA prohibit the take of endangered and threatened species without a special exemption. "Incidental take" is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity (50 CFR 402.02). Under the terms of section 7(b)(4) and Section 7(o)(2), taking that is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the ESA provided that such taking is in compliance with the reasonable and prudent measures and terms and conditions of the ITS. The proposed action results in the incidental take of endangered leatherback sea turtles, East Pacific green sea turtles, East Indian-West Pacific green sea turtles, Central West Pacific green sea turtles, Southwest Pacific green sea turtles, Central South Pacific green sea turtles, endangered Mexico breeding population of olive ridley sea turtles, endangered hawksbill sea turtles, and threatened olive ridley sea turtles, threatened oceanic whitetip sharks, endangered Indo-West Pacific scalloped hammerhead sharks, and threatened giant manta rays.

We have not yet promulgated an ESA section 4(d) rule prohibiting take of oceanic white tip sharks, Indo-West Pacific scalloped hammerhead sharks, or giant manta ray, so an exemption from the take prohibitions of Section 9 of the ESA is neither necessary nor appropriate for these species. Consistent with the decision in *Center for Biological Diversity v. Salazar*, 695 F.3d 893 (9th Cir. 2012), we have included an incidental take statement to serve as a check on the no-jeopardy conclusion by providing a reinitiation trigger if the level of take analyzed in the biological opinion is exceeded. In addition, 50 CFR 402.14(i)(3) provides that in order to monitor the impacts of incidental take, "the Federal agency or any applicant must report the progress of the action and its impact on the species to the Service as specified in the ITS."

The measures described below are nondiscretionary, and must be undertaken by NMFS for the exemption in section 7(o)(2) to apply. NMFS has a continuing duty to regulate the activity covered by this incidental take statement. If NMFS fails to assume and implement the terms and conditions, the protective coverage of section 7(o)(2) may lapse. In order to monitor the impact of incidental take, NMFS must monitor the progress of the action and its impact on the species as specified in the incidental take statement (50 CFR 402.14(I)(3)).

7.1 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 below in Table 68. The number of takes occurring annually is highly variable due to fluctuations in fishery target-species distribution, ESA-listed species distribution and abundance, fishing effort, sea surface temperatures, ocean currents, and other factors that are difficult to predict. As a result, using the estimated mean annual take levels is not reasonable or practical. Warden et al. (2015) state "when the population is large compared to the incidental mortality, frequent (e.g., annual) monitoring is not likely to produce results that are substantially different from the previous assessment. Less frequent but more comprehensive assessments, which explicitly address uncertainty, may provide more reliable information." For these reasons, and based on our experience monitoring fisheries, we believe that the maximum 5-year running sum is the most appropriate metric for meaningful tracking of take with respect to the ITS. Year to year variation in capture numbers is expected, and managing the incidental take by the 5-year running sum accounts for this annual variation, allowing for years with higher than average captures and years with lower than average captures. Exceeding the maximum 5-year running sum (Table 68) within any five consecutive years is a reinitiation trigger. This does not imply we will wait five years to assess take. Observed interactions are documented in real time and uploaded to our internal database. The database is continually monitored by NMFS SFD and PRD. Therefore, we will know of any exceedance of the numbers regardless of when it happens during any consecutive 5-year period. As an example, if 46 Indo-West Pacific scalloped hammerhead sharks were to be caught within a 2 or 3-year period, reinitiation would be triggered when the exceedance occurs (i.e. 2 or 3 years), and not at the 5-year mark.

Table 68. Anticipated take by 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.

Species	Maximum 5-Year Running Sum
Green sea turtle (all species)	96
Leatherback sea turtle	48
Olive ridley sea turtle (all species)	61
Hawksbill sea turtle	30*
Oceanic whitetip shark	3,520
Indo-West Pacific scalloped hammerhead sharks	45
Giant manta ray	57

*Note that the maximum 5-yr. running average for hawksbill sea turtles is 2.4. The mean estimated by McCracken (2019a) for 2016-2017 was 6. Given that the maximum 5-yr. running average is confounded with multiple years of zero captures, we use the 5-year cumulative sum of 30 for the ITS.

7.2 Reasonable and Prudent Measures

Reasonable and prudent measures are those actions the Director believes necessary or appropriate to minimize the impacts, i.e., 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 and endangered species and to monitor the level and nature of any incidental takes. These measures are non-discretionary—they must be undertaken by NMFS SFD for the exemption in ESA section 7(o)(2) to apply.

1. NMFS SFD shall require that ESA-listed species incidentally caught alive be released from fishing gear in a manner that minimizes injury and the likelihood of further gear entanglement or entrapment to increase their post-release survivorship.
2. NMFS shall ensure that the ASLL fishery has a monitoring and reporting program sufficient to confirm that extent of take is not exceeded, and that the terms and conditions in this incidental take statement are effective in minimizing incidental take.

7.3 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 9.3 above. These terms and conditions are non-discretionary, and if NMFS SFD fails to adhere to these terms and conditions, or fails to implement measures requiring the fishery to comply with these terms and conditions, the protective coverage of section 7(o)(2) may lapse.

1. The following terms and conditions implement Reasonable and Prudent Measure No. 1:
    a. NMFS SFD shall implement measures to minimize the amount of trailing gear left on ESA-listed species to the maximum extent practicable to reduce the amount of injury and harm, the likelihood of further gear entanglement or entrapment, and improve the post-release mortality of ESA-listed species. This may include using new technologies once proven, such as fighting lines (i.e., additional gear that reduces tension on the branchline), line cutting devices, and other gear modifications.
    b. To reduce post-release mortality, within two years, NMFS SFD shall require species handling training for crewmembers and at a minimum have one trained person on deck who directs and oversees activities of the vessel when retrieving fishing gear. Training shall include best practices identified in 1.a above.
2. The following terms and conditions implement Reasonable and Prudent Measure No. 2:
    a. NMFS SFD shall maintain observer coverage at levels reliable for estimating protected species interaction rates onboard ASLL vessels. This may include electronic monitoring.
        i. NMFS SFD shall collect standardized information regarding the incidental capture, injury, and mortality of ESA-listed marine species for each interaction by species, gear, and set information, as well as the presence or absence of tags on these species.
        ii. NMFS SFD shall improve length estimates of leatherback sea turtles that are not boarded. These lengths will be recorded in a specified field rather than only noted in the comment field.

        iii. To the maximum extent practicable, observers shall identify the hooking location for every interaction, and estimate the length of any trailing gear left on ESA-listed species at release when those species cannot be boarded. These data are intended to allow NMFS to improve estimates of harm, injury, and mortalities within this fishery.
        iv. The observers shall document the method or technology used to release all ESA-listed species.
    b. NMFS SFD shall collaborate with the NMFS PRD to develop annual data products to be provided to NMFS PRD detailing levels of captures, updates to interaction mortality rates and locations of fishing effort and subsequent captures. These reports will be provided to NMFS PRD once all data from the previous year have been finalized but not later than July of the following year. The report from the first year will include all data for calendar year 2023, as well as the period from the date this opinion was signed to the end of the year. Only the take that occurs post-signature is exempted by this opinion, but the calendar year data will be used to ensure incidental captures are within the expectations of our analysis.
    c. Every five years after signing of this opinion, NMFS SFD shall collaborate with the NMFS PRD to evaluate the data:
        i. Use the data from 2.a.i to explore ecosystem and area based management techniques with potential to reduce interactions with ESA-listed species in the future. Examples include time area closures, rotational zone management, dynamic closures, hotspot analyses, and move-on methods, etc.
        ii. Evaluate the effectiveness of methods designed to reduce post release mortality in ESA-listed species and make improvements or incorporate changes as warranted.

7.4 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 initiate research of 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 incidentally captured in the fisheries 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 are being captured (*M. alfredi* vs. *M. birostris*), and where these species are being captured in the fishery in order to avoid these areas by reducing potential interactions in the future.

3. NMFS should promote studies on ecology, habitat use, fecundity, genetics, and post interaction survivability of ESA-listed marine species.
4. NMFS should explore how climate change, including ocean warming, may affect habitat quality, prey abundance and distribution, and the physiological ecology (e.g., thermal tolerance) of the ESA-listed species considered herein.
5. NMFS should encourage RFMOs to require reporting of oceanic whitetip catches and discards, and for Parties to increase reporting of oceanic whitetip catch and disposition to improve data quality and quantify the impact of fishing on the species.
6. NMFS should enhance bilateral cooperation and engagement with key countries that have large international longline fleets to promote conservation and recovery of leatherback sea turtles, oceanic whitetip sharks, and other ESA-listed species considered herein.
7. NMFS should enhance capacity building among the international fishing community for increasing data collection and information sharing.
8. NMFS should develop an outreach and education campaign for the public to increase awareness of the status and importance of ESA-listed species, while incorporating cultural insights and perspectives from various regions/locations of the species' range.
9. NMFS should explore implementing an Electronic Monitoring program in the fishery to improve our understanding of interactions with ESA-listed species and reduce uncertainty when those interactions occur. At minimum, NMFS should consider a voluntary EM selection pool with a random strata selection that maintains fleet-wide observer coverage in accordance with term and condition 2a.
    a. Explore cost share and assistance programs for EM installations, explore feasibility, etc.
    b. Consider reviewing and updating an Annual Deployment Plan for the Region including an EM selection pool or fleet wide deployment.
    c. Coordinate with other NMFS Regions where EM technologies were successfully implemented for comparable fisheries and vessel class.

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

7.5 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.

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9 APPENDIX A: GENERAL EXPOSURE PROFILE

To assess potential exposure of threatened and endangered species in the Action Area, we began by parsing species by the general location of their exposure (coastal or pelagic), whether there were unique temporal characteristics to their potential exposure; for instance, would exposure likely occur only when a vessel was transiting to and from harbor (e.g., chambered nautilus and *Acropora spp*., *Euphyllia paradivisa* and *Isopora crateriformis*). Next we reviewed whether we had data (observations) on the species exposure or reasoned information that exposure could occur (potential) to one or more of the action's stressors: fishery interactions (e.g., vessel noise, vessel collision, hooking or entanglement in gear); vessel waste, discharge, and emissions. We ended up with four general exposure scenarios (Table 69): (1) observed pelagic exposure; (2) observed coastal exposure; (3) potential pelagic exposure; (4) potential coastal exposure. Each exposure profile that results for each species is unique, and may represent different combinations of stressors of a different magnitude or exposure to those stressors. Given the nature of vessel waste, discharge and emissions, these stressors have the potential to affect all exposure profiles. Even so, some patterns emerged, specifically when we focused on the primary stressors within each profile.

For example, even with the proposed LVPA longline vessel exemption seaward of 12 nm from Tutuila, Manua Islands, and Swains Island, (Figure 2) vessels are restricted from fishing in coastal waters. Therefore, the species listed in the potential coastal exposure would not be exposed to hooking or entanglement from fishing gear; however, they would likely be exposed to vessel transit and vessel noise. The empty observed coastal exposure profile reflects the number of documented interactions (vessel collisions) with coastal species. There were no accounts in the observer database of any observed vessel collisions with any species, much less the species within the coastal exposures.

Conversely, there are documented observations of interactions with pelagic species and fishing gear resulting in hooking and/or entanglement. These data shaped the observed pelagic exposure profile. Although species within this exposure profile may collide with a vessel, the primary stressor from the proposed action is hooking and entanglement. Lastly, the potential pelagic exposure is comprised of species that occur in the Action Area that could potentially interact with fishing gear or collide with a vessel. However, there are no documented observations of any fishery interactions between these species and the ASLL fishery.

Table 69. General location (coastal or pelagic waters) of likely exposure of species to proposed action's stressors. *These species fell within two exposure profiles.

Observed Pelagic Exposure	**Potential Pelagic Exposure**
Leatherback sea turtle	South Pacific Loggerhead sea turtle
Green sea turtle*	Sperm whale
Olive Ridley sea turtle	
Hawksbill sea turtle*	-
Giant manta ray*	-
Oceanic whitetip shark	-
Indo West-Pacific scalloped hammerhead	-
Observed Coastal Exposure	**Potential Coastal Exposure**
none	Hawksbill sea turtle*
-	Giant manta ray*
-	Green sea turtle*
-	Chambered Nautilus
-	*Acropora globiceps*
-	*Acropora jacquelineae*
-	*Acropora retusa*
-	*Acropora speciosa*
-	*Euphyllia paradivisa*
-	*Isopora crateriformis*

This subsection also summarizes the results of our evaluations of the probability of a response given exposure, which considers susceptibility. Here we briefly describe the species and stressors that are not likely to adversely affect listed species and our reasoning for this conclusion. Based on our evaluation, the following species are not likely to be adversely affected by the proposed action: South Pacific loggerhead sea turtle; sperm whale; chambered nautilus; *Acropora globiceps; A. jacquelineae; A. retusa; A. speciosa; Euphyllia paradivisa, and Isopora crateriformis.*

Potential Coastal Exposure

Because some species fall within two categories, a total of ten species are listed in the potential coastal exposure category. However, seven species are limited to coastal areas and they include chambered nautilus, *Acropora globiceps, A. jacquelineae, A. retusa, A. speciosa, Euphyllia paradivisa, and Isopora crateriformis.*

As mentioned above, fishing is restricted in the coastal areas. As such, these species are only exposed to stressors emanating from vessel transits (e.g., vessel collision, vessel noise, vessel waste, discharge and emissions). These stressors may result in immediate exposures, or exposures later in time (e.g., interaction with derelict gear or vessel waste). However, NMFS has determined that for all the species in the category of potential coastal exposures, effects from all or any stressors related to vessel transiting to be either highly unlikely and therefore discountable or insignificant. For detailed information, see the applicable stressor discussion below.

Observed Coastal Exposure

Fishing is restricted in coastal areas, as a result stressors emanating from vessel transits (e.g., vessel collision, vessel noise, vessel waste, discharge and emissions) are the only stressors expected in coastal waters. We know of no recorded observations of interactions between the proposed action and listed resources in coastal areas.

Potential Pelagic Exposure

Two species fall into the potential pelagic exposure category: South Pacific loggerhead sea turtle and the sperm whale. These species could be exposed to all or any of the stressors emanating from the proposed action: fishery interactions (e.g., vessel noise, vessel collision, or entanglement in gear); and vessel waste, discharge, or emissions. These stressors may result in immediate exposures, or exposures later in time (e.g., interaction with derelict gear or vessel waste).

None of the species listed above have been observed to be hooked or entangled in ASLL fishery gear over the last 25 years. Therefore, NMFS has determined that for all the species in the potential pelagic exposure profile effects from all or any stressors related to fishery interactions to be either highly unlikely and therefore discountable or insignificant. For more information, see the applicable stressor discussion below.

Observed Pelagic Exposure

Seven species fall into the observed pelagic exposure category: leatherback sea turtles, hawksbill sea turtles, green sea turtles (five DPS'), olive ridley sea turtles, Indo-West Pacific scalloped hammerhead sharks, oceanic white-tip sharks, and giant manta rays. These species could be exposed to all or any of the stressors emanating from the proposed action: fishery interactions (e.g., vessel noise, vessel collision, hooking or entanglement in gear); and vessel waste, discharge, or emissions. These stressors may result in immediate exposures or exposures later in time (e.g., interaction with derelict gear or vessel waste). As previously described, there are documented observations of interactions with fishing gear resulting in hooking and/or entanglement with the species listed in the observed pelagic exposure profile and are described in greater detail in Section 4 of this Opinion in the Effects of the Action section.

9.1 Stressors Not Likely to Adversely Affect Listed Resources

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.

Given the size of the ASLL fishery (the small number of vessels in the fishery), 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, and 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 would generally be ignored by animals that are temporarily exposed to sounds emanating from the vessels in this fishery. 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 would elicit behavioral reactions from ESA-listed species considered in this consultation. Thus, the effects will not rise to the level of harm or harassment, and we are reasonably certain the effects from this stressor will be insignificant on the ESA-listed resources in Table 3.

Collision with vessels

The proposed action would expose all ESA-listed marine species under NMFS' jurisdiction found in both the coastal and pelagic exposure categories to the risk of collision with vessels. However, given the small number of vessels participating in the fishery, the small number of anticipated vessel trips, the slow vessel speeds during fishing operations and vessel transiting, and the expectation that ESA-listed marine species would be widely scattered throughout the proposed Action Area, the probability of an incidental vessel strike is extremely unlikely to occur.

Jensen and Silber (2003) provided a review of ship strike records worldwide from 1975 to 2002. In total, Jensen and Silber (2003) documented a total of 292 records of confirmed or possible ship strikes to large whales. The sperm whale was one of the eleven confirmed species identified and found in the ASLL Action Area (Jensen and Silber 2003). According to Jensen and Silber (2003), fin whales are the most often reported species hit worldwide (75 records of strike), followed by humpback (44 records), and sperm whales (17 records). Of these 292 ship strike records, no strikes were specifically documented in the Action Area but included records nearby in Japan, Australia, and New Zealand (Jensen and Silber 2003).

Ship strikes were identified as a source of mortality for green sea turtles in Hawaii waters where there is a high density of turtles. Chaloupka et al. (2008) reported that 2.5% of green turtles found dead on Hawaiian beaches between 1982 and 2003 had been killed by boat strikes. Ship strikes have also been reported as a potentially important threat to sea turtle populations by researches in other parts of the world including the Canary Islands (Oros et al. 2005), Italy (Casale et al. 2010), and the Galapagos Islands (Parra et al. 2011; Denkinger et al. 2013). While we do have estimates of ship strikes around Hawaii (NMFS 2008a, 2008b, 2018a, 2019b), the ASLL vessels offload their catch in Pago Pago Harbor, a smaller port with less vessel activity compared to the port of Honolulu. Further, the density of sea turtles in the nearshore of American Samoa is less than that of Honolulu.

Kelly (2020) documented vessel collisions with sea turtles resulting in lethal and sub-lethal injuries. Sea turtles could potentially be struck by the transiting vessel during the proposed activities. NMFS (2008) estimated 37.5 vessel strikes of sea turtles per year from an estimated

577,872 trips per year from vessels of all sizes in Hawaii. More recently, we estimated as many as 200 green sea turtle strikes annually in Hawaii (Kelly 2020). If these turtle strikes are evenly distributed around the islands, the probability of a green sea turtle strike from any one vessel trip is extremely low (on average 0.035%, calculated by dividing the most recent strike estimate of 200 per year by the best estimate of all vessel transits of 577,872 per year). However, green sea turtle strikes are not evenly distributed throughout the islands. Increased vessel speed decreases the ability of sea turtles to recognize a moving vessel in time to dive and escape being hit, as well as the vessel operator's ability to recognize the turtle in time to avoid it. The *Action Area* is not in a location identified by Kelly (2020) as a hot spot for green sea turtle strikes, nor is it an area with significant overlap of high-density boating activity and sea turtle habitat. Therefore, the probability of a green sea turtle strike is likely less than the overall rate calculated above.

We estimate that the turtles killed by vessel strikes in the Action Area is less than the 250 green turtles struck around Hawaii. The vessels in the ASLL fishery range in size from 30 ft. to over 50 ft. (WCPFC 2011). These vessels travel at speeds less than 10 kt (NMFS 2018a). Since 2007, the number of vessels participating in the ASLL fishery has fluctuated. However, numbers have always been low with a high of only 29 vessels in 2007. Similarly, there is variability in the amount of fishing trips each year with a high of 377 trips in 2007. Additionally, the other sea turtle species have a lower rate of striking than green sea turtles. This is likely mostly due to their low abundance numbers and preference for deeper offshore waters (Kelly 2020). Lastly, no turtles are known to have been struck in the ASLL fishery. Because the probability of a vessel striking any other sea turtles is even lower than that of a green sea turtle, and because of the transit speeds in to port are slow, and given no interactions documented since 2006, NMFS is reasonably certain the probability of a ship strike will be discountable on the ESA-listed resources in Table 69.

Introduction of vessel wastes and discharges, derelict gear, and vessel emissions

The diffuse stressors associated with the longline fisheries: vessel waste discharge, derelict gear, and carbon emissions and greenhouse gasses, can affect both pelagic and coastal areas. ESA listed resources could be exposed to discharges, and run-off from vessels that contain chemicals such as fuel oils, gasoline, lubricants, hydraulic fluids and other toxicants. The amount of vessel waste discharge from the ASLL fishing fleet is difficult to determine with any accuracy and presumably quite small relatively to other sources of similar wastes.

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 expect the contribution to global greenhouse gases to be relatively inconsequential based on the low number of participants in the fishery.

Although leakage, wastes, derelict gear, and vessel emissions 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 small chance that ESA-listed resources would be exposed to measurable or detectable amounts of wastes, gear, or emissions from this fishery, we are reasonably certain the effects will not rise to the level of harm or harassment. Thus, we are reasonably certain the effects from this stressor will be insignificant on the ESA-listed resources in Table 69.

10 APPENDIX B: POWER ANALYSIS FOR DETECTING DIFFERENCES IN POPULATION GROWTH RATES WITH AND WITHOUT ASLL FISHERY MORTALITIES

As part of our risk analysis for species/DPSs that are in decline, we assumed that the current estimated population trends were inclusive of mortalities from the ASLL. We projected populations for 40 years (20 years for West Pacific leatherbacks, see Section 1.3.1, Jeopardy Analysis), each year adding back in the projected mortalities from the fishery. We then calculated the new population growth rate after removing fishery mortalities for 40 years (20 years for West Pacific leatherbacks). Here, we detail our power analysis approach to determining if the difference in population growth rates with and without fishery mortalities is statistically significant.

There are four key quantities in determining statistically significant differences between two mean population growth rates (Cohen 1988):

- Sample size (number of years needed to detect the difference)
- Effect size (difference between the means divided by the pooled standard deviation; Cohen 1988)
- Significance level or probability of making a Type I error (probability of determining the fishery is causing a statistically significant difference in a species rate of decline when it is not).
    - We use 0.05 for determining significance, in this case we are saying we will accept a 5% chance of making a Type I error, or we are 95% sure the fishery is not causing a statistically significant difference in a species rate of decline.
- Power (P) which equates to a (1-P) probability of making a Type II error (probability of determining that the fishery is not causing a statistically significant difference in a species rate of decline when it is).
    - We used P=80% which implies that we will accept a 20% (100%-80%) chance of determining that the fishery is not causing a statistically significant difference in a species rate of decline when it is. Similar to 0.05 being the common convention for determining statistical significance, 0.80 is the common convention for determining power (Button et al. 2013).

In a power analysis, you can determine any one of these quantities providing you know the other three.

Assumptions of the analysis:

- If we only had a 95% confidence or credible interval for a mean population growth rate, we assumed a normal distribution and that the standard deviation was the width of the confidence interval divided by 3.92. In a normal distribution, the 95% CI is 1.96 standard deviations away from the mean; therefore, there are 1.96*2 = 3.92 standard deviations across the width of the 95% CI.
- We assume the same variance for the population growth rates with and without the fishery.

In our analysis, we asked the question "how many years of data would be needed to detect a significant difference between population growth rates with and without 40 years of mortalities from the ASLL fishery (20 years for West Pacific leatherback sea turtles) with a significance level of 5% and with a power of 80%". We used the R package pwrrs (Bulus 2023) to calculate the minimum number of years (i.e. sample size) required to detect the difference between population growth rates with and without 40 years of fishery mortalities given the level of significance and power describe above. Given that we know removing fishery mortalities will increase population growth rates (or reduce a population's rate of decline), we use a one-tailed test to determine the sample size needed to detect that the population's rate of decline is significantly reduced by the removal of fishery mortalities.

Hypotheses:

- Null (H0): population growth rates are not different with and without ASLL fishery mortalities
- Alternate (HA): population growth rates are lower (declining faster) with the ASLL fishery mortalities compared to population growth rates with the fishery mortalities removed

We used the following function of the 'pwrrs' package in R for this test is (with MHI IFKW values used as an example, see Table 1):

> <pwrss.t.2means(mu1 = -0.0553, mu2 = -0.05331, sd1 = 0.0212, sd2 = 0.0212, kappa = 1, power = 0.80, alpha = 0.05, alternative = "less")>

- mu1 is the population growth rate with the fishery mortalities
- mu2 is the population growth rates with the fishery mortalities removed
- sd1 and sd2 are the standard deviations which we assume to be the same
- kappa is the ratio of the sample size needed for each group (i.e. mu1 and mu2), in this case the sample size is the same for both groups resulting in kappa = 1
- power is as we described above, the probability of making a Type II error and we use 0.80
- alpha is the significance level as describe above, the probability of making a Type I error and we use 0.05.
- alternative is the direction of the hypothesis test, in this case we are using a one-tailed test to detect that the population growth rate with the fishery mortalities is less than the population growth rate without the mortalities. For a two-tailed test where we didn't know the direction of the difference we would use alternative = "not equal".

Table 70 contains the input and output values for the species/DPSs for which we made the calculations. We found that at least 1,405 years of data would be needed to detect a significant difference between the population growth rates with and without ASLL mortalities for the species/DPSs considered. These numbers of years are well beyond the 40 years (20 years for West Pacific leatherbacks) over which we removed fishery mortalities. Furthermore, they are well beyond any timeframe we can reasonably predict the action's effects or the populations' abundances We therefore would not be able to reject the null hypothesis that population growth rates are the same with and without the fishery mortalities with 40 years of data.

Overall, for all species/DPSs considered, we do not have the predictive power to detect a difference in population growth rates with and without mortalities from the ASLL fishery.

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Table 70. Input and output values used in R to determine the number of years of data required to detect a difference in population growth rates with and without ASLL fishery mortalities.

Species/DPS	**Inputs used in the pwrss command in R**			**Number of years needed to detect a significant difference**
	mu1 (population growth rate with fishery mortalities)	**mu2 (population growth rate after removing fishery mortalities for 40 years [20 for WP leatherbacks])**	**Sd (standard deviation of population growth rate)**	
Central South Pacific Green Sea Turtle	-0.058	-0.05783	0.018	138,627
Central West Pacific Green Sea Turtle	-0.058	-0.057929	0.018	794,744
East Indian-West Pacific Green Sea Turtle	-0.0111	-0.0110986	0.028	4,946,045,787
Leatherback Sea Turtle (West Pacific Population)	-0.06	-0.05945	0.0896	328,164