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

Action Agency: National Marine Fisheries Service, Pacific Islands Region, International Fisheries Division

Federal Action: Authorization of the United States Western and Central Pacific Ocean Purse Seine Fishery

Consultation Conducted by: National Marine Fisheries Service, Pacific Islands Region, Protected Resources Division

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Approved By:

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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 (ITS) 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 ITS 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)).

The action agency for this consultation is NMFS' Pacific Islands Regional Office (PIRO) International Fisheries Division (IFD), which proposes to authorize the United States Western Central Pacific Ocean (United States WCPO) purse seine fishery, as currently managed under the existing regulatory framework. 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. This biological opinion has been prepared in accordance with

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

the requirements of Section 7 of the ESA, the implementing regulations (50 CFR 402), agency policy, and guidance and considers and is based on information contained in IFD's biological assessment (NMFS 2017a), NMFS and FWS recovery plans and status reviews for the species under consideration (Conant et al. 2009; NMFS 2010b, 2010c, 2011a, 2015c, 2018f; NMFS and FWS 1998a, 1998b, 1998c, 1998d, 2007a, 2007b, 2007c, 2007d, 2010b, 2010c, 2011a, 2013a, 2014b, 2015, 2019d, 2020; Miller et al. 2014; Seminoff et al. 2015; Miller and Klimovich 2017; Young et al. 2017) and other sources of information as cited herein.

Updates to the regulations governing interagency consultations (50 CFR part 402) became effective on October 28, 2019 [84 FR 44976]. This consultation was completed under the new regulations, which were designed to improve clarity and consistency, streamline consultations, and codify existing practice. As the preamble to the final rule adopting the regulations noted, "[t]his final rule does not lower or raise the bar on Section 7 consultations, and it does not alter what is required or analyzed during a consultation. Instead, it improves clarity and consistency, streamlines consultations, and codifies existing practice."

1.1 Consultation History

The proposed federal action addressed by this biological opinion is the authorization of the United States WCPO purse seine fishery, as currently managed. On November 1, 2006, NMFS completed a biological opinion for this fishery and concluded that the continued authorization of the United States WCPO purse seine fishery was not likely to jeopardize the continued existence of blue whales, fin whales, humpback whales, sei whales, sperm whales, green sea turtles, hawksbill sea turtles, leatherback sea turtles, loggerhead sea turtles, or olive ridley sea turtles. There were no prior consultations on this fishery.

Since NMFS completed the 2006 biological opinion, the following species that occur in the area of operation of the United States WCPO purse seine fishery have been listed as threatened or endangered under the ESA: threatened Indo-West Pacific and endangered Eastern Pacific scalloped hammerhead sharks (*Sphyrna lewini*); 15 species of threatened corals (*Acropora globiceps, Acropora jacquelineae, Acropora lokani, Acropora pharaonis, Acropora retusa, Acropora rudis, Acropora speciosa, Acropora tenella, Anacropora spinosa, Euphyllia paradivisa, Isopora crateriformis, Montipora australiensis, Pavona diffluens, Porites napopora,* and *Seriatopora aculeata*); threatened chambered nautilus (*Nautilus pompilius*); threatened giant manta ray (*Manta birostris*); threatened oceanic whitetip shark (*Carcharhinus longimanus*); three distinct population segments (hereinafter referred to as species) of loggerhead sea turtles (*Caretta caretta*): endangered North Pacific, endangered South Pacific, and threatened Southeast Indo-Pacific loggerhead sea turtles; six species of green sea turtles (*Chelonia mydas*): threatened East Indian-West Pacific, endangered Central West Pacific, threatened Southwest Pacific, endangered Central South Pacific, threatened Central North Pacific, and threatened East Pacific green sea turtles; and the endangered Western North Pacific humpback whale (*Megaptera novaeangliae).*

On October 21, 2014, NMFS IFD requested reinitiation of formal Section 7 consultation on the effects of the United States WCPO purse seine fishery on the Indo-West Pacific scalloped hammerhead shark. PRD acknowledged the request however there was insufficient data on the number of interactions or expected effects to the species from this fishery at that time. NMFS PRD worked with NMFS IFD to collect additional information to facilitate initiation of consultation for all ESA-listed species that may be affected by the proposed action.

On December 5, 2017, NMFS IFD requested consultation on, and provided a biological assessment for, the effects of the continued operation of the United States WCPO purse seine fishery on the following species: blue whales, sei whales, sperm whales, East Indian-West Pacific green sea turtles, Southwest Pacific green sea turtles, Central West Pacific green sea turtles, Central South Pacific green sea turtles, Central North Pacific green sea turtles, East Pacific green sea turtles, hawksbill sea turtles, leatherback sea turtles, South Pacific loggerhead sea turtles, Southeast Indo-Pacific loggerhead sea turtles, North Pacific loggerhead sea turtles, olive ridley sea turtles, Eastern Pacific Scalloped hammerhead sharks, and Indo-West Pacific scalloped hammerhead sharks. On the same date, NMFS IFD also requested our concurrence that the continued operation of the fishery was not likely to adversely affect the following species: fin whales, Western North Pacific humpback whales, *A. globiceps, A. jacquelineae, A. lokani, A. pharaonis, A. retusa, A. rudis, A. speciosa, A. tenella, A. spinosa, E. paradivisa, I. crateriformis, M. australiensis, P. diffluens, P. napopora,* and *S. aculeata.*

On January 22, 2018, NMFS published a final rule to list the giant manta ray as a threatened species under the ESA (83 FR 2916); this rule became effective February 21, 2018.

On January 30, 2018, NMFS published a final rule to list the oceanic whitetip shark as a threatened species under the ESA (83 FR 4153). The rule became effective March 1, 2018.

On May 2, 2018, NMFS IFD provided information via email on oceanic whitetip shark and giant manta ray interactions with the United States WCPO purse seine fishery and on May 17, 2018, they requested consultation on these two species.

On July 16, 2018, PRD sent a letter to IFD to inform them that they provided all information required by regulation for initiating interagency consultation, as outlined in 50 CFR 402.14.

On September 28, 2018, NMFS published a final rule to list the chambered nautilus as a threatened species under the ESA (83 FR 48976) effective October 29, 2018.

On December 6, 2018, NMFS IFD submitted a request for concurrence on their determination that the United States WCPO purse seine fishery was not likely to adversely affect the chambered nautilus.

On February 27, 2019, NMFS PRD requested that IFD provide updated data to include more recent years since updated information was available regionally for the entire WCPO fleet and the data previously submitted was through 2015.

On March 5, 2019, NMFS denied American Tunaboat Association's (ATA) request for Applicant status. That decision was affirmed in litigation. See American Tunaboat Association v. Ross, 391 F.Supp.3d 98 (DDC 2019). Although ATA is not an Applicant, NMFS agreed to provide the organization with an opportunity to review the draft biological opinion consistent with its authority under the ESA Section 7 Consultation Handbook, 2-13.

On August 19, 2019, NMFS IFD provided updated information on the future anticipated level of effort to be analyzed based on effort levels from 2015-2018 (NMFS 2019c). They also provided an updated analysis on the future level of interactions for listed species to be considered in this consultation based on the anticipated level of effort (Western and Central Pacific Fisheries Commissions (WCPFC) unpublished data).

On December 10, 2019, NMFS IFD and PRD cooperatively conducted an updated analysis for sei whales, South Pacific loggerhead sea turtle, East Indian-West Pacific, Central West Pacific, Southwest Pacific, Central South Pacific, and East Pacific green sea turtles after receiving more data from the SPC.

On January 29, 2020, PRD provided IFD with a rough draft of the biological opinion and requested their limited review of the description of the action and the consultation history.

On February 27, 2020, NMFS IFD provided comments to NMFS PRD on these select aspects of the January 29, 2020 draft biological opinion.

On May 29, 2020, NMFS PRD provided an updated copy of the draft biological opinion to NMFS IFD for review.

On June 19, 2020, NMFS IFD provided additional comments in response to NMFS PRD.

On June 23, 2020, NMFS PRD brought a concern to NMFS IFD regarding the *Action Area* and whether it properly defined the use of foreign ports and species that could be affected by vessel transiting activities in the Eastern Pacific Ocean (EPO).

On June 25, 2020, NMFS IFD provided an initial response to NMFS PRD confirming vessels transport to foreign ports, but indicating that vessel transiting in the EPO was likely outside the scope of the proposed action.

On July 29, 2020, NMFS IFD requested by electronic memo to Ann Garrett, that provided supplemental information to expand the *Action Area* to include vessel transiting. NMFS PRD confirmed the request. By expanding the *Action Area*, three additional species fell within the *Action Area*. NMFS IFD requested concurrence in the course of the consultation, with their determination that the proposed action would not adversely affect the Guadalupe fur seal (*Arctocephalus townsendi*), or two distinct population segments (DPS) of humpback whale (*Megaptera novaeangliae*) - the Mexico DPS, and the Central America DPS.

After further consideration, on July 29, 2020, NMFS PRD found a fin whale (*Balaenoptera physalus*) record in the interaction data. Reviewing the GEN2 forms revealed two other interactions with the species.

On July 30, 2020, NMFS PRD and IFD met to review these records and to determine a path forward in regards to the effects analysis. It was agreed that NMFS IFD's initial may affect, not likely to adversely affect determination for fin whale, was no longer applicable and the species was included in the analysis for species likely to be adversely affected.

On August 6, 2020, NMFS PRD provided a response to IFD's comments regarding population structure, use of interactions rather than Catch Per Unit Effort (CPUE), and incorporation of historic CPUE as a proxy for abundance trends for the giant manta ray analysis.

On January 13, 2021, NMFS PRD provided a subsequent draft biological opinion to NMFS IFD to share with the American Tunaboat Association.

On February 5, 2021, the American Tunaboat Association provided comments to NMFS on the draft biological opinion.

On March 11, 2021, NMFS IFD proposed changes to the action, reducing the expected number of average annual sets from 7,200 to 3,100; with up to 1,581 of them being FAD sets and shifting the expected distribution of fishing effort, based in part on the reduction in number of vessels in the fleet.

On April 2, 2021, NMFS IFD presented NMFS PRD with a memorandum documenting its changes to the proposed action, including a reduction in expected level of fishing effort and a shift in the spatial distribution of fishing effort.

On April 6, 2021, NMFS IFD provided NMFS PRD the information on vessels that recently left the fishery for further analyses. The change in expected fishery effort provided substantial new evidence requiring a new analysis for the change in expected sets, spatial distribution of the fleet, and the number of interactions with ESA-listed species which are reasonably certain to occur in the future in the *Action Area*. NMFS PRD confirmed the information was complete and reinitiated consultation.

1.2 Description of the Proposed Action

NMFS IFD proposes to authorize the United States purse seine fishery (fishery) in the WCPO as currently managed, including through the maintenance/issuance of regulations under the authorities described below. As specified above in Section 1.1 *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 WCPO purse seine fishery as currently managed. This fishery is governed in part by the "Treaty on Fisheries between the Governments of Certain Pacific Island States and the Government of the United States of America" also known as the South Pacific Tuna Treaty (Treaty), an international agreement to which the United States is a party. NMFS implements the terms of the treaty by issuing regulations under the authority of the South Pacific Tuna Treaty Act of 1988 (SPTA) (16 U.S.C. Chapter 16C). The regulations considered as part of this action include both regulations currently in effect (50 CFR 300 Subpart D) and new regulations being developed by the IFD to implement recent technical modifications under the Treaty and Treaty amendments including those agreed to in 2016. The action also includes regulation of the United States WCPO purse seine fishery 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). Accordingly, this consultation includes the effects of the authorization of all purse seine fishing in or transiting through the *Action Area* by United States-flagged vessels as currently managed.

United States tuna purse seine vessels in the WCPO range in length from 53 to 79 m in registered length with the largest being able to hold up to 2,200 m^3 of frozen fish (WCPFC 2021b). Vessels in this fishery rely on multiple methods to identify tuna school locations including the use of fish aggregating devices (FADs, discussed in detail below) and observations of concentrated seabird activity. In 2020, 14 out of 24 of the United States WCPO purse seine fishery fleet maintained helicopters onboard to additionally assist in spotting schools of tuna. Once a school is identified, the purse seining technique for catching tuna involves employing a net that is set vertically in the water with floats attached to the upper edge and chains for weight on the lower edge (Figure 1). A series of rings is attached to the lower edge of the net, and a pursing cable passes through the rings, enabling a winch on board the vessel to draw the net closed on the bottom. Purse seine nets can be up to 1,500 meters (m) or more in length and 150 m deep. When the net is deployed from the purse seine vessel, a large skiff attached to the end of the net is slid off the stern of the fishing vessel. The skiff holds one end of the net as the net is deployed by the fishing vessel. On a successful set the purse seine vessel encircles the school of tuna, keeping it in visual contact if at the surface, or using sonar if below the surface, and then retrieves most of the pursed net back onto the vessel. At this point the skiff is used to tow the fishing vessel as needed to prevent it from drifting over the net. The fish are confined in the "sack" portion of the net, which consists of finer mesh webbing that prohibits their escape. The catch is removed from the sack alongside the vessel and brought onto the vessel with large "scoops," known as brails, which hold several metric tons (mt), and is then placed in one of many brine tanks for freezing and later storage (NMFS 2006).

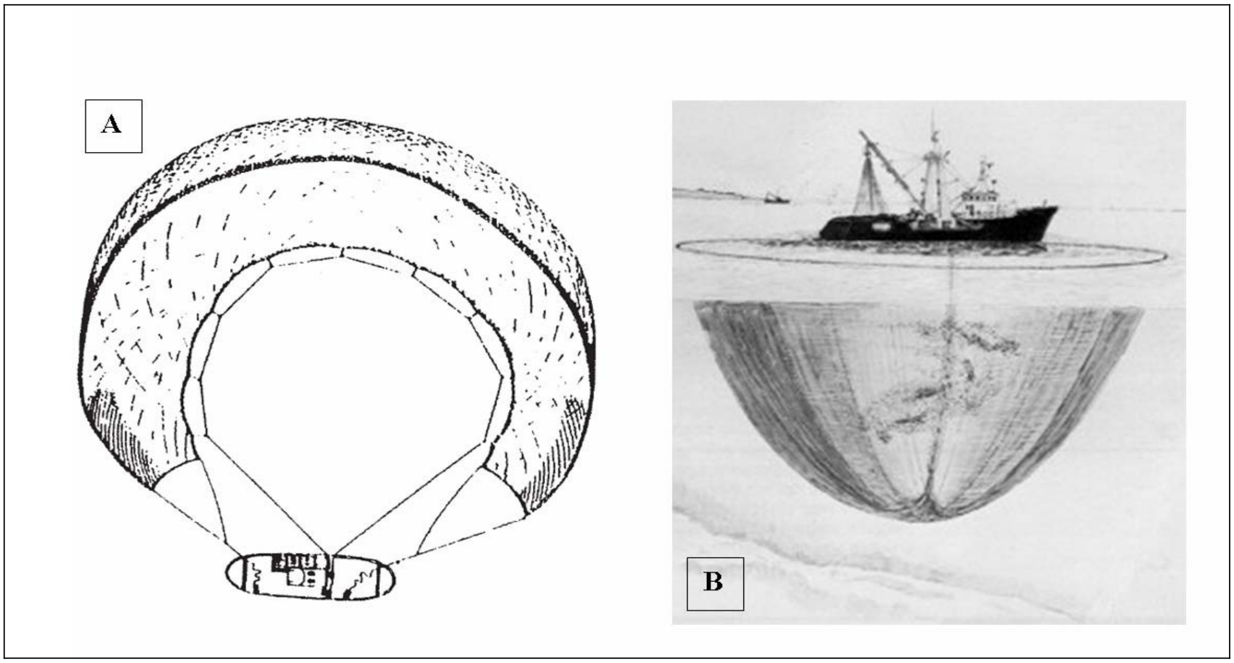


Figure 1. Examples of purse seine net deployment from above (A; illustration from Sainsbury 1971) and the side (B; illustration from Industry Canada 1996).

United States WCPO purse seine sets can be divided into two types, those made on drifting FADs (artificial or natural), also known as "associated sets", and those made on free-schools of tuna, also known as "unassociated sets". The latter are generally made on concentrations of seabirds, baitfish and tuna. Manmade (also, artificial) FADs are small rafts, often made of bamboo, plastic pipe or wood, and typically have net or rope hanging from them, which passively drift with the currents (Restrepo et al. 2017). These FADs are typically outfitted with a tracking buoy. Natural FADs include natural logs, and other objects.

Sets, including all associated activities such as deploying, encircling, pursing, retrieving, bagging up, brailing and final net retrieval, are typically conducted during the day. Set times are usually recorded as the time when the net first hits the water. Unassociated sets are predominately initiated (90%) in daylight hours and the associated sets are predominantly (90%) initiated prior to sunrise (Brouwer and Pilling 2017). Currently, sets are prohibited at night during the three-month FAD closure (July through September; discussed in more detail below) in Pacific Island Countries' Economic Exclusion Zones (EEZs) under the terms of the South Pacific Tuna Treaty

(Regional Terms and Conditions). The time it takes to set and haul varies depending on the catch. If there is no catch, then it is fairly quick, but if there is a large school it may take many hours to brail the entire haul. On average in recent years, about one set is made per day fishing.

Since the late 1990s, the United States WCPO purse seine fishery has increased its reliance on sets made on drifting FADs (Figure 2) setting on natural floating objects had been a practice since inception of the fishery. Reliance on FADs has varied greatly however, accounting for more than 90% of all sets in some years, and less than 30% on other years. There are many factors that cause this variability for the United States fleet, not all of which are fully understood (i.e., other than perhaps by the purse seine vessel operators themselves). However, some general determinates can be postulated: FADs provide an almost guaranteed location of fish although the size of the schools associated with FADs can vary considerably (NMFS 2017a), and the average size of the bigeye and yellowfin caught on FAD sets is generally smaller than those caught in unassociated sets (Leroy et al. 2013). Electronic buoys with sonar devices can better indicate the volume or biomass of tuna held by a FAD.

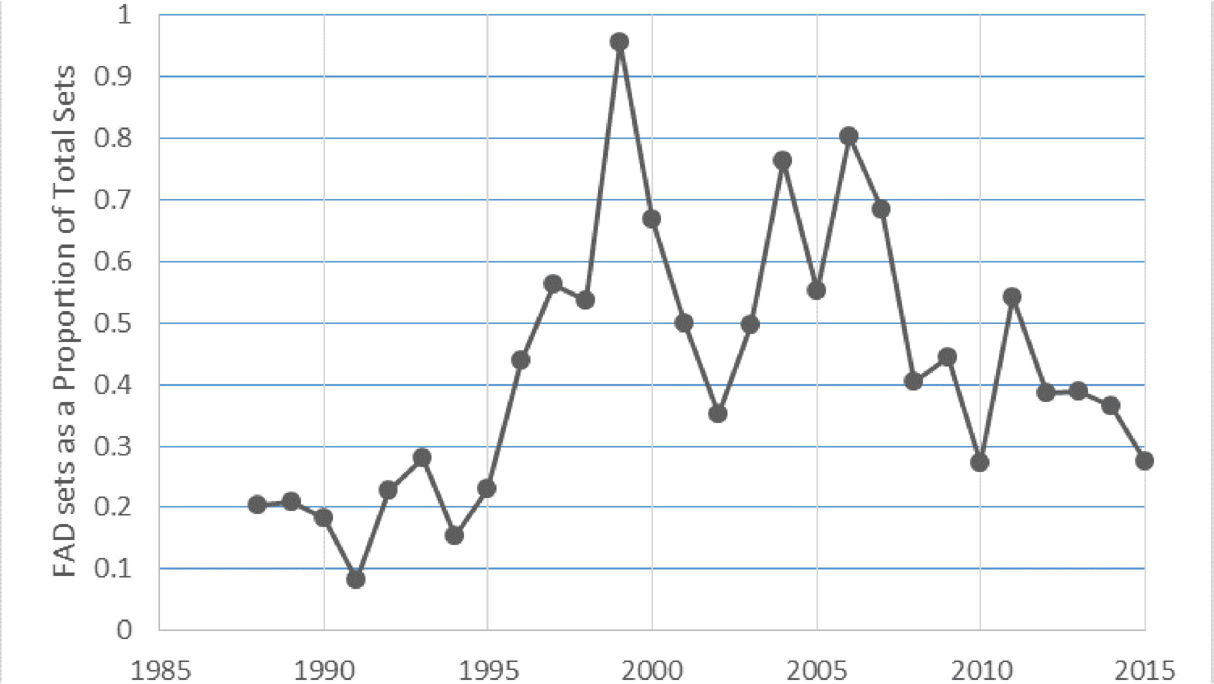


Figure 2. FAD sets as a Proportion of all Sets by United States WCPO Purse Seine Fleet, 1988-2015 (NMFS 2017a).

Beginning in 2009, the proportion of sets made on FADs has been constrained by regulations promulgated under the Western and Central Pacific Fisheries Convention Implementation Act (WCPFC Implementation Act; 16 U.S.C. 1801 *et seq*.) to implement decisions of the Commission that restrict the use of FADs in the United States WCPO purse seine fishery. The regulations established one or more closed periods each year during which United States WCPO purse seine vessels were prohibited from fishing on FADs in the WCPFC Convention Area, which comprises the WCPO and a section of the East Pacific Ocean between 150°W and 130°W longitude and south of 4°S. The established FAD closure period was from August 1 through

September 30 in 2009; from July 1 through September 30 in 2010, 2011, and 2012; from July 1 through October 31 in 2013, 2014, and 2015; and from July 1 through September 30 in 2016 through the present. In addition, in the years 2016 and 2017 there was an established limit of 2,522 FAD sets per year for the United States WCPO purse seine fleet, and in the years from 2018, there has been a limit of 350 active drifting FADs per vessel, along with an additional FAD closure for high-seas sets in November and December (83 FR 33851). FADs remain in the water for six months or more, and the time between checking FADs varies with tracking buoy information and/or vessel proximity. Given this array of complex and offsetting factors, fishing operations for the United States WCPO purse seine fleet will likely continue using a combination of unassociated and associated sets.

Current Management Regime

The South Pacific Tuna Treaty (SPTT) currently provides the United States WCPO purse seine fleet access to the fishing grounds in most of the EEZs of the Pacific Island Parties of the Treaty (PIPs), and it is in these areas where the majority of United States WCPO fishing takes place. Some countries, like Australia and New Zealand do not allow access to their EEZ. Additionally, this fishery is regulated under the WCPFCIA (16 U.S.C. 6901 *et seq*.) and implementing regulations (50 CFR Subpart O), High Seas Fishing Compliance Act (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 (Pelagic FEP) pursuant to the MSA (50 CFR Part 665). In addition, NMFS may regulate the fishery to meet the requirements of the Endangered Species Act (16 U.S.C. 1531 et seq.), Marine Mammal Protection Act (16 U.S.C. 1361 et seq.), Magnuson-Stevens Fishery Conservation and Management Act (16 U.S.C. 1801 et seq.), and other applicable laws. Of the main fishery management regulations, we highlight the following for this consultation:

Fishing authorizations with implementing act in parentheses;

- If fishing on the high seas anywhere, a fishing permit in accordance with the High Seas Fishing Compliance Act.
- If fishing on the high seas in the WCPFC Convention Area, a WCPFC Area Endorsement (WCPFCIA).
- To fish in the areas to which the terms of the SPTT apply, a license issued by the Pacific Islands Forum Fisheries Agency (FFA) as Treaty Administrator on behalf of the PIPs. Under the SPTT, vessels purchase a certain amount of fishing days each year to fish in the waters of the PIPs (SPTA).

Prohibitions;

- Prohibited from transshipping fish at sea in the WCPFC Convention Area and from transshipping fish caught in the WCPFC Convention Area anywhere else (50 CFR 300.216; WCPFCIA).
- Vessels operating the WCPFC Convention Area subject to fishing day limits in the United States EEZ and on the high seas in the WCPF Convention Area between the latitudes of 20N and 20S (an area known as the ELAPS). The annual ELAPS limit currently in effect is 1,828 fishing days (50 CFR 300.223(a); WCPFCIA).

- Restrictions on the use of FADs. The restrictions have varied somewhat over the years, but have typically involved seasonal prohibitions on setting on FADs, or "FAD closures." The FAD closures currently in effect are July-September (applicable throughout the WCPF Convention Area), and November-December (applicable on the high seas in the WCPF Convention Area) (50 CFR 300.223(b); WCPFCIA);
- Vessels equal to or greater than 50 feet (ft.; 15.2 m) in length overall generally cannot fish in a certain portion of the United States EEZ around American Samoa (50 CFR 665.806(b); MSA).

Observer Placement;

- Since 2010, vessels are required to carry WCPFC observers on all trips (the observers are typically deployed by the FFA and its member observer programs in their capacity as observer providers under the WCPFC Regional Observer Programme; WCPFCIA). However, due to the COVID pandemic observer coverage has been less than 100% in 2020 and 2021.

Reporting, Monitoring and Gear Identification;

- Holders of SPTT licenses are required to submit both reports on their fishing activities in the *Action Area* to NMFS and the FFA (50 CFR 300.34); SPTA);
- Required to carry and operate mobile transmitting units to provide automated position information as part of vessel monitoring system (VMS) administered by NMFS and by the FFA (50 CFR 300.219; WCPFCIA).

Sea Turtle Mitigation Measures (WCPFCIA);

- Possession and use of a dip net with an extended reach handle with a minimum length of 150% of the freeboard height that is made of wood or other rigid material able to support a minimum of 100 pounds (lbs.) without breaking or significant bending or distortion.
- A net hoop of at least 31 inches (in; 78.74 cm) inside diameter and a bag depth of at least 38 in (96.52 cm). The bag mesh openings may be no more than 3 inches × 3 inches (7.62 cm × 7.62 cm) in size.
- An optional turtle hoist used for the same purposes as a dip net. It is not a required piece of gear, but a turtle hoist may be carried on board and used instead of the dip net to handle sea turtles. The minimum design standards for turtle hoists that are used instead of dip nets to meet the requirements of this section are:
    - *Frame and net.* The turtle hoist must consist of one or more rigid frames to which a bag of mesh netting is securely attached. The frame or smallest of the frames must have a minimum opening (e.g., inside diameter, if circular in shape) of 31 inches (78.74 cm) and be capable of supporting a minimum of 100 lbs. (34.1 kg). The frame or frames may be hinged or otherwise designed so they can be folded for ease of storage, provided that they have no sharp edges and can be quickly reassembled. The bag mesh openings may be no more than 3 inches × 3 inches (7.62 cm × 7.62 cm) in size.

        - *Lines.* Lines used to lower and raise the frame and net must be securely attached to the frame in multiple places such that the frame remains stable when lowered and raised.

- Any captured or entangled sea turtle must be handled in a manner to minimize injury and promote survival.
- If the sea turtle is observed enclosed in a purse seine but not entangled, it must be released immediately from the purse seine with the dip net or turtle hoist.
- If the sea turtle is observed entangled in a purse seine, the net roll must be stopped as soon as the sea turtle comes out of the water, and must not start again until the turtle has been disentangled and released.
- If the sea turtle is observed entangled in a FAD, it must be disentangled or the FAD must be cut immediately so as to remove the sea turtle.
- After disentanglement, if the sea turtle is not already on board the vessel and it is too large to be brought aboard or cannot be brought aboard without sustaining further injury, it shall be left where it is in the water, or gently moved, using the dip net or turtle hoist if necessary, to an area away from the fishing gear and away from the propeller.
- After disentanglement, if the sea turtle is not too large to be brought aboard and can be brought aboard without sustaining further injury, the following actions shall be taken:
    - Using the dip net or a turtle hoist, the sea turtle must be brought aboard immediately.
    - If a sea turtle brought aboard appears dead or comatose, the following actions must be taken:
        - The sea turtle must be placed on its belly (on the bottom shell or plastron) so that it is right side up and its hindquarters elevated at least 6 inches (15.24 cm) for a period of no less than 4 hours and no more than 24 hours. The amount of the elevation varies with the size of the sea turtle; greater elevations are needed for larger sea turtles;
        - A reflex test must be administered at least once every 3 hours. The test is to be performed by gently touching the eye and pinching the tail of a sea turtle to determine if the sea turtle is responsive;
        - The sea turtle must be kept shaded and damp or moist (but under no circumstances place the sea turtle into a container holding water). A water-soaked towel placed over the eyes (not covering the nostrils), carapace and flippers is the most effective method of keeping a sea turtle moist; and
        - If the sea turtle revives and becomes active, it must be returned to the sea. Sea turtles that fail to revive within the 24-hour period must also be returned to the sea, unless NMFS requests that the turtle or part thereof be kept on board and delivered to NMFS for research purposes.

    - After handling a sea turtle, the sea turtle must be returned to the ocean after identification unless NMFS requests the retention of a sea turtle for research. In releasing a sea turtle, the vessel owner or operator must:
        - Place the vessel engine in neutral gear so that the propeller is disengaged and the vessel is stopped;
        - Using the dip net or a turtle hoist to release the sea turtle with little impact, gently release the sea turtle away from any deployed gear; and
        - Observe that the turtle is safely away from the vessel before engaging the propeller and continuing operations.
- No sea turtle, including a dead turtle, may be consumed, or sold. A sea turtle may be landed, offloaded, transshipped, or kept below deck only if NMFS requests the retention of a dead turtle or a part thereof for research.

Shark Handling and Mitigation Measures (WCPFCIA);

- United States purse seine vessels must not set or attempt to set on around a whale shark (*Rhincodon typus*) and must release any whale shark that is encircled (50 CFR 300.223(g));
- United States purse seine vessels cannot retain on board, tranship, store, or land any part or whole carcass of an oceanic whitetip shark (*Carcharhinus longimanus)* or silky shark (*Carcharhinus falciformis*) and must release any oceanic whitetip shark or silky shark as soon as possible (50 CFR 300.226);

Mobulid Ray Bycatch Mitigation (WCPFCIA; will go into force January 1, 2021, and NMFS anticipates implementing through regulations soon after);

- HMS fishing vessels must not target or intentionally set on mobulid rays in the WCPFC Area.
- HMS fishing vessels must not retain on board, tranship, or land any part or whole carcass of mobulid rays caught in the WCPFC Area.
- HMS fishing vessels must promptly release alive and unharmed, to the extent practicable, mobulid rays as soon as possible, and to do so in a manner that will result in the least possible harm to the individuals captured.
- HMS fishing vessels are encouraged to implement specific handling practices, while taking into consideration the safety of the crew.

Beyond the closed areas cited above, in 2006 Kiribati formed the Phoenix Islands Protected Area (PIPA) in a portion of its EEZ, which is about 140,000 m^2 in size. On January 1, 2015, Kiribati banned all commercial fishing within the PIPA. This prohibition applies to the United States WCPO purse seine fleet.

As mentioned above, regulations to implement Commission decisions, including limits on fishing effort and restrictions on the use of FADs, have been in place since 2009. The specifics of these regulations vary from year to year, based on Commission decisions, but are likely to continue for the reasonably foreseeable future.

For this consultation, NMFS IFD (T. Graham memo to A. Garrett April 2, 2021) described the foreseeable future for the United States WCPO purse seine fishery as:

- A maximum of 40 vessels participating (the 13-year average from 2008 to 2020 was 36 participating vessels; from 2016 to 2020, 37, 35, 34, 31 and 23 vessels participated, respectively).
- Total fishing effort in most years is expected to be no more than 3,100 sets, and the proportion of sets on FADs is expected to be about the same as the rate set from 2008-2018. Therefore, we expect up to 1,581 FAD sets a year. The basis for these numbers is described in the memo to T. Graham to A. Garrett April 2, 2021, which represents an adjustment to previous *correspondence* (*or BE*), to account for changes in the anticipated level of fishing effort presented in both NMFS 2017a and the T. Graham memo to A. Garrett, dated August 19, 2019.
- Spatial distribution of fishing effort, including inter-annual variability, determined primarily by: (1) access to specific fishing grounds, and (2) El Nino-Southern Oscillation (ENSO) conditions. Lacking specific information on those two factors, the spatial distribution is expected to be about the same as in the 2008-2020 period. Except the longitudinal aspect of the distribution of sets is expected to be approximately as follows (for the reasons described in the T. Graham memo to A. Garrett on April 2, 2021). This represents the number of sets assuming 20 vessels operating, and the expectation that additional entrants to the fleet are not expected because the fleet is not building new capacity that meets United States documentation requirements; and no departed vessels are expected to reenter. Additionally, the remaining vessels in the fleet have historically fished further east than the vessels that recently exited the fleet, and the agency expects the remaining vessels to continue fishing in the same geographic area they have been.

Geographical Range	135E-160E	160E-175E	175E-175W	175W-130W
Expected distribution (percent)	7%	20%	27%	45%
2008-2018 distribution (for reference; percent)	19%	33%	22%	26%

- Temporal distribution of fishing effort within years (seasonal), including inter-annual variability, about the same as in recent years (e.g., 2008-2018) but seasonal distribution of specific set types might be strongly influenced by seasonal prohibitions on the use of FADs, as in 2009-2018.

From January 2018 through February 2021, 15 United States purse seine vessels exited the United States fishery, and no vessels entered (T. Graham memo to A. Garrett April 2, 2021). Since 2021 two additional vessels left the fishery. The current number of vessels authorized as part of the United States WCPO purse seine fishery is 18. Anecdotal information suggests that two additional vessels might leave the fishery soon. NMFS IFD has no firm or anecdotal information suggesting that vessels will enter the fishery in the foreseeable future.

Of the 15 vessels that left the fishery between 2018 and 2021, 14 were managed by a single company, and they comprised all the vessels in the fishery that operated solely under the "alternative business model" described in Section 2.2.3.3 of the BA. While the remainder of the fleet delivered most catches to the canneries in Pago Pago, the vessels operating under the alternative model were not based (as a practical matter) out of any specific port, and tended to transship their catches at the port nearest to their preferred fishing grounds. As a result, they tended to fish further west than the rest of the fleet. They offloaded most of their catches in foreign ports well to the west of Pago Pago, including Majuro, Pohnpei, Tarawa, Rabaul, and Honiara. Although there are other influences on the longitudinal aspect of the spatial distribution of the fleet's fishing effort, particularly ENSO conditions, the two different business models in the fishery resulted in marked differences in fishing grounds for the vessels that recently left the fishery and those that remain. We highlight that the vessels, which have historically practiced this alternative business model, have left the fleet.

Vessel Offloading

The purse seine catch is stored on board as a frozen whole product- typically in large brine wells although some may store the fish in refrigerated spaces called "dry wells". Most of the catch of the United States WCPO purse seine fleet was historically offloaded to canneries in Pago Pago, American Samoa, located slightly beyond the southeastern limit of the fishery's main fishing grounds. In 2008 a component of the United States WCPO purse seine fleet began adopting an alternative business model, which called for vessels to transship to carrier vessels at to the closest possible port in the shortest possible time (Gillett et al. 2002; Hamilton et al. 2011). These vessels do not base out of a single specific port in the region, rather they maintain the flexibility to transship in ports depending on where they are fishing. Vessels that are part of the United States WCPO purse seine fleet may also fish in the EPO on the same trip. These vessels may land catch caught on the same trip in the WCPO and in the EPO in ports in both the WCPO and in the EPO (Table 1). NMFS manages the fishing activities of United States purse seine vessels fishing in the EPO under a separate management regime. This United States EPO purse seine fishery is managed under the West Coast Highly Migratory Species Management Plan and its implementing regulations (See 50 CFR 660 Subpart K), as well NMFS regulations implementing decisions of the Inter-American Tropical Tuna Commission (IATTC) under the Tuna Conventions Act (16 U.S.C. 951 et seq.) (See 50 CFR 300 Subpart C) and NMFS regulations implementing the Agreement of the International Dolphin Conservation Act under the Marine Mammal Protection Act (16 U.S.C. 1361 et seq.) (See 50 CFR 200 Subpart C). NMFS completed ESA Section 7 consultation for the United States EPO purse seine fishery in 2004 (NMFS 2004). Hence, while the fishing operations that may occur in the EPO are not part of the action considered in this biological opinion, any transit and offloading activities of United States vessels carrying catch from operations in the WCPO to ports outside of the WCPO are considered. EPO fishing activities and impacts from those fisheries are considered in the *Environmental Baseline* if they are present in the *Action Area* for the proposed action. As shown in Table 1, since 2008, most of the catch of the United States fleet has been offloaded in foreign ports, from which it is eventually transported to canning facilities in Southeast Asia, Latin America, and the United States (NMFS 2017a).

Fishing Days

The regulations under which the United States fleet operates require changes in response to new decisions of the Commission and new provisions of the Treaty, as well as changes in other laws. The Commission typically adopts new conservation and management measures relevant to this fishery annually. The PIPs and the United States agreed to amendments to the Treaty and its Annexes in 2016, along with a Memorandum of Understanding regarding their intent to provisionally apply some of the amendments pending completion of ratification and entry into force. Some provisions of the Treaty Annexes extend only through 2020 or 2022.

Under the SPTT, vessel owners have the opportunity to buy "upfront" fishing days from the Pacific Island parties. To buy these days, by July 1 of each year, vessel owners must commit to buying a specific number of fishing days for the following calendar year. The commitment is on a per-owner basis, broken down into a few pools of PIC EEZs. Once that commitment is made, the full payment has to be made, regardless of whether the days are used. Fees for 2020 upfront days for each of the pools were:

PNA members (except Kiribati): $13,000/day

Kiribati: $13,000/day

Cook Islands: $9,533/day

Other FFA members: flat payment + $6,000/"successful" day

In addition to those "upfront" days, vessel owners have the opportunity to buy "additional" fishing days on a bilateral basis with particular Pacific Island countries. There are no preset constraints on price or timing for additional days. The opportunity to buy additional days brings less of a risk/cost than the upfront days, but the upfront days have a premium value because of their pooled nature and pre-negotiated price and availability.

Lastly, there are no access fees to fish on the high seas (typically 15% of total effort) or in the United States EEZ (typically 5% of total effort).

Table 1. Retained Tuna Catch (mt) of United States WCPO Purse Seine Vessels, by Port where Landed or Transhipped, 2005-2018. Country/territory abbreviations are as follows: American Samoa (AS), Federated States of Micronesia (FSM), Papua New Guinea (PNG), Republic of the Marshall Islands (RMI), Solomon Islands (SB) (United States Coast Guard 2007; United States Coast Guard and NMFS 2008, 2009, 2010, 2011, 2012, 2013, 2014, 2016, 2017, 2018).

Port	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**	**2018**
Pago Pago, AS	64,777	50,144	52,156	76,584	69,380	27,165	38,913	58,823	49,312	86,462	67,540	83,710	66,726	85,714
Pago Pago, AS Transshipment	0	0	0	0	0	29,268	18,964	33,451	33,453	53,131	43,285	0	0	0
Foreign Ports														
Pohnpei, FSM	1,705	6,054	4,477	25,641	75,983	59,781	26,620	23,274	69,945	11,848	3,470	33,112	42,599	41,824
Christmas Island, Kiribati	–	–	–	–	–	–	–	7,877	8,522	3,857	1,620	–	4,642	3,514
Tarawa, Kiribati	1,503	869		4,552	7,417	8,135	5,577	10,647	7,120	5,809	3,158	–	–	–
Rabaul, PNG	1,657	2,896	787	12,198	6,473	14,449	11,801	12,428	4,920	–	–	3,062		2,720
Majuro, RMI	12,642	6,869	8,006	46,381	71,592	78,939	78,662	106,820	74,134	133,631	98,635	49,846	38,982	38,566
Honiara, SI	–	658	3,421	14,713	21,846	22,356	21,668	–	–	–	4,530	–	–	–
Wewak, PNG	–	85	–	4,272	14,405	–	–	–	–	–	–	–	–	–
Noro, SI	–	434	411	464	–	–	–	–	–	–	–	–	–	–
Bangkok, Thailand	775	–	–	1,675	–	–	–	–	–	–	–	–	–	–
Funafuti, Tuvalu	–	–	–	–	–	–	–	–	–	–	20,177	14,830	10,425	6,750
Manta, Ecuador	1,204	–	–	–	–	–	–	–	–	–	–	–	–	9,968
Mazatlan, Mexico												9,970	21,463	5,665

Port	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**	**2018**
Other	–	–	–	–	173[1]	4,701[2]	2,942[3]	5,214[4]	4,232[5]	1,828[6]	4,978	5,932[7]	10,439[8]	4,220[9]
Foreign Ports Total	19,486	17,865	17,102	109,896	197,889	188,361	147,270	166,260	206,966	156,973	136,568	116,752	128,550	113,227
Total	82,606	68,009	69,258	191,480	272,798	244,793	205,147	258,534	251,638	296,566	247,394	200,462	195,276	198,941
Foreign Ports %	24%	26%	25%	57%	73%	77%	72%	64%	82%	53%	55%	58%	66%	57%

[1] Combined data from the following ports: Christmas Island, Kiribati; General Santos, Philippines; Madang, Papua New Guinea; Manta, Ecuador; Noro, Solomon Islands; and Misaki, Korea

[2] Combined data from the following ports: Funafuti, Tuvalu; General Santos, Philippines; Manta, Ecuador; Noro, Solomon Islands; Wewak, Papua New Guinea; and Zhoushan, China.

[3] Combined data from the following ports: Noro, Solomon Islands and Suva, Fiji.

[4] Combined data from the following ports: Funafuti, Tuvalu; General Santos, Philippines; and Honiara, Solomon Islands.

[5] Combined data from the following ports: General Santos, Philippines, and Honiara, Solomon Island.

[6] Combined data from the following ports: Manta, Ecuador; Honiara, Solomon Islands; Kaohsiung, Taiwan; and Funafuti, Tuvalu

[7] Combined data from the following ports: Manta, Ecuador, La Union, El Salvador, and Mazanillo, Mexico

[8] Combined data from the following ports: Manta, Ecuador, La Union, El Salvador, Mazanillo, Mexico, and Paita, Peru

[9] Combined data from the following ports: Kosrae, Federated States of Micronesia, Mazanillo, Mexico, and Honiara, Solomon Islands

2 APPROACH TO THE ASSESSMENT

2.1 Overview of NMFS Assessment Framework

Biological opinions address two central questions: (1) has a Federal agency insured that an action it proposes to authorize, fund, or carry out is not likely to jeopardize the continued existence of endangered or threatened species and (2) has a Federal agency insured that an action it proposes to authorize, fund, or carry out is not likely to result in the destruction or adverse modification of critical habitat that has been designated for such species. Every section of a biological opinion from its opening page and its conclusions and all of the information, evidence, reasoning, and analyses presented in between is designed to help answer these two questions. What follows summarizes how NMFS generally answers these two questions; that is followed by a description of how this biological opinion will apply this general approach to the United States WCPO purse seine fishery.

Before we introduce the assessment methodology, we want to define the word "effect." An *effect* is defined by the ESA regulations to include a "consequence to listed species or critical habitat that is caused by the proposed action," and may include "the consequences of other activities that are caused by the proposed action (50 CFR 402.02)." 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 (50 CFR 402.02, 402.17)." A conclusion of "reasonably certain to occur" must be based on "clear and substantial information, using the best scientific and commercial data available (50 CFR 402.17)." This definition of "effect" is neutral: it applies 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 or state of a system caused by an action or an exposure (see Figure 3). In consultation, the prior global condition of species and their 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 an opinion. Extending this baseline condition over time to form a *future without the project* condition (line **b** in the 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 project is implicit in almost every effects analysis.

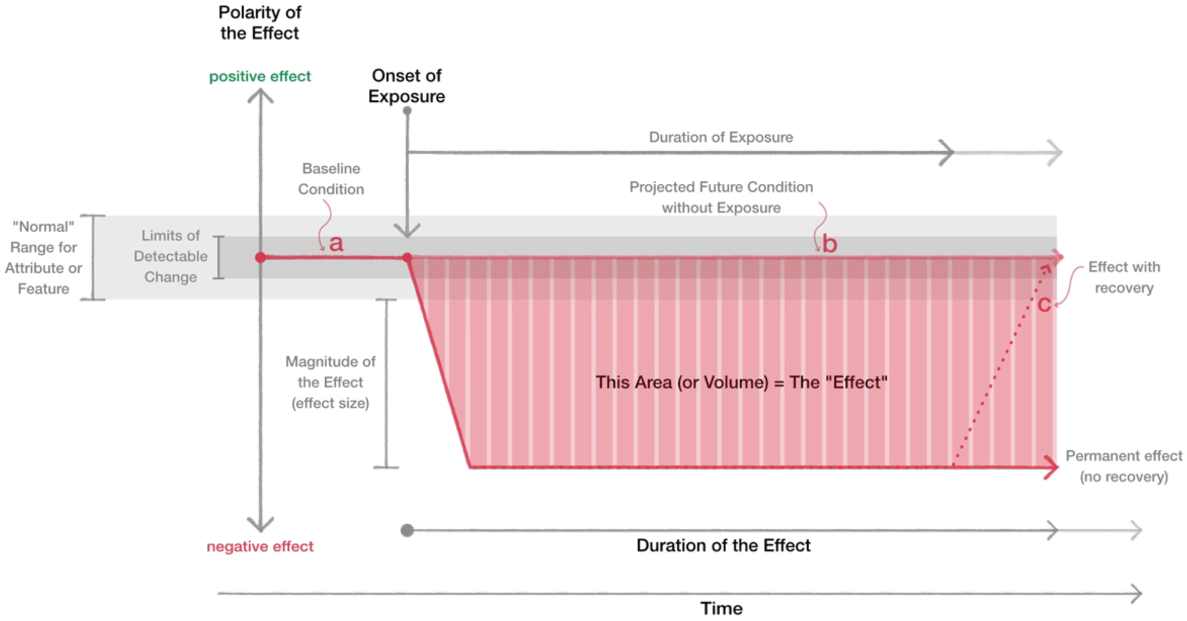


Figure 3. A schematic of the various elements encompassed by the word "effect." The vertical bars in the figure depict a series of annual "effects" (negative changes from a pre-existing or "baseline" condition) that are summed over time to estimate the action's full effect.

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 Figure 3) and *degree of recovery* (complete or partial). For instance, the recovery rate allows us to estimate how long it would take for a coral reef and associated benthic communities would take to recover.

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

2.1.1 Jeopardy Analyses

The Section 7 regulations define "jeopardize the continued existence of" as "to engage in an action that reasonably would be expected, directly or indirectly, *to reduce appreciably* the *likelihood of both the survival and recovery* of a listed species in the wild by reducing the *reproduction, numbers, or distribution* of that species" (50 CFR 402.02, emphasis added). 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 focus on changes in the *number of populations*, which provides the strongest evidence of a species' extinction risks or its probability of recovery. Jeopardy analyses also focus on 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 the same natural and anthropogenic stressors and whether some populations occur in protected areas or are at least protected from stressors that afflict other populations affect all of a species' populations.

To assess whether reductions in a species' reproduction, numbers, or distribution that are caused by an action measurably 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 focus on 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. Here the term "viability" refers to all of the factors that may cause populations (or species) to decline, collapse, and become extinct or avoid extinction (Burgman et al. 1993, Gilpin and Soule 1986) 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). 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.

Section 7(a)(2) requires us to insure that threatened or endangered species are not likely to become extinct in the wild and, instead, insure that they are likely to end up in the fourth category (survived and recovered). We fulfill that mandate, by studying data and other information on how and why species ended up in these four categories, identifying common patterns in the data, and using the knowledge those studies produce to inform our jeopardy determinations.

2.2 Application of this Approach in this Consultation

NMFS has identified several aspects of the United States WCPO purse seine fishery and fishing vessels that represent potential stressors to the environment, and threatened or endangered species or critical habitat that has been designated for them. 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 from the proposed action 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, such as listed species, or their prey;
- entanglement with FADs;
- derelict gear;
- introduction of oily discharges, cardboard, plastics, and other waste into marine waters; and
- vessel emissions.

2.2.1 Action Area

The *Action Area* includes all areas where the United States WCPO purse seine fishery vessels operate, including transiting, fishing, and transshipping. The *Action Area* shown in Figure 4 depicts the area where the fishery has operated since the mid to late 1970s to the present and where the fishery is expected to continue to operate. This area is equivalent to the WCPFC Convention Area between 11°N and 18°S. Specifically, the *Action Area* is a generally rectangular shaped area bounded on the west by 129°E longitude running from 11°N latitude to 18°S latitude (with 11°N and 18°S being the northern and southern boundary lines, respectively), and on the East by 150°W longitude down to 4° S latitude and then by 130°W longitude. A portion of the *Action Area* for the proposed action overlaps with the IATTC convention area east of 150°W as shown in Figure 4.

The fishery operates in the EEZs of the Pacific Island parties to the Treaty and that of the United States including the portions of the U.S. EEZ around American Samoa and the United States possessions of Howland, Baker, and Jarvis; as well as on the high seas in the WCPO. The portion of the United States EEZ around Jarvis is not available to fishing by the United States WCPO purse seine fishery. Figure 5 shows locations where purse seine fishing sets occurred in the *Action Area* from 2008 through 2016, and Figure 6 depicts the same information in two-year increments. In the figures, the logbook data are presented in terms of density and depict the average sets per square mile within a 5° × 5° radius of each 1° × 1° block for confidentiality purposes (NMFS 2017a).

During the 1997-2015 period, the fleet spent about 5% of its total annual effort in the United States EEZ, 19% on the high seas, and the remainder in the EEZs of the Pacific Island Parties. The percentages for any given year during that period ranged from 0 to 21% for the United States EEZ, 5% to 29% for the high seas, and 60% to 95% for the EEZs of the Pacific Island Parties.

American Samoa is the home port for many the vessels operating in this fishery and this port together with Majuro, Republic of Marshall Islands, and Pohnpei, Federated States of Micronesia, are the most common ports for offloading and transshipping, although ports throughout the *Action Area*, and several outside, are also used (Table 1). Therefore, the *Action Area* includes vessel transits routes between the fishing grounds and ports outside of the fishing area, including Bangkok, Thailand; Manta, Ecuador; La Union, El Salvador, Paita, Peru, and Mazanillo and Mazatlan, Mexico.

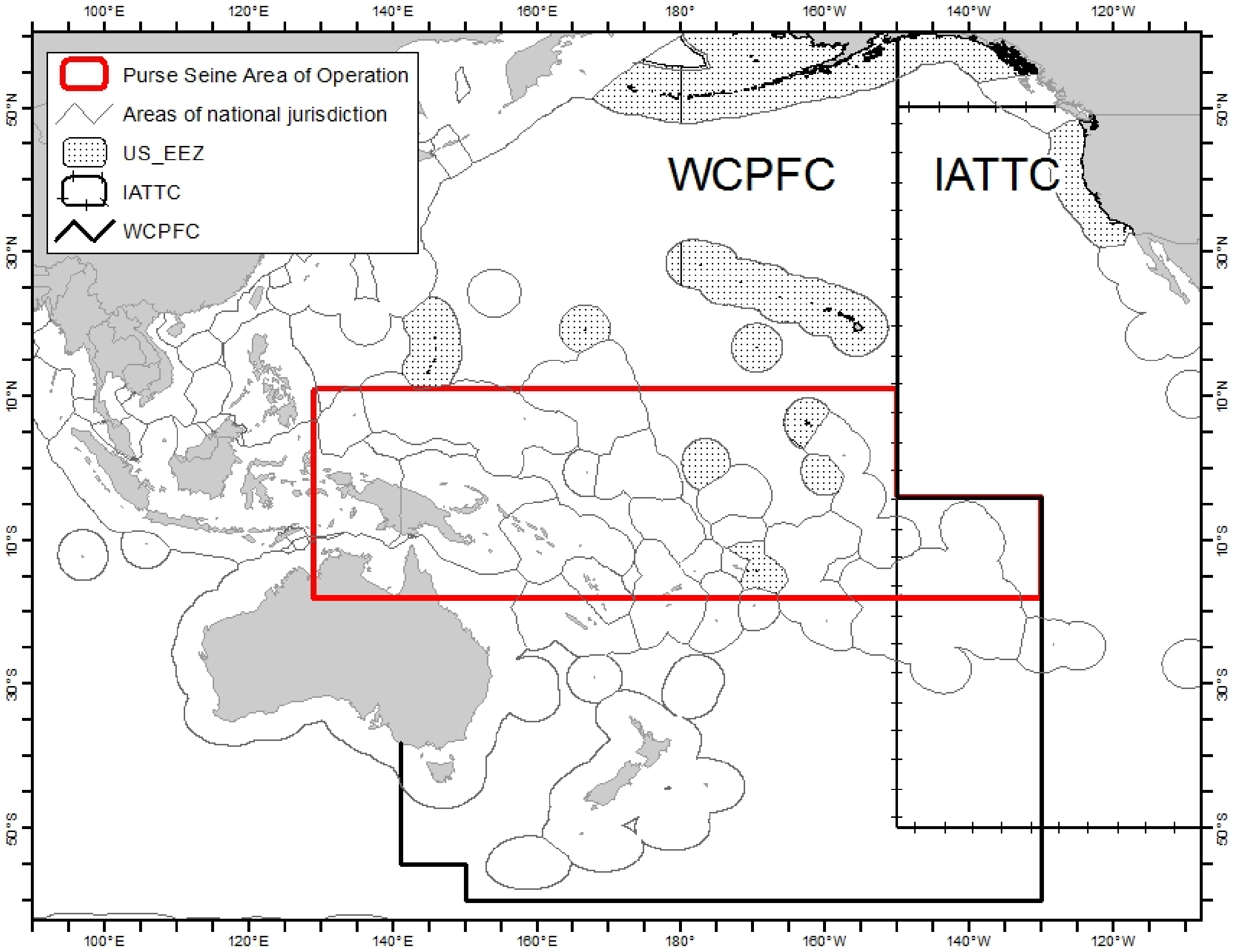


Figure 4. Fishery portion of the *Action Area* for the United States Western and Central Pacific Ocean purse seine fishery with national exclusive economic zones and regional fishery management organization (RFMO) boundaries depicted. The *Action Area* for the proposed action also includes transit routes from the fishery area to offloading ports outside of the fishing area including Bangkok, Thailand; Manta, Ecuador; La Union, El Salvador, Paita, Peru, and Mazanillo and Mazatlan, Mexico (see Table 1).

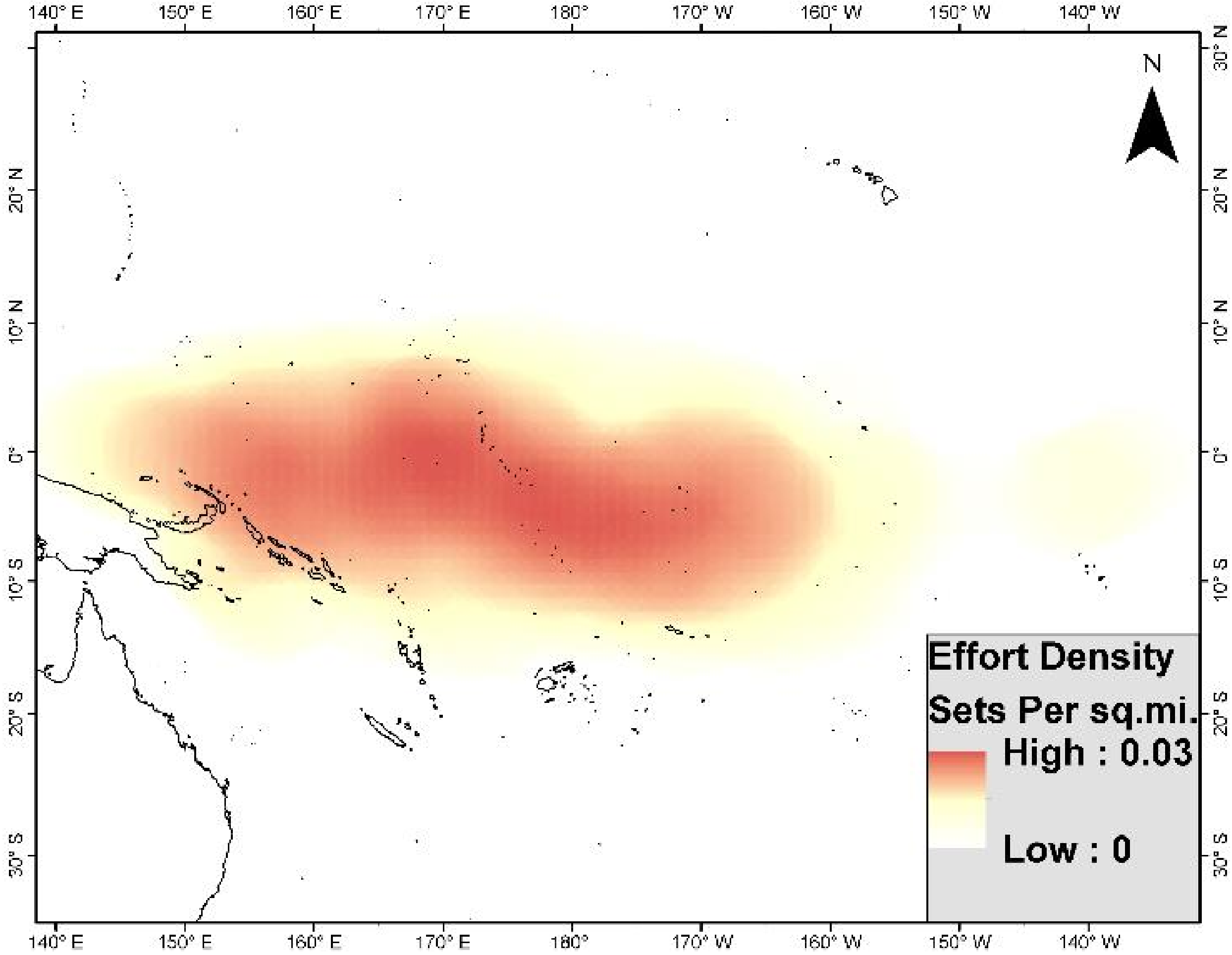


Figure 5. United States WCPO purse seine effort density from 2008-2016 from logbook data (NMFS 2017a).

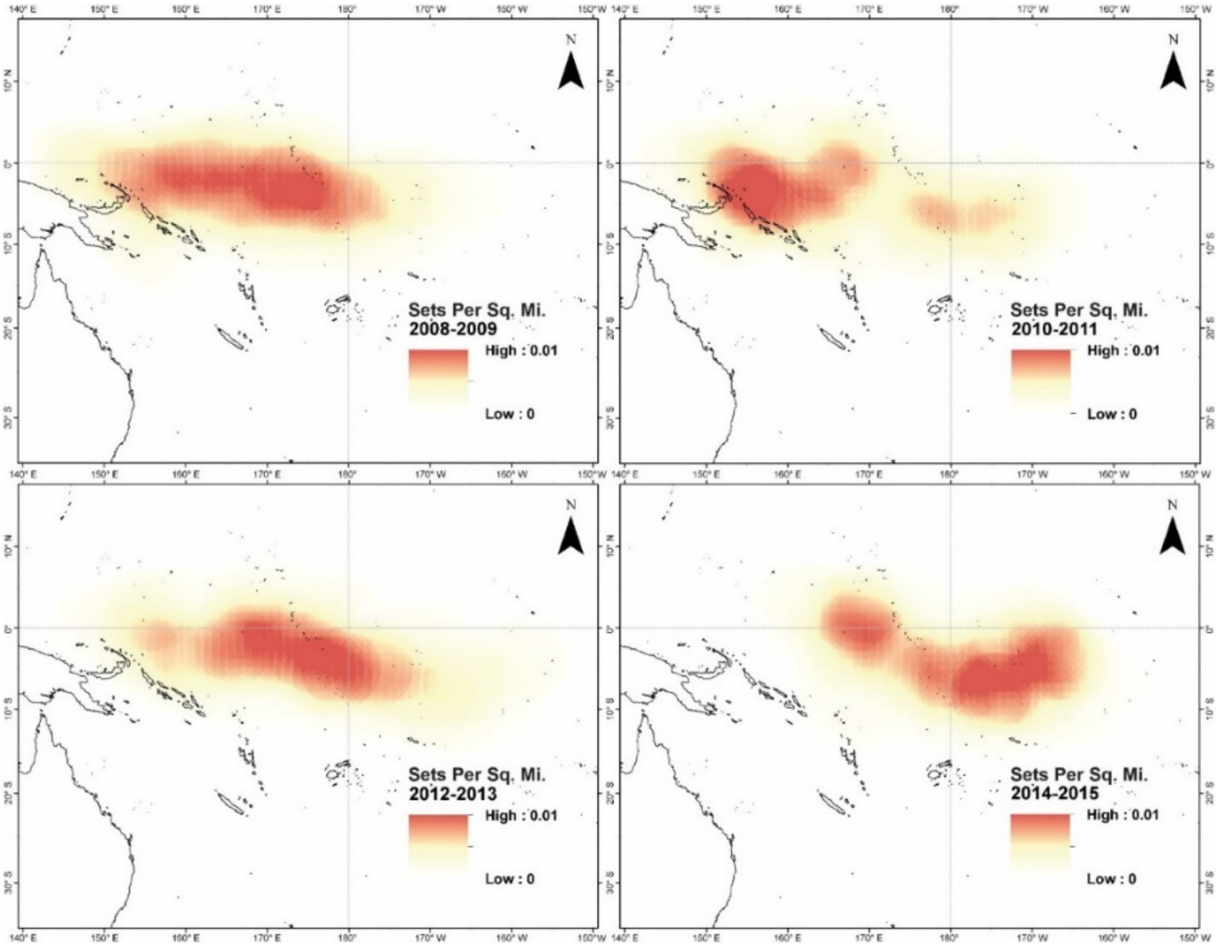


Figure 6. Effort density in 2 year increments from 2008-2015 from logbook data (NMFS 2017a).

As previously noted, from January 2018 through February 2021, 15 United States purse seine vessels exited the United States fishery, and no vessels entered. In April of 2021, two additional vessels left the fishery. Figure 7 shows the longitudinal distribution of sets for 2008-2018 (in the number of sets) for two exclusive components of the fleet: the vessels that have since left the fishery (i.e., primarily those that used the alternative business model), and those that remain.

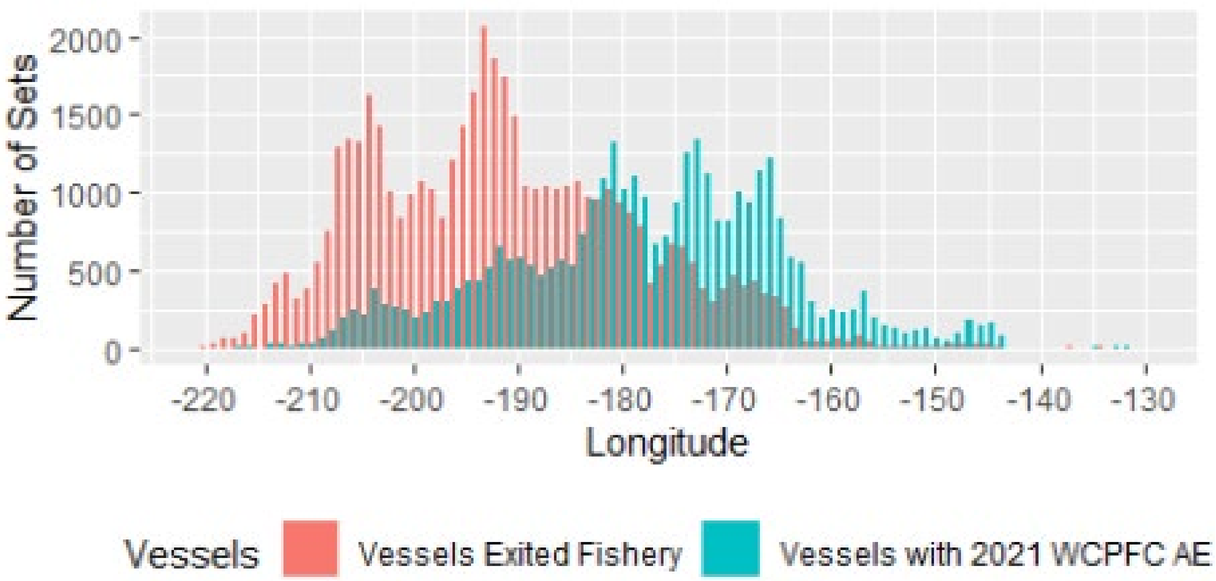


Figure 7. Number of sets by longitude for 2008-2018 for the United States WCPO purse seine fishery: vessels no longer in the fishery (without a WCPFC Area Endorsement as of March 2021); and vessels currently in the fishery (with a WCPFC Area Endorsement as of March 2021). Bin width is 1° longitude. On the horizontal axis, “-130” means 130°W, “-180” is the 180th meridian or antimeridian, and “-220” means 140°E.

Figure 8 shows the longitudinal distribution of sets (in the percent of all sets for the specific group) in the same period, for the entire fleet that operating during that period, and for just those vessels that have since left the fishery. However, it does not include the two vessels that left the fishery in April of 2021.

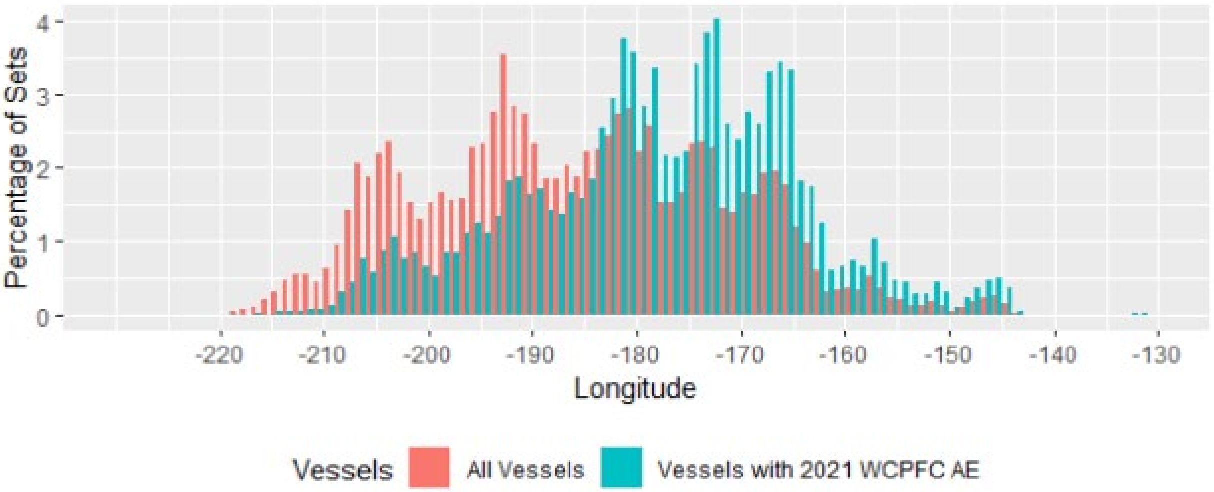


Figure 8. Percentage of sets by longitude for 2008-2018 for the United States WCPO purse seine fishery: all vessels (with a WCPFC Area Endorsement at any time in 2008-2018); and vessels currently in the fishery (with a WCPFC Area Endorsement as of March 2021). Bin width is 1° longitude. On the horizontal axis, "-130" means 130°W, "-180" is the 180th meridian or antimeridian, and "-220" means 140°E.

2.2.2 Approach to Evaluating Effects

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

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

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

We used available data to describe the United States WCPO purse seine fishery location and its stressors. Interactions by entrapment, entanglements, and landings represent the best data available on the United States WCPO purse seine fishery because it has been collected under 20% observer coverage prior to 2010, with 100% coverage afterwards. In contrast, no data are available to characterize exposure to vessel strikes or discharges of waste.

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., corals). We then sub-divided this information into four general categories: potential coastal exposures, potential pelagic exposures, observed coastal exposures, and observed pelagic exposures. In total, 36 listed species occur within the *Action Area* identified for this consultation.

We then evaluated the likelihood that each species would be exposed to the stressors described above. Where we concluded that that there is a low likelihood of exposure or that the potential for an adverse response is unlikely to result in adverse effects to listed species in the *Action Area*, we do not include them further in our exposure or response analyses. We present the basis for these determinations 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 United States WCPO purse seine fishery produce responses that range from likely exposed and not likely adversely affected – perhaps, being encircled and then released alive unharmed; landed on deck through brailing process 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 (Hall and Roman 2013).

Figure 9 presents our conceptual model of how we translate an interaction between an animal's exposure and the United States WCPO purse seine 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 United States WCPO purse seine 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.

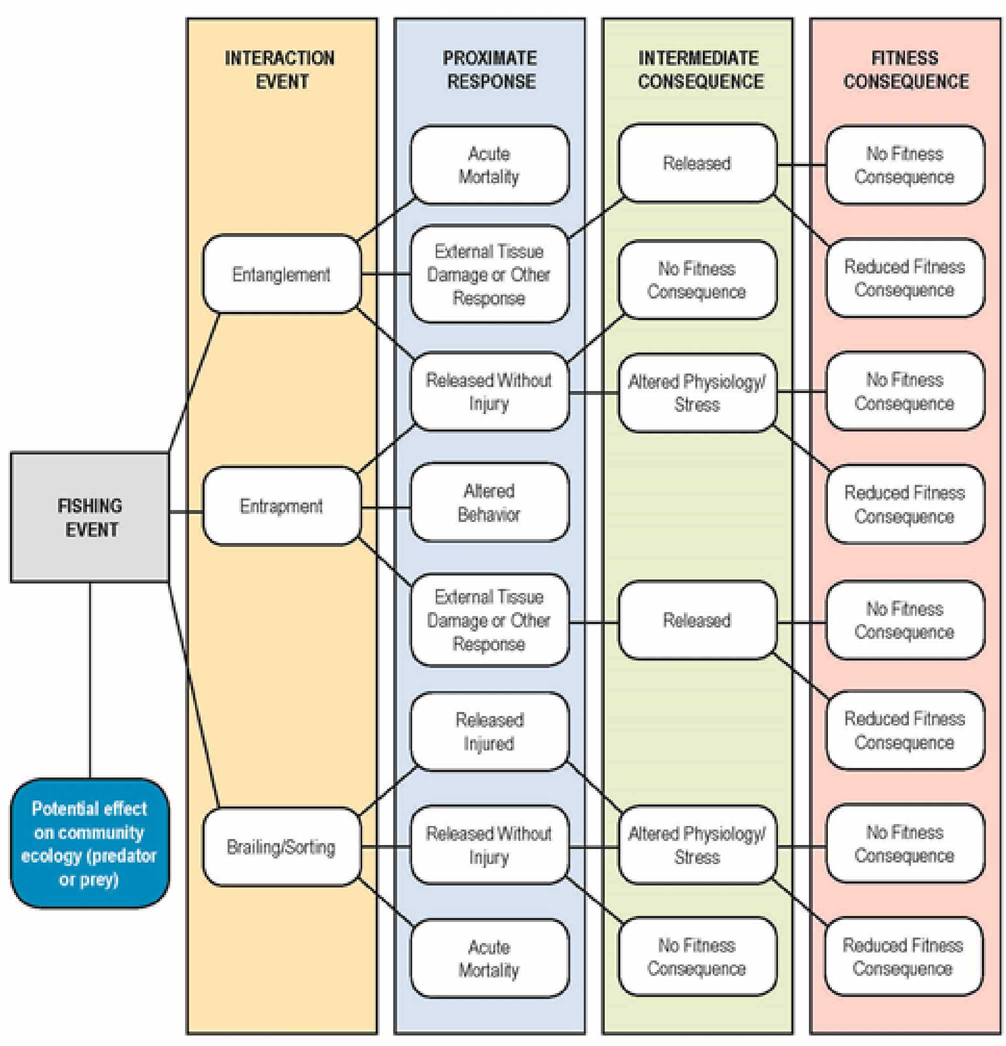


Figure 9. Conceptual model of the potential responses of an animal's exposure in the United States WCPO purse seine fishery and the pathways between exposure (the interaction event the individual animal has with the fishing gear), responses, and potential fitness consequences to the individual.

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. We captured the relevant life history information for each listed species that interacts with the United States WCPO purse seine fishery as causal loop diagram, which allows us to: (a) visualize a species life history in a way that reveals the main variables that promote population growth (or decline); (b) explicitly identify the various stressors that are known to act on different life history stages; (c) identify the probable consequences of those stressors on those stages; (d) estimate how long an effect might take to be detected in census data; and (e) infer the probable effect on a species' pattern of growth or decline.

Specifically, causal loop diagrams allow us to visually display the negative and positive feedback loops within the stressor-response relationships and the interactions between the variables that comprise the model (e.g., assessment endpoints like nesting success or abundance). The causal loop diagram depicts the direction of effect of one variable on another, and whether the effect of the variable increases, decreases, or remains unchanged (see Puccia and Levins 1985, and others for more on causal loop models). Arrows in these diagrams (see Figure 10 for an example) represent links between two variables; the sign associated with these arrows represents the type of behavior the path will produce (whether the effect is positive, negative, or neutral). An arrow with a positive sign (+) means the two components move in the same direction (or the following variable enhances the effect of the preceding variable), whereas an arrow denoted with a negative sign (-) means that the two components move in opposite directions (that there is a negative or decreasing effect from one variable to the next). An arrow path that starts and returns to the same starting point is a "loop" and illustrates that there is some self-effecting feedback occurring.

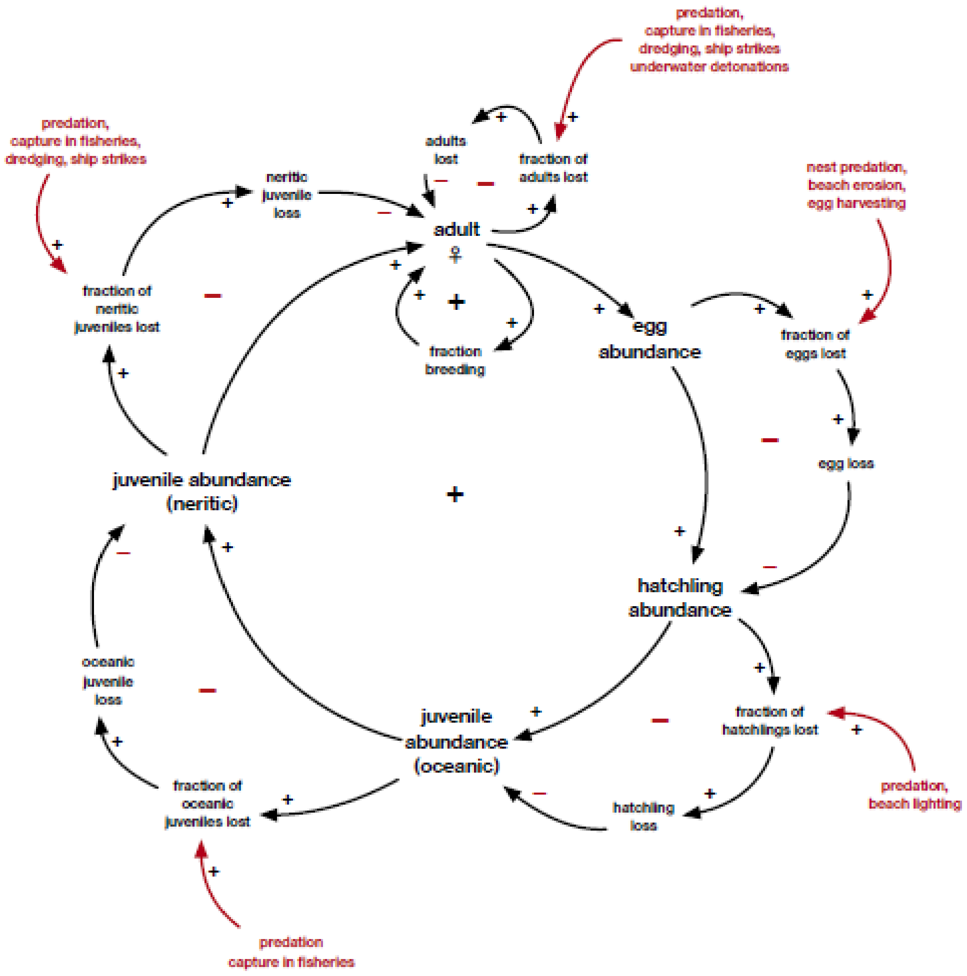
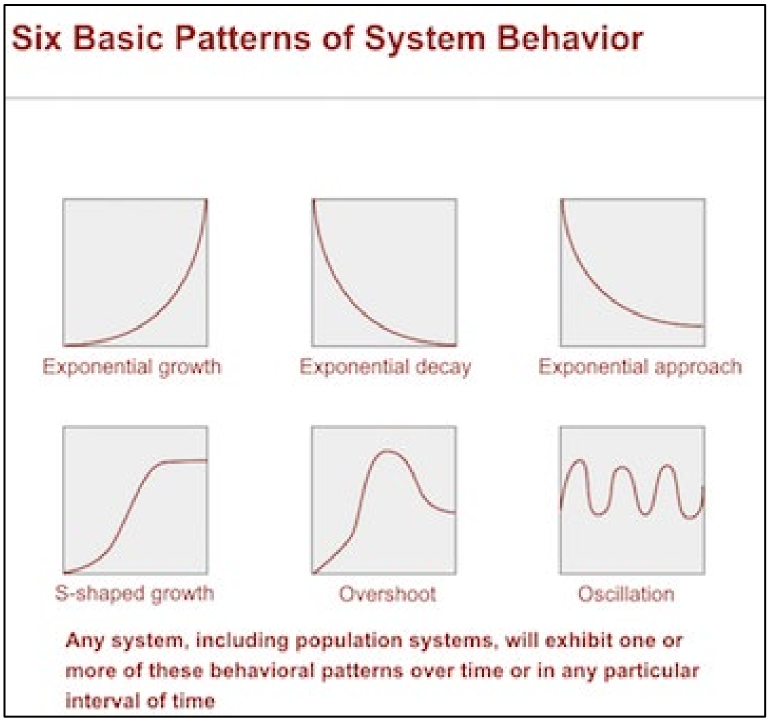


Figure 10. Example causal loop model for loggerhead sea turtles.

A causal loop is either reinforcing (R) – it reinforces change with more change (amplifying), or balancing (B) - it reaches a plateau (has a stabilizing effect). That is, when the pathways between one variable to the next are traced around the loop all the way back to the original variable, the loop either reinforces (creates growth or collapses) the original variable, or causal influences in the loop keep things in equilibrium or balance the system. Thus, the causal loop model is designed to capture the life cycle of the listed species and their habitats, and illustrate the effects of environmental or stressor changes on the variables that influence a species survival and resilience over time, which includes the stressors associated with the proposed action. The model does not require us to assume that animals exist in pristine environments; in those circumstances in which animals are regularly or chronically confronted with stress regimes that would require them to engage in sub-optimal behavior, or adapt to sub-optimal conditions, we assume that a change to their exposure from a particular stressor or stress regime would contribute to conditions that are even further from optimal for their success.

While our models do not try to reflect the whole system, we try to capture the major factors that have the greatest influence on a particular species survival and recovery. As part of this process, we tried to identify what, if any, variables are associated with time lags or identify where time lags can develop or accumulate in the system because they are often a source of imbalance in the feedback structure. In this case, we would be particularly concerned about variables that create a source of imbalance in a population's ecology or the processes that drive a species survival and recovery.

According to system dynamics, each loop in the diagram will likely correspond to one of six common dynamic patterns of behavior (Figure 11). Where possible, we use reference data to discern the pattern of loop behavior and describe the specific problem under consideration. For instance, we used nesting aggregation data for loggerhead sea turtles and leatherback sea turtles to understand the nesting patterns in these species as an indication of their population viability. Reference patterns are important because the aggregation of individual patterns predict the behavior of the higher-level system. We included a graphic to represent the behavior of the reference variables (data) that influenced the behavior of that particular loop. The aggregated causal loop diagram ultimately presents the overall pattern of behavior of the system, or in this case, the species. Causal loop models are used widely in system dynamics methods modeling for ecology, health care, and business decision-making to simulate system behavior and we have adapted these concepts for this consultation.



Six Basic Patterns of System Behavior

Any system, including population systems, will exhibit one or more of these behavioral patterns over time or in any particular interval of time

Figure 11. Six common patterns of system behavior (recreated from Ford 2009).

Importantly, causal loop diagrams are a means to an end – we used causal loop diagrams as a point of discussion within our analytical team to achieve methodical conceptualization, elicit expert input, engage in scientific exchange, and establish a setting for collective learning, dialogue and inquiry. Where we could, we included in our causal loop diagram a graphic to represent the behavior of influential data and how it affected a particular loop. However, the point of the causal loop diagram was to aid us in deciphering how the species' life history, and the myriad of threats faced by the species, play upon the species chances of survival and recovery including how the proposed action effects the species chances of persistence. Creating the causal loop diagram became a means to an end to address this multidimensional complex problem in an interactive team setting where we exchanged observations, points of view, and mental models to be certain we examined the parts and the whole, and the interdependencies between the parts and the whole so that we could be sure we adequately evaluated each species and their associated risk, and see what, if anything, we were inadvertently leaving out as we conducted our analyses. In this respect, the causal loop diagram helped us focus our analyses and our work as a team to complete this consultation.

Next, we analyzed historic interactions rates, including the severity of those interactions, to inform our estimation of probable future interactions. The analysis presented in the biological assessment (NMFS 2017a) used data from 2008 to 2015 and was largely missing information on scalloped hammerhead sharks and giant manta rays. We subsequently received observer data up to 2018 and new information on sharks and ray captures. In addition, the analysis in NMFS (2017a), while it did incorporate percent of FAD sets as a covariate in the model, presented interaction rates that only differentiated between 'landed' and 'unlanded' interactions and we considered it important to determine interaction rates separately for FAD sets, unassociated sets, and direct interactions with FADs to understand the impact of FAD use on ESA-listed species.

Thus, we conducted a new analysis on the observer data to determine interaction rates and we decided to follow the approach of Martin et al. (2015) to conduct these analyses. Using this approach, we were able to separate the interactions as described above and also assess the anticipated condition of the individual animal using the fisheries condition codes. From there, we developed mortality estimates for both unobserved historic interactions and anticipated future interactions. This method produces a single interaction rate in terms of numbers of individuals caught per set for each species and set or interaction type and assumes that this interaction rate will be applicable into the future regardless of any changes in the fishery or changes in species abundance. We acknowledge that this is a limitation of the approach, however, this is a robust statistical analysis incorporating available data on observed captures and we anticipate that the results will be representative of interaction rates in the near future (10 to 15 years). NMFS will monitor observer data and, as feasible, population abundances, to reassess if the interaction rates produced by this analysis continue to be valid. Below we detail the steps for those analyses and estimations.

2.2.2.1 Data Sources and Determination of an Interaction

We obtained observer data for the period 2008-2018 from IFD, which received it from the WCPFC. Observers in the fishery are deployed by the Pacific Islands Forum Fisheries Agency (FFA) observer program. The FFA observer data are stored and maintained by the WCPFC. We obtained the observer data we used in this analysis on June 19, 2019. 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.

We reviewed all records for ESA-listed species across the 2008-2018 observer data from these data sources and scored each as to whether it constituted an interaction with the fishing operations or was a sighting off in the distance. We included all records scored as an interaction in our exposure analysis. In almost all cases, species observations involving a fishing set were considered interactions for our analysis. There were only two rationales for not including an observer record involving a fishing set: (1) records that were duplicated, presumably through data entry error; or, (2) records where the observation occurred outside of the *Action Area* for this consultation. Interactions observed during fishing sets occurring in the eastern Pacific Ocean outside of the WCPFC area – East of 150°W longitude when North of 4°S latitude or East of 130°W longitude when South of 4°S latitude – are part of a separate federal action, authorization of the EPO fishery. In this consultation, the effects of the EPO fishery that occur in the *Action Area* are addressed in the *Environmental Baseline* of this biological opinion.

Records collected outside of fishing sets, most commonly sightings of animals by the observer, were reviewed but not considered interactions in this analysis. Most of these sighting records were marine mammals. We reviewed observer narratives to determine if an interaction occurred. Based on observer records, there were some instances where whales were driven away from a school of fish to allow for a set unimpeded by the whale. As this is not a lawful activity we did not consider these as effects of the proposed action.

2.2.2.2 Release Condition

As noted above, the stressors associated with the United States WCPO purse seine fishery produce responses that range from behavioral impacts (i.e. from presence of the vessel) to more stressful impacts such as from being encircled by the net and then released alive unharmed, to injurious impacts that stem from being landed on deck through brailing process and released injured, to death (immediate, or later in time following injury). For observed interactions with the United States WCPO purse seine fishery, observers recorded the release condition of animals at the conclusion of interactions. For records during fishing sets, the animal's condition at the start and end of the interaction as well as ultimate fate were generally recorded; although condition records were often missing, especially for species that were not considered species of special interest by the SPC at the time, such as sharks and mantas. We used condition codes at the end of the interaction for our analysis where they existed in the data. Condition codes in the observer data included:

A0 – Alive, condition unknown;

A1 – Alive and healthy;

A2 – Alive, but injured or distressed;

A3 – Alive, but unlikely to live;

A4 – Alive, entangled in the net, released untangled;

D – Dead;

U – Unknown.

2.2.2.3 Set Type Classification

There are multiple pathways for interactions with the United States WCPO purse seine fishery. The fishery uses FADs on a proportion of their sets; the use of FADs may increase the likelihood of interactions with some ESA-listed species. The FADs themselves can also pose entanglement risks. For each interaction, we captured the type of purse seine set that had occurred from the observer data as either 'free' or 'fish aggregating device' sets, distinguishing if the school of tuna set on were associated with a FAD or not. For most sets, the observer recorded set type. A small subset of purse seine sets lack a description of the set type; interactions occurring during these sets were categorized as having 'unknown' set type. We know the total number of sets that occurred annually based on logbook records from vessel captains, and we subtracted the number of observed sets from the total sets recorded by captains to determine the number of sets where the type was unknown.

We also had interactions meeting the interaction criteria without a set. These were limited to interactions associated with FADs that were not being set on. We termed interactions associated with FADs that were not being set on were coded as set type 'FAD interaction'.

2.2.2.4 Classification of Unobserved Interactions

Observers did not consistently record two ESA-listed species, scalloped hammerhead sharks and giant manta rays, across the whole observer dataset. Scalloped hammerhead sharks were not recorded until 2012 (i.e. there were no observed sets for 2008 – 2011). Similarly, giant manta rays were not recorded until 2011. In both of these situations, we considered all sets for years were these species were not recorded to have been unobserved for our analysis.

All data records constituting interactions were summarized by year, species, set type, and condition and used in combination with estimates stemming from our interaction rate analysis to determine the most likely numbers of unobserved interactions.

2.2.2.5 Classification of Unknown Condition or Unknown Species

We used the proportions of the known conditions for a given species across all interactions in the 2008-2018 observer data to assign a condition to those with unknown condition codes for our analysis. We calculated the proportion of each condition code, excluding condition code U, for each species. We then multiplied these proportions by the number of condition code U individuals of that species to assign a condition code for analysis. As was the case for proportions calculated for partitioning unidentified species, we did not consider the uncertainty in calculations of the proportions of condition codes.

The condition of those animals we estimated as unobserved, in addition to estimates of future interactions, is also unknown. We used the species-specific observed condition code proportions to assign condition to these estimated interactions.

Observers occasionally are unable to identify observed animals to species, due to distance to the animal, restricted visibility, or other reasons. In all cases, observers classify unidentified species to higher taxonomic categories. We reviewed all of these ‘unidentified categories’ in the observer data and identified all that could potentially contain ESA-listed species.

For each of the unidentified categories that could contain ESA-listed species, we completed the same analysis as described previously. We reviewed each observer record and applied to same criteria to score them as interactions for further analysis. For each unidentified category, we then applied the same interaction rate analysis to estimate the number of unobserved and future interactions would be expected to occur given their occurrence in the observer data.

To attribute both the unidentified category interaction observations and the estimates of the number of unobserved and future interactions of these unidentified categories to ESA-listed species, we used a simple proportional approach. We first identified all ESA-listed and unlisted species in the observer data that could occur within each unidentified category. We then calculated the direct proportion each of the ESA-listed species that was of a particular unidentified category using:

$$Prop_i = {Int_i}\Big/\left(\sum_{i=1}^{n} Int_i + \sum_{j=1}^{m} Int_j\right)$$

Where $Prop_i$ is the proportion ESA-listed species i is of a given unidentified category, Int_i is the total number of interactions scored as captures of ESA-listed species i, Int_j is the total number of interactions during purse seine sets for unlisted species j that is a member of that unidentified category, and n and m are the number of ESA-listed and unlisted species that are constituents of a given unidentified category, respectively. These proportion calculations include all interactions across the full observer data set we used (2008-2018).

In the case of the giant manta ray, seven potential unidentified categories could potentially have unidentified giant manta rays scored to them. Based on this large number, we contacted the SPC, for clarification on which unidentified categories to use based on their observer training protocols. The SPC responded that only one of the potential unidentified categories specific to rays: ‘unidentified manta,’ should be used for our analysis. To ensure accuracy and verify this guidance from SPC, we examined observer records for the ‘sharks, rays and skates’ unidentified category for giant manta rays. Our review of observer records uncovered one interaction in this category over the nine years of data and including it amounted to less than 0.5 of additional giant manta ray interactions per year and therefore did not affect our results when considered.

We multiplied the number of individuals observed and estimated for unobserved interactions for the applicable unidentified category by $Prop_i$ to estimate the number of species i expected to be from that unidentified category. We added these species-specific estimates to both observed and previously estimated numbers based on identified species to estimate total observed and unobserved interactions each year and expected future interactions by species.

Finally, we assigned fate to the proportion of each unidentified category estimate that was applied to a given ESA-listed species following the methods described previously; where the proportions of identified fates recorded from that specific unidentified category were applied.

2.2.2.6 Use of Bayesian Inference to Estimate Annual Interactions of Eleven Protected Species

Once the observer data were organized into the classes described above (‘free’, ‘fish aggregating device’, ‘FAD interaction’; and assignment of unidentified condition or species) and observed numbers of captures per year for each class were calculated, we employed a Bayesian Markov Chain Monte Carlo method (Martin et al. 2015) to account for unobserved or unreported captures to estimate annual interaction rates. Interaction rates were estimated for each class and combined as total interactions per year for the following species: loggerhead sea turtles, leatherback sea turtles, green sea turtles, hawksbill sea turtle, olive ridley sea turtles, sei whales, sperm whales, oceanic white tip sharks, scalloped hammerhead sharks, and giant manta rays.

We used the Bayesian analysis approach of Martin et al. (2015) to estimate interaction rates in the fishery. Briefly, the method assumes that there is a consistent and species-specific interaction rate coefficient describing the expected number of interactions at a given fishing effort level, equivalent to saying there is a linear relationship between the amount of effort and the number of expected interactions by species:

$$Int_i = \theta_i \cdot E_i$$

Where Int_i is the number of interactions occurring with species i, theta (θ_i) is the interaction rate coefficient for species i, and E_i is the measure of effort that could result in an interaction with species i. Effort in this fishery is most often the number of purse seine sets. The method estimates the posterior probability distribution of θ_i from annual total observed numbers of interactions (Int_i) by species and fishing effort (E_i) using the 2008 – 2018 observer data. The model assumes the observations of interactions occur following a Poisson distribution with an uninformative gamma prior for rarely encountered species and a normal distribution with uninformative mean (normal) and precision (gamma) priors for more commonly encountered species. More technical details related to the model and model selection are provided in NMFS (2019c).

For each ESA species considered, five specific classes of interactions were evaluated in our analysis. There were three classes of interactions associated with purse seine sets:

- interactions during free-school sets;
- interactions during FAD sets; and
- interactions during sets where the set type was undefined in the observer data.

There were also two classes of interactions that occurred during fishing activities other than setting:

- non-set interactions, largely due to the presence or movements of the fishing vessel, the vessel's launches, or the vessel's helicopter; and
- FAD interactions, where an interaction with a FAD occurred independent of a set.

We estimated the interaction rate, theta, for these interaction classes for each of the 16 ESA-listed species, unless there were no observed interactions associated with a particular interaction class. Free-school and FAD interactions were analyzed relative to the number of observed free-school and FAD sets, respectively. When set type was undefined, we analyzed interaction rates relative to the total number of observed sets. We also analyzed both non-set and FAD interactions relative to total observed sets.

Total observed sets may not be an ideal scalar for estimation of rates for non-set or FAD interactions. For the rate of non-set interactions, perhaps a better scalar would be the total vessel activity of the fleet, such as total days fishing. However, the observer data only records non-set interactions that occurred during observed trips. We argue that total observed sets is an easily obtained proxy for fishing activity during observed trips and thus the appropriate scalar in this case. A similar argument applies to FAD interactions rate estimates: the rate of FAD interactions scaled to total observed sets relates to the fishing effort during observed trips.

We considered alternatives to total observed sets, particularly for FAD interactions, such as estimates of the number of deployed FADs as a proxy for the number of functioning FADs each year. If known precisely, the number of functioning FADs would be a good scalar for observed FAD interactions. However once deployed, FADs can continue to drift for many months, making any measure of deployments a potentially poor measure of the number of FADs. Because of these problems, estimates of FAD deployments from either the observer data or independent estimates (Escalle et al. 2019) may not be well correlated with the number of functioning FADs deployed by the United States WCPO purse seine fleet and that are in the water in a given year, which would introduce considerable bias into estimation of FAD interaction rates. The total number of sets observed is known precisely, it relates the amount of fishing effort that is occurring when FAD interactions are being observed in the fishery, and it is an easily obtainable metric for later estimation of unobserved or future interactions using our interaction rates estimates. Therefore, the FAD interaction rates estimated in our analysis represent a minimum interaction rate as it does not account for all United States fleet deployed FADs, but only those that are investigated by the vessel.

We combined set interactions lacking set type information and non-set interactions because they both use the total sets as a scalar. We considered combining FAD interactions with these other two analysis classes as well, but to preserve the ability to separately track FAD interactions, they were analyzed separately.

The estimates of the interaction rates, theta, across these different interaction classes were used in two ways in our analysis. Firstly, they were used to estimate the number of interactions that occurred in fishing activities that were not observed, or for which we did not have observer data in the 2008 – 2018 data set. Second, we used these interaction rates to estimate what the potential future impact of the fishery on ESA-listed species would be given anticipated numbers of FAD, free school, and total sets in future years. The species-specific estimates of interaction rates across all interaction classes were summed to generate species-specific interaction rate estimates for both past unobserved and future interactions. We projected future fishing effort as advised by IFD (see Section 1.2 – *Description of the Proposed Action*). It is anticipated that up to 3,100 sets will be made each year, with up to 1,581 of them being FAD sets, for the reasonably foreseeable future, and we use this effort level – in combination with our estimates of interaction rates – to estimate future ESA-species interactions. We used this effort level in combination with our estimates of interaction rates to estimate the anticipated future numbers of annual interactions for the 16 ESA-listed species.

In comparing observed interaction rates, calculated as the number of observed interactions divided by the number of observed sets, to the output from the Bayesian inference model used to estimated anticipated captures, we discovered that the credible intervals produced by the model were too narrow. For example, for oceanic whitetip sharks captured on FAD sets, the 5th and 95th percentiles for the interaction rates estimated by the model were 0.030 and 0.050 with a mean of 0.040. The observed interaction rates from 2008 to 2018, calculated as described above, ranged from 0.027 to 0.19, making five of the 11 years greater than the estimated 95th percentile. As a second example for a species with much lower interaction rates, hawksbill sea turtles, on free/unassociated sets, the modeled interaction rates had a mean of 0.00125 with 5th and 95th percentiles of 0.00088 to 0.00166. The observed interaction rates ranged from 0.00 to 0.0058, resulting in four of the 11 years having values greater than the 95th percentile from the Bayesian inference model.

As an alternative approach, we used the statistical software package R to conduct nonparametric bootstraps of the observed interaction rates, conducting separate analysis for FAD, free, and unknown sets for all species as well as FAD interactions for sea turtles. The non-parametric bootstrap is used to estimate a parameter or parameters of a population or probability distribution from a set of observations where we don't wish to make a guess of the distributional form (e.g. Normal, Gamma, log-normal). It is conducted by repeated resampling from the original observed data. The frequency distribution of the resulting sampled data can then be assessed to estimated population parameters. For each species and set/interaction type, we created 10,000 bootstrap samples with replacement. We then found the 95th percentile for the resulting samples.

2.2.2.7 Defining Future Anticipated Captures

In order to assess the impact of fishery captures on ESA-listed species, we need to determine an appropriate metric. The 95th percentile value from the Bayesian inference model described above is representative of the highest level of captures we would expect in any one year, but may overestimate both the expected number of captures and the potential impact of the fishery on species/population(s). The mean value from the Bayesian inference model estimates captures, over time, and represents expected number of captures and impacts of the fishery more accurately. However, in any given year, there is a 50% chance of captures being greater than the mean, thereby rendering the mean, in and of itself, not a useful metric for describing anticipated annual captures and may underestimate the impacts of the fishery.

In order to more closely approximate actual anticipated captures by a fishery but allow for years of higher-than-average number of interactions, we use the maximum 5-year running average of the observed interaction rates from the historic time series. By applying running averages to historic interaction rates, we can allow for higher years of captures, while at the same time these higher years of capture are smoothed by years of mean and lower numbers of captures. In order to only capture more recent fishery activity and reliable species identifications, we limited this analysis to the 2010 to 2018 data. In the sections below we report the maximum 5-year running averages and running sums calculated as described above along with the 1-year mean and 95th percentile values calculated from the Bayesian inference model described in Section 2.2.2.6. Below we specify how the maximum 5-year running averages were calculated:

1. For each species, year and set type (FAD, free or unknown), calculate an interaction rate by dividing observed captures by number of observed sets in the same year. For unknown set type we used total observed sets.
2. Calculate 5-year running averages from 2010 to 2018 on the resulting time series of interaction rates and find the maximum value of those running averages.
3. Estimate the number of captures for each resulting maximum 5-year running average interaction rate by multiplying the interaction rate by the anticipated number of annual sets (3,100 total, 1,581 FAD sets, and 1,519 free sets).
4. Sum across all set types for each species to estimate total anticipated captures represented by the maximum 5-year running average. Multiplying this value by five gives the maximum 5-year running sum.

2.2.3 Tipping Points

The concept of "tipping points," arose in a series of court opinions on prior biological opinions (e.g. Oceana, Inc. v. National Marine Fisheries Service, 705 Fed. Appx. 577 (9th Cir. 2017); Wild Fish Conservancy v. Salazar, 628 F.3d 513 (9th Cir. 2010); National Wildlife Federation v. National Marine Fisheries Service, 524 F.3d 917, (9th Cir. 2008)). In the Wild Fish Conservancy case, the Ninth Circuit Court of Appeals concluded that FWS' argument ignored the recovery component of the jeopardy standard because it failed to recognize that ""even before a population is extinguished, it may reach a point at which it is no longer recoverable: 'a species can often cling to survival even when recovery is far out of reach.' Nat'l Wildlife Fed., 524 F.3d at 931. The Service has not determined when the tipping point precluding recovery of the Icicle Creek bull trout population is likely to be reached, nor necessarily, whether it will be reached as a result of the 2006-2011 operations and maintenance of the Hatchery." Wild Fish Conservancy, 628 F.3d at 527 (quoting Nat'l Wildlife Fed., 524 F.3d at 931). The Service has not determined when the tipping point precluding recovery of the Icicle Creek bull trout population is likely to be reached, nor, necessarily, whether it will be reached as a result of the 2006-2011 operations and maintenance of the Hatchery" (Wild Fish Conservancy 628 F.3d at 527, *emphasis added*). The biological opinion implicated in that case argued (in part) that a jeopardy determination would only be appropriate when an action would both: (1) reduce appreciably the likelihood of survival; and (2) reduce appreciably the likelihood of recovery. However, as noted in the updated 2018 regulations (84 CFR 44976), neither the ESA nor our regulations state any requirement for the Services to identify a "tipping point" or recovery benchmark for making Section 7(a)(2) determinations. Rather, the Services have discretion as to how we will determine whether the statutory prohibition on jeopardy or destruction or adverse modification is exceeded. We also note in the 2019 regulations, that the state of science often does not allow the Services to identify a "tipping point" for species.

2.2.4 Climate Change

Future climate will depend on warming caused by past anthropogenic emissions, future anthropogenic emissions and natural climate variability. NMFS' policy (NMFS 2016) is to use climate indicator values projected under the Intergovernmental Panel on Climate Change

(IPCC)'s Representative Concentration Pathway (RCP) 8.5 when data are available or best available science that is as consistent as possible with RCP 8.5. RCP 8.5, like the other RCPs, were produced from integrated assessment models and the published literature; RCP 8.5 is a high pathway for which radiative forcing reaches >8.5 W/m^2 by 2100 (relative to pre-industrial values) and continues to rise for some amount of time. A few projected global values under RCP 8.5 are noted in Table 2. Presently, the IPCC predicts that climate-related risks for natural and human systems are higher for global warming of 1.5 ºC but lower than the 2 ºC presented in Table 2 (IPCC 2018). Changes in parameters will not be uniform, and IPCC projects that areas like the equatorial Pacific will likely experience an increase in annual mean precipitation under scenario 8.5, whereas other mid-latitude and subtropical dry regions will likely experience decreases in mean precipitation. Sea level rise is expected to continue to rise well beyond 2100 and while the magnitude and rate depends upon emissions pathways, low-lying coastal areas, deltas, and small islands will be at greater risk (IPCC 2018).

Table 2. Projections for certain climate parameters under Representative Concentration Pathway 8.5 (values from IPCC 2014).

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

Given the limited data available on 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 Hawaii-based shallow-set longline fishery (SSLL), we've learned a few key important lessons: the climate based model incorporating fixed age (lag) is unrealistic given variability in 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 causal loop diagrams and 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 affect 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.

2.2.4.1 Statistical Intervals 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. number of expected 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)). When the available data contain substantial uncertainty, Congress directed the Services to provide the "benefit of the doubt" to the species concerned (H.R. Conference Report No. 697, 96th Congress, Second Session 12 (1979)).

For the purpose of ensuring that the action satisfies the requirements of Section 7(a)(2) of the ESA, our assessment uses 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. We use the mean, the maximum 5-year running average, and the 95th percentile estimates in our assessment. The mean values represent the number of interactions that we expect in an average year, and provides the most reliable estimate of the expected number of interactions between the fishery and the listed species over an extended time frame. However, in some years higher or lower numbers of interactions may occur. The maximum 5-year running average captures these higher years by considering average interactions over shorter timeframes and identifying the timeframe with the highest running average. The upper 95th percentile values represents our upper estimate of interactions in any given year. By selecting the 95th percentile, we can be confident that there is a 95% probability that, given the data, the true population parameter is contained within the credible interval. In terms of the number of interactions, this means that there is a 95% probability that the true number of animals captured or killed is within the credible interval.

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

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

2.2.5 Evidence Available for this Consultation

We used the following procedure to ensure that this consultation complies with NMFS' requirement to consider and use the best scientific and commercial data available. We started with the data and other information contained in IFD's 2017 *Biological Evaluation on the United States WCPO purse seine Fishery*, NMFS marine mammal stock assessment reports, available recovery plans for affected species, and the Bycatch Management Information System (BMIS). We supplemented this information by conducting 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 *Aquatic Sciences and Fisheries Abstracts*, *First Search*, *Science Direct, BioOne, Conference Papers Index, JSTOR, Google Scholar, Web of Science, the Western and Central Pacific Fisheries Commission's* *publications library, Marine Biology, Marine Ecology Progress Series, Marine Pollution Bulletin, Copeia, Public Library of Science - Biology (PLoS Biology), and Public Library of Science - One (PLoS One)*. We recognize this is not an exhaustive list of all resources that were referenced.

For our literature searches, we used paired combinations of the keywords: "purse seine," "sea turtles," "Pacific," taxonomic orders for the pertinent species under consideration like "*Dermochelys*," "*Caretta*," "*Chelonia," "Eretmochelys," Lepidochelys," "Carcharhinus," "Manta,""Mobula," "Balaenoptera," etc.*; "life history," "capture stress," "entanglement," "encirclement," "brailing," "crushing," "physiological," "histopathological," "whale," "cetacean, "population structure," "population trend," "demography," "vital rates," "bycatch," "mortality," "fishery impacts," "climate change," and "temperature-related sexual determination," while including species-specific nomenclature. 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: "FADs," "Fish aggregating device(s)," "ontogenetic shifts," "skeletochronology," "stable isotope," "lost years," "age at first reproduction," "shark," "manta," "cryptic mortality," "unaccounted mortality," "unobservable mortality," "survivorship," "slipping," "slipped catch," "dolphin sets," "whale associated," "net wall," "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," "causal loop analysis," "tag retention," and "egg mortality."

Electronic searches have important limitations, however. 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 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, 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, that might produce the same behavioral responses) higher than field experiments were not designed to control those 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. When two sources of data and other information were comparable in terms of quality, we relied on the data source that would provide the benefit of the doubt to the species. That is, we relied on the data or other information that would minimize our chances of falsely concluding "no effect."

Finally, we also relied on data from the Western and Central Pacific Fisheries Commission observer program for past interactions between the United States WCPO purse seine fishery and sea turtles, marine mammals, and sharks, recorded as bycatch between 2008 and 2018. These data represent credible and relevant commercial data. As a result, these represent the best scientific and commercial data available at the time of consultation. For reasons we discuss later in this biological opinion, we have determined that the observer program's data on giant manta rays are not credible because it likely contains a significant number of *Mobula* that are not giant manta rays. As a result, we found the data on giant manta rays to be implausible. Importantly, however, we believe the observer data is credible for the genus *Mobula.* Since this was the only data available to examine the effects of the action on threatened giant manta rays, this data also represent the *best scientific and commercial data available* for this consultation.

Importantly, observer data is raw data. To be useful for consultation and decision-making in general, raw data requires transformation. Once transformed, data becomes information that is useful for decision-making. The transformation of data is typically a multistep operation and typically involves such steps as organization (e.g., sorting, integration, splitting, restructuring) and processing (e.g., validating, duplicating, revising). When raw best available scientific and commercial data is transformed it becomes the best available scientific and commercial *information*. When a federal action agency has observational data collected from their specific action, as is the case for the United States WCPO purse seine fishery that raw data is usually going to be the best scientific and commercial data available for consultation on that action. Through collection, sorting, analyses and interpretation that raw best scientific and commercial data available is transformed into the *best scientific and commercial information available* for consultation. We would not be able to examine the past effects or predict future effects from the fishery if we did not transform the data.

3 STATUS OF LISTED RESOURCES

NMFS has determined that the action that NMFS IFD proposes to authorize, the United States WCPO purse seine fishery, 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	**SCIENTIFIC NAME**	**STATUS**
	SEA TURTLES	
Leatherback sea turtle	*Dermochelys coriacea*	Endangered
North Pacific loggerhead sea turtle	*Caretta caretta*	Endangered
South Pacific loggerhead sea turtle		Endangered
Southeast Indo-Pacific loggerhead sea turtle		Threatened
Eastern Pacific green sea turtle	*Chelonia mydas*	Threatened
Central North Pacific green sea turtle		Threatened
East Indian - West Pacific green sea turtle		Threatened
Central West Pacific green sea turtle		Endangered
Southwest Pacific green sea turtle		Threatened
Central South Pacific green sea turtle		Endangered
Olive ridley sea turtle, Mexico (Pacific coast) breeding populations	*Lepidochelys olivacea*	Endangered
Olive ridley, all other breeding populations		Threatened
Hawksbill sea turtle	*Eretmochelys imbricata*	Endangered
	MARINE MAMMALS	
Western North Pacific humpback whale	*Megaptera novaeangliae*	Endangered
Central America humpback whale		Endangered
Mexico humpback whale		Threatened
Fin whale	*Balaenoptera physalus*	Endangered
Blue whale	*Balaenoptera musculus*	Endangered
Sei whale	*Balaenoptera borealis*	Endangered
Sperm whale	*Physeter macrocephalus*	Endangered
Guadalupe fur seal	*Arctocephalus townsendi*	Threatened
	FISH	
Indo-West Pacific scalloped hammerhead shark	*Sphyrna lewini*	Threatened
Eastern Pacific scalloped hammerhead shark		Endangered
Oceanic whitetip shark	*Carcharhinus longimanus*	Threatened
Giant manta ray	*Manta birostris*	Threatened
	MARINE INVERTEBRATES	
Staghorn coral	*Acropora globiceps*	Threatened

SPECIES COMMON NAME	SCIENTIFIC NAME	STATUS
Staghorn coral	*Acropora jacquelineae*	Threatened
Staghorn coral	*Acropora lokani*	Threatened
Staghorn coral	*Acropora pharaonis*	Threatened
Staghorn coral	*Acropora retusa*	Threatened
Staghorn coral	*Acropora rudis*	Threatened
Staghorn coral	*Acropora speciosa*	Threatened
Staghorn coral	*Acropora tenella*	Threatened
Briar coral	*Anacropora spinosa*	Threatened
Branching frogspawn coral	*Euphyllia paradivisa*	Threatened
Catch bowl coral	*Isopora crateriformis*	Threatened
Pore coral	*Montipora australiensis*	Threatened
Leaf coral	*Pavona diffluens*	Threatened
Hump coral	*Porites napopora*	Threatened
Birdsnest coral	*Seriatopora aculeata*	Threatened
Chambered Nautilus	*Nautilus pompilius*	Threatened

3.1 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.

3.2 Listed Resources Not Considered Further

As described in the *Approach to the Assessment* section of this biological opinion, NMFS uses two criteria to identify those endangered or threatened, and candidate species that are not likely to be adversely affected by the United States WCPO purse seine fishery. The first criterion was exposure or some reasonable expectation of a co-occurrence between one or more potential stressors associated with the United States WCPO purse seine fishery and a particular listed or candidate species. If we concluded that a listed or candidate species is not likely to be exposed to the United States WCPO purse seine fishery, we also concluded that the species is not likely to be adversely affected by those activities. The second criterion is the probability of a response given exposure, which considers susceptibility: species that may be exposed to vessel noise from fishing vessels operating near them, for example, but are likely to be unaffected by the noise the vessel makes (at noise levels they are likely exposed to) are also not likely to be adversely affected by the United States WCPO purse seine fishery.

Based on the general exposure profiles that we developed during the course of this consultation, and described in Appendix A of this biological opinion, the following threatened and endangered species are not likely to be adversely affected by the United States WCPO purse seine fishery: 15 species of corals (*A. globiceps, A. jacquelineae, A. lokani, A. pharaonis, A. retusa, A. rudis, A. speciosa, A. tenella, A. spinose, E. paradivisa, I. crateriformis, M. australiensis, P. diffluens, P. napopora, Seriatopora aculeate),* chambered nautilus, blue whales, Western North Pacific humpback whales, Central America humpback whales, Mexico humpback whales, North Pacific loggerhead sea turtles, Southeast Indo-Pacific loggerhead sea turtle, Central North Pacific green sea turtles, Eastern Pacific scalloped hammerhead sharks, and Guadalupe fur seals.

Lastly, NMFS IFD previously determined in the biological evaluation, that the United States WCPO purse seine fishery may affect, but is not likely to adversely affect the fin whale (NMFS 2017a). However, as previously discussed in the *Consultation History*, NMFS PRD found records of the species in the data. Therefore, NMFS PRD and IFD agreed on July 30, 2020 that the species should be considered likely to be adversely affected for the remainder of this consultation and is analyzed as such.

3.3 Introduction to the Status of Listed Species

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 United States WCPO purse seine fishery. In each narrative, we present a summary of information on the distribution and population structure of each species to provide a foundation for the exposure analyses that appear later in this opinion. Then we summarize information on the threats to the species and the species' status given those threats to provide points of reference for the jeopardy determinations we make later in this opinion. That is, we rely on a species' status and trend to determine whether the action's direct or indirect effects are likely to increase the species' probability of becoming extinct.

3.3.1 Fin whale

3.3.1.1 Distribution and Population Structure

Fin whales occur worldwide and favor colder waters away from the tropics. They occur year-round in a wide range of latitudes and longitudes, but the density of individuals in any one-area changes seasonally. Fin whales are migratory, moving seasonally into and out of high-latitude, feeding areas, but the overall migration pattern is complex, and specific routes have not been documented. Fin whales were potentially recorded in the Eastern Tropical Pacific in 1996 at a latitude that is within that of the United States WCPO purse seine fishery but relatively far to the east of the eastern-most range of the fishery (Stafford et al. 1999). Three subspecies have been identified, based on geographical distribution and reproductive isolation. Fin whales migrate seasonally between summer and winter. Information about distribution and movement varies with location, and migratory routes are not well known. In general, food requirements drive distribution of the species.

Three subspecies are identified defined by separate populations in the North Atlantic, North Pacific, and the Southern Hemisphere, the last of which is the largest and least defined. The majority of the fin whales potentially affected by the action would be from the Southern Hemisphere, estimated at 82,000 mature individuals.

3.3.1.2 Status and Trends

Fin whales were listed globally as endangered under the ESA in 1970. The NMFS (2019d) reports the total global population of fin whales is estimated at about 100,000 mature individuals with an increasing trend. A key result of the 5-year status review NMFS (2019d) was a recommendation to downlist the species from endangered to threatened, however to-date NMFS has not proposed a change in the species status. In the North Pacific, the best minimum abundance estimates are for California/Oregon/Washington: 8,127 (Nadeem et al. 2016); Hawaii: 75 (Bradford et al. 2017); the Northeast Pacific: 2,554 (Moore et al. 2002; Zerbini et al. 2006) for a total minimum population estimate of 10,756 individuals. Although there is low confidence in the abundance estimate for fin whales in the southern hemisphere, the most current population estimate, calculated for 1978/1979, is 85,200 (no coefficient of variation [CV] given) based on the history of catch trends (International Whaling Commission [IWC] 1979). Zerbini et al. (2006) found evidence of a 4.8% (95% CI: 4.1 to 5.4%) per year increase for fin whales in Alaska. At this time, we do not have population growth trend information for any other populations of fin whale.

3.3.1.3 Population Dynamics

Age at sexual maturity for North Pacific fin whale was estimated to have declined over a 22-year period of exploitations, from 12 to 6 years for females and 11 to 4 years for males (Ohsumi 1986). Similarly, Lockyer (1972) suggests that age at sexual maturity Southern fin whales declined since the 1930's, when fin whales were killed by commercial whaling in large numbers. She estimates the age decreased from 10 or 11 years to 6 years for both males and females based on annual layer counts in the earplug (Lockyer 1972). Aguilar et al. (1988) estimated the age and size at sexual maturity to be 6.7 years and 17.4 meters for males and 7.7 years and 18.5 meters for females off of Spain.

Females breed every 2 to 3 years (Agler et al. 1993), with a gestation period of 11-12 months (Ohsumi et al. 1958; Christensen et al. 1992). Calves are about 20.5 feet long at birth (Ohsumi et al. 1958).

3.3.1.4 Diving and Social Behavior

Fin whales can be found in social groups of 2-7 whales. During the summer, fin whales feed on krill, small schooling fish (e.g., herring, capelin, and sand lance), and squid by lunging into schools of prey with their mouth open, using their 50-100 accordion-like throat pleats to gulp large amounts of food and water. They then filter the food particles from the water using the 260-480 baleen plates on each side of the mouth.

3.3.1.5 Threats to the Species

Fin whales are affected by natural and anthropogenic threats throughout their range. Climate change is affecting all oceans where warming temperatures and ocean acidification could affect prey abundance (Brodeur and Ware 1992; Roemmich and McGowan 1995; Francis et al. 1998; Kawaguchi et al. 2013), or change ocean circulation where it can affect the distribution of prey (Hays et al. 2005; Walther et al. 2002). Other natural threats include predation, disease, and food availability.

Commercial whaling decimated fin whales almost to extinction in the last couple of centuries. From 1900 through 1999, the whaling industry killed and processed nearly 2.9 million large whales globally. Fin whales were killed in larger numbers (874,068) than any other species (Rocha et al. 2015) before the International Whaling Commission banned the commercial killing of fin whales in 1986. Legal and illegal whaling continues throughout the world, including in the southern hemisphere where whales affected by this action exist. In addition to direct harvest, the recovery plan (NMFS 2010b) also included fishery interactions, vessel interactions, pollutants, entanglement in marine debris, research, and anthropogenic noise as potential anthropogenic threats to fin whales. Most of these threats were considered low except for direct harvest, which was considered a medium threat, and noise, vessel strikes, and climate change could potentially be high.

One incidental mortality of a fin whale due to entanglement in the ground tackle of a commercial mechanical jig fishing vessel was reported to the NMFS Alaska Region in 2012 (Helker et al. 2016). This mortality resulted in an estimated mean annual mortality and serious injury rate of 0.2 fin whales in 2010-2014 (Muto et al. 2018). There are currently two distinct longline fisheries based in Hawaii: the deep-set longline (DSLL) fishery that targets primarily tunas, and the SSLL that targets swordfish. Both fisheries operate within United States waters and on the high seas. Between 2004 and 2018, no fin whales were observed hooked or entangled in the SSLL fishery (100% observer coverage; NMFS 2019b) and between 2002 and 2017, none were observed hooked or entangled by the DSLL fishery (20-22% observer coverage) (McCracken 2019).

One fin whale death (in 1999) was observed in the California swordfish drift gillnet fishery from over 8,900 observed sets between 1990 and 2017 (Carretta et al. 2019b). Although no fin whales have been observed interacting with the fishery since 1999, new model-based bycatch estimates include a very small estimate of 0.2 whales (CV= 2.8) for the most recent 5-year period, 2013-2017 (Carretta et al. 2019b). The large CV of this bycatch estimate is a consequence of the mean estimate being very small. This estimate is based on inclusion of 28 years of observer data spanning 1990-2017 and reflects a very low long-term observed bycatch rate scaled up to levels of unobserved fishing effort. Some gillnet mortality of large whales may go unobserved because whales swim away with a portion of the net. One fin whale sighted at-sea was determined to be seriously injured (line cutting into the whale) as a result of interactions with unknown fishing gear during 2012-2018 (Carretta et al. 2018b). Including systematic fishery observations in the California swordfish drift gillnet fishery and opportunistic sightings of fishery-related injuries, the mean annual serious injury and mortality of fin whales for 2013-2017 is 0.2 whales (Caretta et al. 2019b). Gillnets have been documented to entangle marine mammals off Baja California (Sosa-Nishizaki et al. 1993 as cited in Carretta et al. 2019a), but no recent bycatch data from Mexico are available.

Foreign net-based or trap fisheries (e.g. purse seine, gillnet, trawl, pot/trap) that operate in pelagic areas that overlap with the fin whale's range may also impact the species, however no data are available to assess this threat (NMFS 2010b, 2019d). Krill fisheries in the Southern Ocean may impact the fin whale's prey base but any effect on the species is currently unknown (NMFS 2019d). We discuss the international WCPO purse seine fishery in the *Environmental Baseline* section.

3.3.1.6 Conservation

Protection for the fin whale occurs under both the ESA and the Marine Mammal Protection Act (MMPA) in the United States. The fin whale was listed as endangered throughout its range under the precursor to the ESA, the Endangered Species Conservation Act of 1969 (35 FR 8491; June 2, 1970), and remained on the list of threatened and endangered species after the passage of the ESA in 1973. Fin whales are also listed in Appendix I of the Convention on International Trade in Endangered Species (CITES) and the Convention on the Conservation of Migratory Species of Wild Animals (also known as the Bonn Convention). The CITES classification is intended to ensure that commercial trade in the products of fin whales does not occur across international borders and are not introduced from the sea. The Convention on the Conservation of Migratory Species of Wild Animals aims to conserve terrestrial, aquatic, and avian migratory species throughout their range. Through amendments to the Schedule to the ICRW, fin whale received complete protection from whaling throughout the North Atlantic Ocean in 1955, the North Pacific Ocean in 1966, and the entire Southern Hemisphere in 1968 (Best 1993). In 1982, the IWC set catch limits for all commercial whaling to zero. These catch limits went into effect in 1986, beginning a commercial whaling moratorium that remains in effect, but subsistence, commercial, and "research" harvest occurs today. Nonetheless, today's increasing trend indicates that the ban appears to be working.

3.3.1.7 Summary of the Status

In this section of this biological opinion, we explained that the fin whale is endangered, and that the species' trend appears to be increasing at a rate of 4.8% per year (95% confidence interval 4.1-5.4%; Zerbini et al. 2006) for the Northeast Pacific stock and 7.5% per year (95% prediction interval 5.1 – 9.8%; Nadeem et al. 2016) for the California/Oregon/Washington stock, and there are no trend estimates for the Hawaii stock. We used our knowledge of the species' demography and population ecology to capture the primary factors that appear to determine the fin whale's population dynamics (Figure 12). Where we could, we included a graphic in our causal loop diagram to represent the behavior of the reference variables (data) that influenced the behavior of that particular loop.

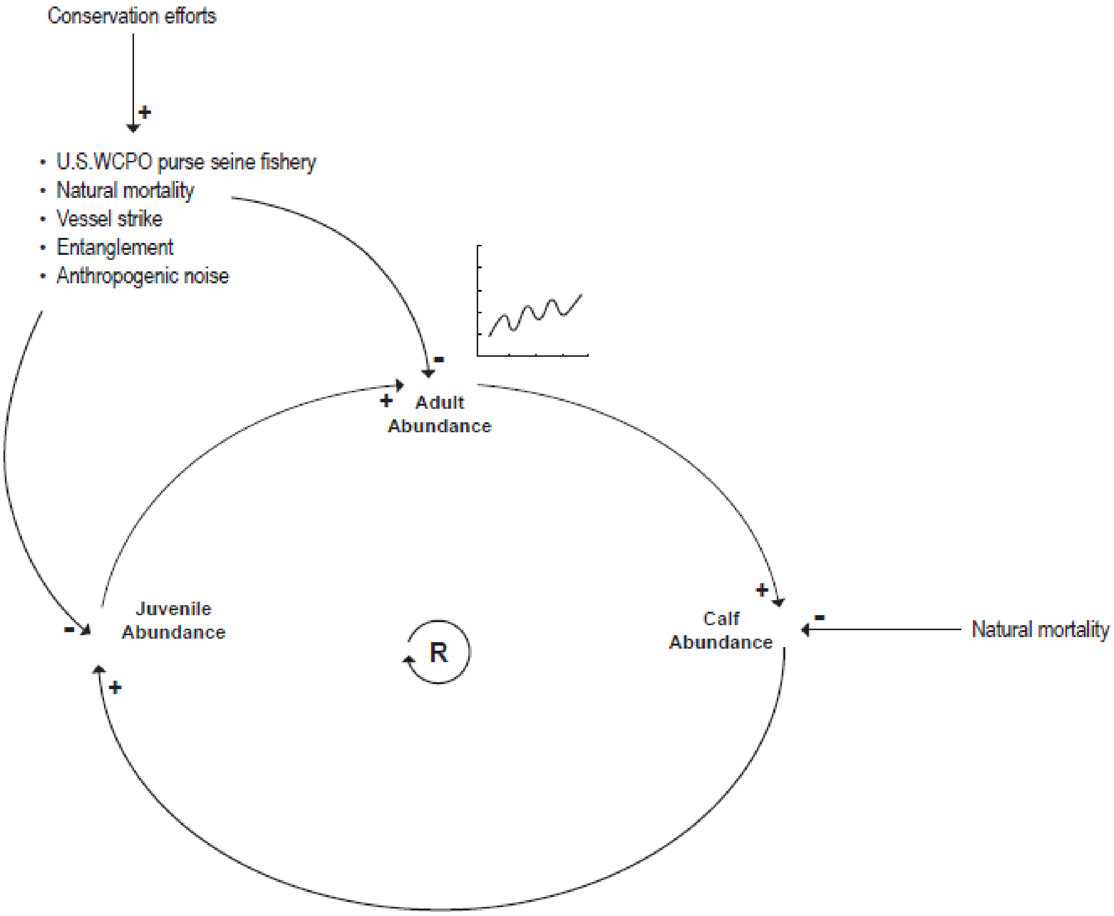


Figure 12. Fin whale casual loop diagram depicting various stressor-response relationships for each life cycle stage.

3.3.2 Sei Whale

3.3.2.1 Distribution and Population Structure

Sei whales have a cosmopolitan distribution and occur in subtropical, temperate, and subpolar waters around the world (Gambell 1985; Horwood 1987). They prefer temperate waters in the mid-latitudes, and exist in the Atlantic, Indian, and Pacific Oceans.

Populations of sei whales, like other rorquals, may seasonally migrate toward the tropical latitudes during the winter and polar latitudes during the summer. This species may occur randomly and unpredictably, sometimes in large numbers. These events may occur suddenly and then not occur again for long periods. This pattern may hold true in the WCPO tuna purse seine fishery area, as numbers sei whale sightings and interactions fluctuated widely over the years 2008-2018 based on observer reports.

Two subspecies have been classified (separated by the Northern and Southern Hemispheres), and three populations are recognized; the North Atlantic, North Pacific, and Southern Hemisphere. Whales in the *Action Area* would most likely be from the Southern Hemisphere population.

3.3.2.2 Status and Trends

Sei whales were listed globally as endangered under the ESA in 1970. Cooke (2018b) estimates the total global population of sei whales at about 50,000 mature individuals. These are rough estimates based on incomplete and decades old data and a conventional population assessment model (Cooke 2018b). The model estimated fitted population sizes for the aged 1+ population size were 12,000 in 1989 in the North Atlantic, 35,000 in 2011 in the North Pacific and 10,000 in 1983 in the Southern Hemisphere. Comparatively, NMFS (2012b) estimated populations of 11,000-13,000 individuals in the North Atlantic, 8,600 individuals in the North Pacific, and 9,800-12,000 individuals in the Southern Hemisphere.

Braham (1991) provided a pre-exploitation estimate of 65,000 sei whales (no CV) in the Southern Hemisphere. This estimate is similar to that of Mizroch et al. (1984), who estimated a total of 63,100 sei whales (no CV) occurring in these waters prior to exploitation. In the Southern Hemisphere, more recent population estimates range between 9,800 and 12,000 sei whales (no CV) (Mizroch et al. 1984; Perry et al. 1999). Similarly, the IWC reported an estimate of 9,718 sei whales (no CV) based on survey results from 1978 to 1988 (IWC 1996). There are no estimates of trend, whether positive or negative, for this species.

3.3.2.3 Population Dynamics

Sei whales become sexually mature at 6 to 12 years of age when they reach about 45 ft. in length. They generally mate and give birth during the winter in lower latitudes.

Females breed every 2 to 3 years, with a gestation period of 11 to 13 months. Calves are about 15 ft. long and weigh approximately 1,500 lbs. at birth. Mothers nurse their calves for 6 to 9 months before weaning them when at their preferred feeding grounds. The sei whales average lifespan is estimated to be between 50 and 70 years of age.

3.3.2.4 Diving and Social Behavior

Sei whales occur singly or in small groups of 2-5 animals, but records of larger (30-50) loose aggregations exist. Sei whales are capable of diving 5-20 minutes while opportunistically feeding on plankton (e.g., copepods and krill), small schooling fish, and cephalopods (e.g., squid). Ishii et al. (2017) determined sei whales display both U and V dive profiles which are dictated by the diel migration of prey within the scattering later. Overall mean dive depths were recorded for two individuals, 17.9 ± 12.6m (deepest depth recorded 57m) and 14.2 ± 9.6m (deepest depth recorded 48 m) respectively with statistically significant deeper dives noted during daytime hours (Ishii et al. 2017).

3.3.2.5 Threats to the Species

Sei whales are affected by natural and anthropogenic threats similar to all pelagic whales throughout their range. Climate change is affecting all oceans where warming temperatures and ocean acidification could affect prey abundance (Brodeur and Ware 1992; Roemmich and McGowan 1995; Francis et al. 1998; Kawaguchi et al. 2013), or change ocean circulation where it can affect the distribution of prey (Hays et al. 2005; Walther et al. 2002). Other natural threats include predation, disease, and food availability.

Sei whales were historically hunted, which reduced historic populations. Most whaling has been banned for at least three decades but Japan continues to harvest sei whales and Iceland has expressed interest in harvesting sei whales. Although some whaling continues, it does not appear to be a major threat presently. Aside from direct harvest, the status review (NMFS 2012b) considers fishery interactions, vessel interactions, entanglement, anthropogenic noise, climate change, and pollution, as potential man-made threats, most of which are considered low. Ship strikes (NMFS 2011; Carretta et al. 2015) and gear entanglement (Waring et al. 2009) have been reported and could be more severe in some regions.

The California swordfish drift gillnet fishery is the only fishery that is likely to interact with sei whales in the Eastern North Pacific, but no fishery mortality or serious injuries have been observed from over 8,600 monitored fishing sets from 1990-2014 (Carretta et al. 2019b). Mean annual interactions with this fishery reflect 2010-2014 data. This results in an average estimate of zero sei whale interactions annually. However, some gillnet mortality of large whales may go unobserved because whales swim away with a portion of the net (Carretta et al. 2019b).

Information on fishery-related mortality of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaiian fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout United States waters. A report of an entangled sub adult sei whale in March 2011 occurred near Lahaina, Maui. The whale had one or two wraps of heavy-gauge polypropylene line around the tailstock and trailing about 30 feet of line including a large bundle (Bradford and Lyman 2013). NMFS made multiple disentanglement attempts, however the gear remained on the individual. Closer examination also revealed line scars on the body near the dorsal fin. Although the source of the line entangling the whale could not be determined, this injury was considered serious based on extent of trailing gear and condition of the whale (NMFS 2012c; Bradford and Lyman 2013). This serious injury record results in an average annual serious injury and mortality rate of 0.2 sei whales from Hawaii for the period 2007 to 2011 (Carretta et al. 2019a). There are currently two distinct longline fisheries based in Hawaii: the DSLL fishery that targets primarily tunas, and the SSLL that targets swordfish. Both fisheries operate within United States waters and on the high seas and neither fishery has had an interaction with a Sei whale based on observer data (McCracken 2019b; NMFS 2019b). Similar to fin whales, foreign net-based or trap fisheries (e.g. purse seine, gillnet, trawl, pot/trap) that operate in pelagic areas that overlap with sei whales' range, including in the South Pacific, may also impact the species, however no data are available to assess this threat (NMFS 2011, 2012b). We discuss impacts from the international WCPO purse seine fishery in the *Environmental Baseline* section.

3.3.2.6 Conservation

Protection for the sei whale occurs under both the ESA and the MMPA in the United States. The sei whale was listed as endangered throughout its range under the precursor to the ESA, the Endangered Species Conservation Act of 1969 (35 FR 8491), and remained on the list of threatened and endangered species after the passage of the ESA in 1973. Sei whales are also listed in Appendix I of CITES and the Bonn Convention. The CITES classification is intended to ensure that commercial trade in the products of sei whales does not occur across international borders and are not introduced from the sea. The Bonn Convention, aims to conserve terrestrial, aquatic, and avian migratory species throughout their range. Through amendments to the Schedule to the ICRW, sei whale received complete protection from whaling throughout the North Atlantic Ocean in 1955, the North Pacific Ocean in 1966, and the entire Southern Hemisphere in 1968 (Best 1992 as cited in NMFS 2012b). In 1982, the IWC set catch limits for all commercial whaling to zero. These catch limits went into effect in 1986, beginning a commercial whaling moratorium that remains in effect today but some harvest still exists today. Today's increasing trend indicates that the ban appears to be working.

3.3.2.7 Summary of the Status

In this section of this biological opinion, we explained that the sei whale is endangered, and there are no data on the species' trend. We used our knowledge of the species' demography and population ecology to capture the primary factors that appear to determine the sei whale's population dynamics (Figure 13). Where we could, we included a graphic in our causal loop diagram to represent the behavior of the reference variables (data) that influenced the behavior of that particular loop.

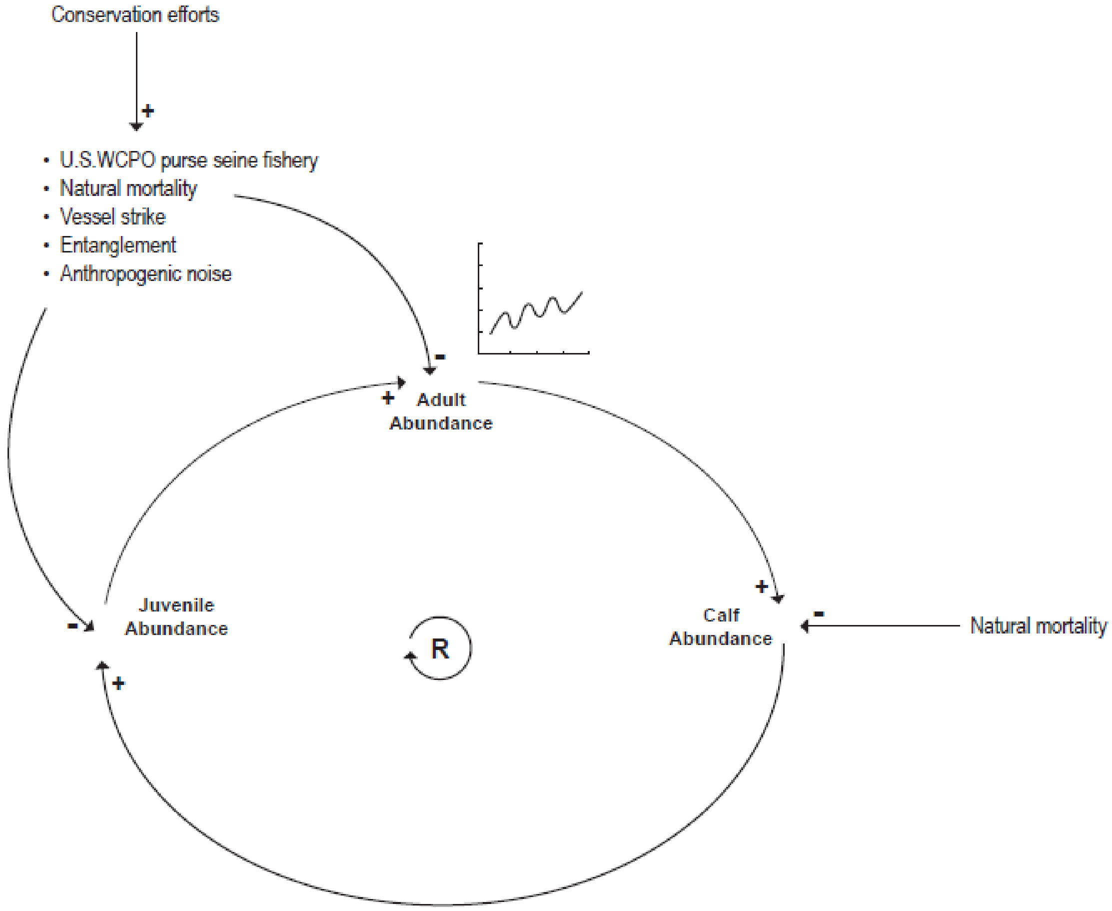


Figure 13. Sei whale casual loop diagram depicting various stressor-response relationships for each life cycle stage.

3.3.3 Sperm Whale

3.3.3.1 Distribution and Population Structure

Sperm whales are one of the most widely distributed whales in the world, occupying all oceans and latitudes. Sperm whales are migratory, moving seasonally into and out of high-latitude feeding areas, but overall migration patterns are complex, and specific routes have not been documented. NMFS recognizes four geographic regions where sperm whales are counted and tracked; Atlantic Ocean, Pacific Ocean, Indian Ocean, and a residential population in the Mediterranean Sea (NMFS 2015c).

3.3.3.2 Status and Trends

Sperm whales were listed globally as endangered under the ESA in 1970. The sperm whale is the most abundant of the large whale species. The most recent estimate indicated a global population of between 300,000 and 450,000 individuals (Whitehead 2002; NMFS 2015c). In the North Pacific the most reliable estimates of abundance are 1,997 for California/Oregon/Washington (Moore and Barlow 2014) and 4,559 for Hawaii (Bradford et al. 2017) resulting in a total minimum estimate of 6,556. We are aware of no reliable abundance estimates specifically for sperm whales in the South Pacific Ocean. Branch and Butterworth (2001) estimate 5,400 to 10,000 individuals in the Antarctic based on summer sighting surveys.

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

3.3.3.3 Population Dynamics

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

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

3.3.3.4 Diving and Social Behavior

Sperm whales are often concentrated around oceanic islands in areas of upwelling, along the outer continental shelf and mid-ocean waters, and rarely found in waters less than 300 m in depth. Sperm whales routinely dive to depths of hundreds of meters, and may occasionally dive as deep as 3,000 m (Rice 1989). Presumed feeding events have been shown to occur at depths greater than 1,200 m (Wahlberg 2002). Sperm whales are capable of remaining submerged for longer than two hours, but most dives probably last a half hour or less (Rice 1989; Watwood et al. 2006).

Most females will form lasting bonds with other females of their family, and on average 12 females and their young will form a social unit. While females generally stay with the same unit all their lives in and around tropical waters, young males will leave when they are between 4 and

21 years old and can be found in "bachelor schools," comprised of other males that are about the same age and size. As males get older and larger, they begin to migrate toward the poles and slowly bachelor schools become smaller; the largest males are often found alone. Large, sexually mature males that are in their late 20s or older will occasionally return to the tropical breeding areas to mate.

3.3.3.5 Threats to the Species

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

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

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

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

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

Interactions between longline fisheries and sperm whales in the Gulf of Alaska have been reported (Rice 1989; Hill and DeMaster 1999). Observers aboard Alaskan sablefish and halibut longline vessels have documented sperm whales feeding on longline-caught fish in the Gulf of Alaska (Hill et al. 1999; NMFS unpublished data) and in the South Atlantic (Ashford et al. 1996). The available evidence does not indicate sperm whales are being killed or seriously injured as a result of these interactions, although the nature and extent of interactions between sperm whales and long-line gear is not clear. Ashford et al. (1996) suggested that sperm whales pluck, rather than bite, the fish from the long-line. Mean estimated mortality of sperm whales in the Gulf of Alaska sablefish longline fishery was 3.7 per year from 2011-2015 (Muto et al. 2018).

There are currently two distinct longline fisheries based in Hawaii: a DSLL fishery that targets primarily tunas, and the SSLL that targets swordfish. Both fisheries operate within United States waters and on the high seas. The Hawaii-based DSLL fishery is estimated to interact with up to three sperms whales each year (McCracken 2019). Two sperm whale interactions have been observed and each of these whales was entangled in fishing gear, one was released with no gear attached while the other was released with a hook, branch line, leader line, and possibly 25 to 30 ft. of mainline (NMFS unpublished data).

The estimated mean annual mortality and serious injury to the Hawaii population of sperm whales as a result of other commercial fishing operations for the period from 2011 through 2015 was 0.7 per year (Carretta et al. 2019a). From 2010 to 2014, there were no reported sperm whale interactions with the American Samoa longline fishery (ASLL; Bradford and Forney 2017).

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

3.3.3.6 Conservation

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

3.3.3.7 Summary of the Status

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

Figure 14). Where we could, we included a graphic in our causal loop diagram to represent the behavior of the reference variables (data) that influenced the behavior of that particular loop.

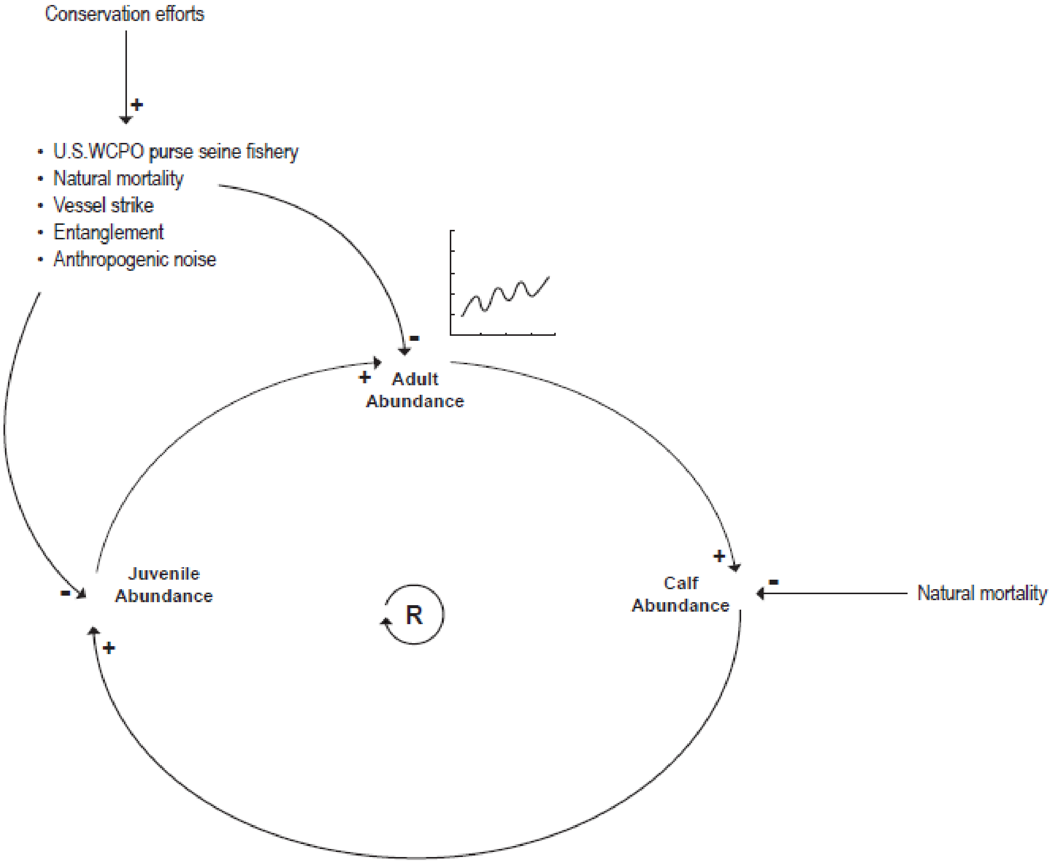


Figure 14. Sperm whale casual loop diagram depicting various stressor-response relationships for each life cycle stage.

3.3.4 Green Sea Turtle

Green sea turtles were listed as threatened on July 28, 1978 (43 FR 32800), except for breeding populations that occur in Florida and the Pacific coast of Mexico, which were listed as endangered. On April 6, 2016, NMFS and the FWS published a final rule finding that the green sea turtle is composed of 11 DPS' (Figure 15) that each qualify as a "species" for purposes of the ESA. The Services removed the current range-wide listing and, in its place, listed eight species of green sea turtle as threatened and three as endangered. The species of green sea turtles most likely to occur in the range of the United States WCPO purse seine 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. The following sections discuss these green sea turtle species because they are the species most likely to be encountered by the United States WCPO purse seine fishery.

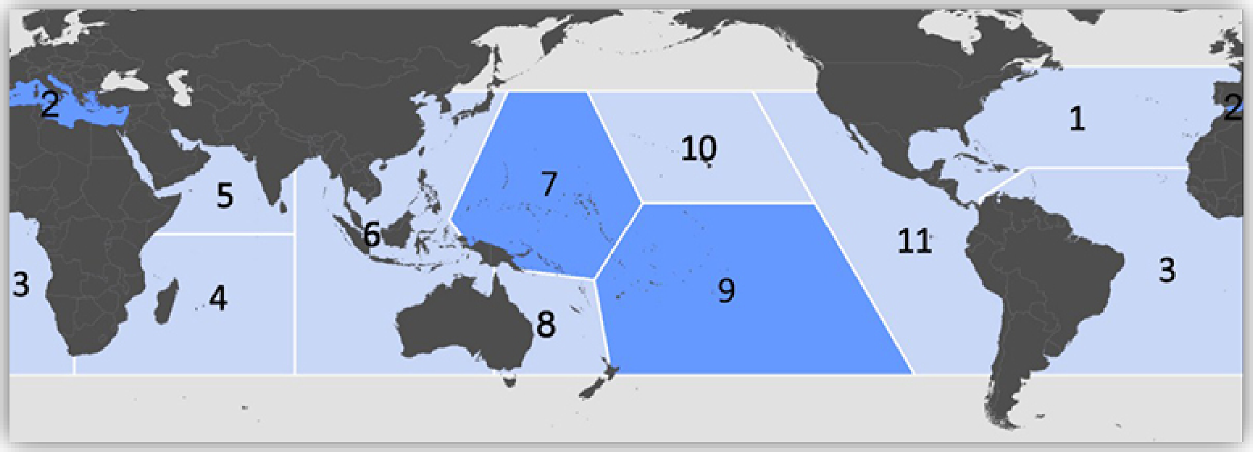


Figure 15. Overview of listed green sea turtle species; (1) North Atlantic; (2) Mediterranean; (3) South Atlantic; (4) Southwest Indian; (5) North Indian; (6) East Indian-West Pacific; (7) Central West Pacific; (8) Southwest Pacific; (9) Central South Pacific; (10) Central North Pacific; and (11) East Pacific. Light blue indicates threatened populations and dark blue indicates endangered populations.

During their pelagic phase, juvenile green turtles have a varied diet that includes planktonic material including crustaceans, jellyfish and ctenophores. Sub-adults and adult green turtles will feed on tuna hooks baited with squid or fish (Beverly and Chapman 2007). The deepest dives recorded for green turtles are from adults migrating from the main Hawaiian Islands 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.

3.3.4.1 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.

3.3.4.2 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 2007). However, 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.

3.3.4.3 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 species ranges from the aforementioned locations in the USA and Chile to 143°W and 96°W, respectively (Figure 16). This species encompasses the Revillagigedos Archipelago (Mexico) and Galapagos Islands (Ecuador). The extent pelagic habitats used by small juveniles or migrating adults are not well understood, however, genetic samples from green sea turtles interacting with the ASLL fishery between 2006 and 2017 indicate that sea turtles with haplotypes specific to this species are captured by that fishery (NMFS unpublished data), therefore we assume that there is a potential for sea turtles from this species to interact with the United States WCPO purse seine fishery, most likely in the eastern-most portion of the *Action Area*.

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 species 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). At this time, we cannot describe which specific population(s) may be interacting with the United States WCPO purse seine fishery, as genetic data is not collected from green sea turtles.

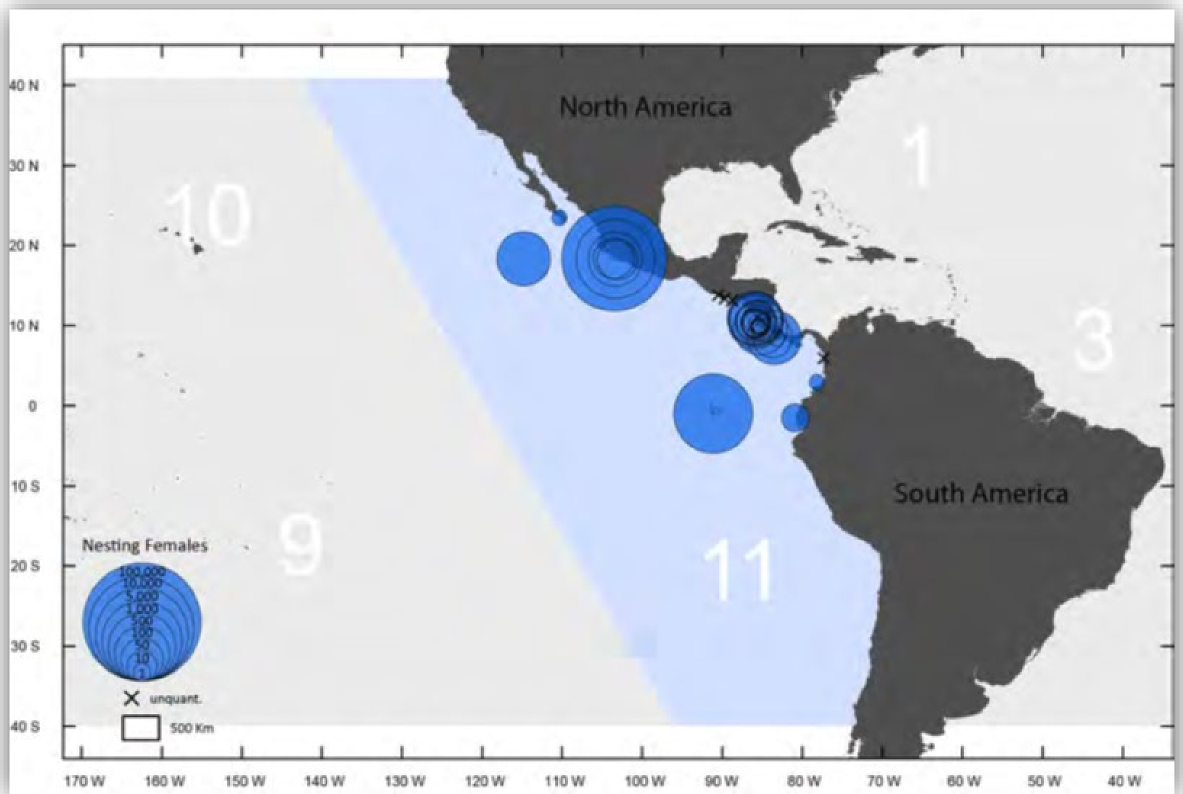


Figure 16. Nesting distribution of East Pacific green sea turtles (blue-shaded area marked with '11'). Size of circles indicates estimated nester abundance.

Status and Trends

The East Pacific green sea turtle is listed as threatened. Seminoff et al. (2015) ranked the species 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 for 20,108 nesting females (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 Michoacan, Mexico (which is estimated to comprise about 58% of the total adult females for the species), the abundance of East Pacific green turtle nesting females appears to have increased since the population's low point in the mid-1980s. 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.

While there are no total abundance estimates for this species, we use the abundance estimate for the Central North Pacific green sea turtle as a proxy for our analyses. There are an estimated 3,846 Central North Pacific green sea turtle nesting females, which is considerably lower than the 20,000 nesting females estimated for the East Pacific green sea turtle. IUCN Redlist categorizes the status of the Central North Pacific green sea turtle as near threatened based on a current estimate that the population size is at levels that corresponds to 83% of pre-exploitation levels at approximately 265,600 turtles or more (Chaloupka and Balazs 2007; Pilcher et al. 2012). The historic population size is estimated to be approximately 320,000 individuals (Pilcher et al. 2012). Given the higher number of East Pacific green sea turtle nesting females compared to Central North Pacific green sea turtles, we consider 265,600 a minimum estimate of total abundance for this species.

Population Dynamics

The population dynamics of this species are similar to that of the other green turtles, but the adult pattern begins earlier because East Pacific green turtles mature at smaller sizes (60–77 cm SCL; Seminoff et al. 2002). Average 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, Seminoff et al. (2002) reports that East Pacific green turtles reach sexual maturity after spending 9 and 21 years in neritic habitats 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

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

The largest threat on East Pacific green sea turtle nesting beaches is reduced availability of habitat due to heavy armament and subsequent erosion. In addition, while nesting beaches in Costa Rica, Revillagigedos Islands, and the Galapagos Islands are less affected by coastal development than green turtle nesting beaches in other regions around the Pacific, several of the secondary green sea turtle nesting beaches in Mexico suffer from coastal development. For example, effects of coastal development are especially acute at Maruata, a site with heavy tourist activity and foot traffic during the nesting season (Seminoff 1994). Nest destruction due to human presence is also a threat to nesting beaches in the Galapagos Islands (Zarate et al. 2006). However, such threats vary by site. All life stages of green turtles are affected by habitat destruction in the neritic/oceanic zone resulting from coastal development and marine pollution. 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. In the Eastern Pacific Ocean, particularly areas in the southern portion of the range of this species, significant bycatch has been reported in artisanal gill net and longline shark and 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 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 small-scale fisheries 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, 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 ridley and 70 leatherback turtles). Small-scale fisheries 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 small-scale fisheries have the potential to severely affect 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 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 turtle excluder devices or remove them entirely from their trawls.

The Hawaii-based shallow-set longline fishery interacted with ten green turtles (zero mortalities) from 2004-2018 (NMFS 2019b). Based on genetic samples form 19 green sea turtles captured since the 1990s, NMFS estimates that between 32 and 77% of the turtles caught in the Hawaii-based shallow-set longline 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 species have been captured by the Hawaii-based shallow-set longline fishery since 2004. The Hawaii-based deep-set longline fishery interacted with approximately 71 green turtles (65 estimated mortalities) between 2004 and 2018 (NMFS unpublished data). Based on genetic samples from 31 green sea turtles, NMFS estimates that between 31 and 80% of the turtles caught in the Hawaii-based deep-set longline fishery may have been East Pacific green sea turtles (P. Dutton pers. comm. June 29, 2018). This means that an estimated 22 to 57 East Pacific green sea turtles were captured in the Hawaii-based deep-set longline fishery since 2004. The ASLL fishery interacted with approximately 286 green turtles (with 259 estimated mortalities) between 2006 and 2018 (NMFS unpublished data). Based on genetic samples from 31 green sea turtles, NMFS estimates that between 2 and 23% may have been East Pacific green sea turtles (P. Dutton pers. comm. July 5, 2018). This means that the ASLL fishery captured an estimated 5 to 66 East Pacific green sea turtles since 2006.

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 for used as a cold remedy and the meat, eggs and other products have been highly valued for their presumed aphrodisiacal qualities.

3.3.4.4 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 17). 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 species 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. With the exception of French Polynesia, all of these areas are within the *Action Area.*

Nesting in this species of green turtle has been reported from 57 locations in the species' range (Figure 17) and most of these locations directly overlap with the *Action Area*. 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%).

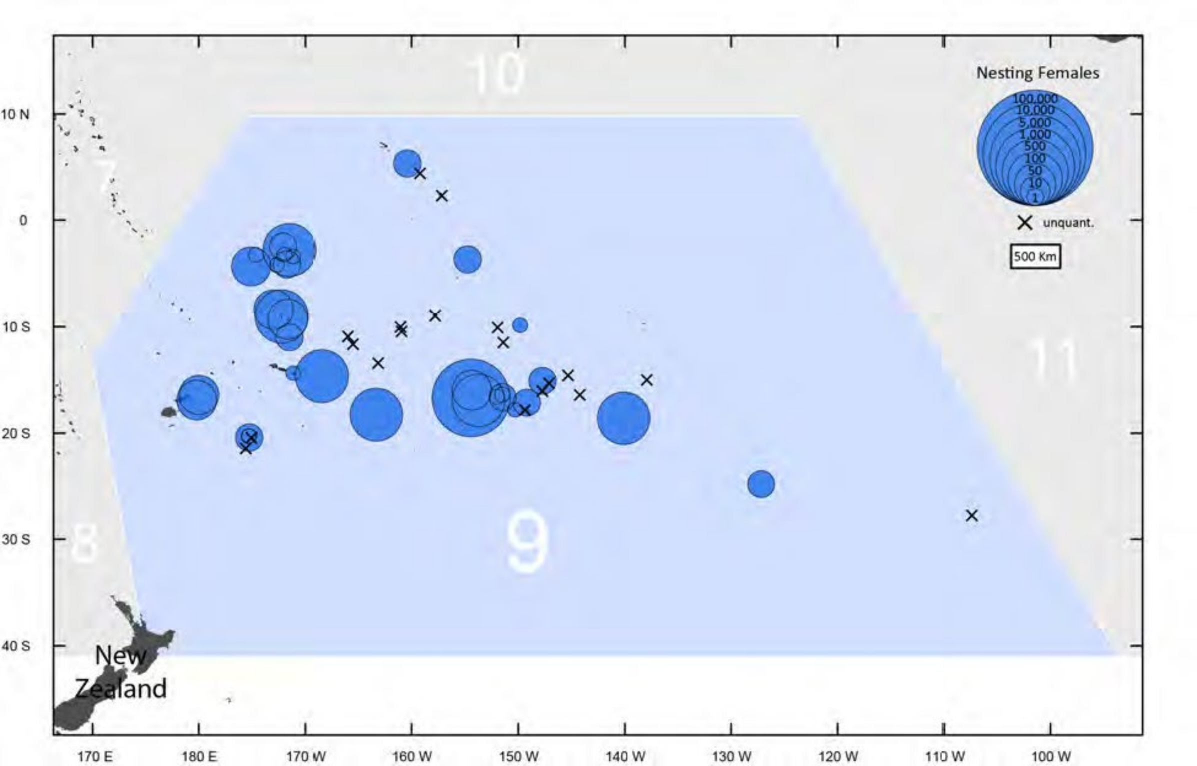


Figure 17. Nesting distribution of Central South Pacific green sea turtles. Size of circles indicates estimated nester abundance. Locations marked with 'X·' indicate nesting sites lacking abundance information.

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 species had a 62% probability of having a greater than 1% extinction risk of extinction over the next 100 years (Seminoff et al. 2015). The Status Review Team (SRT) estimate 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 value of 2,677 as the minimum number of nesters. The largest nesting site, Scilly Atoll represents roughly one third of the species' 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).

While there are no total abundance estimates for this species, we use the abundance estimate for the Central North Pacific green sea turtle as a proxy for our analyses. There are an estimated 3,846 Central North Pacific green sea turtle nesting females. IUCN Redlist categorizes the status of the Central North Pacific green sea turtle as near threatened based on a current estimate than the population size is at levels that corresponds to 83% of pre-exploitation levels at approximately 265,600 turtles or more (Chaloupka and Balazs 2007; Pilcher et al. 2012). The historic population size is estimated to be approximately 320,000 individuals (Pilcher et al. 2012). Given that the estimated number of Central South Pacific green sea turtle nesting females (2,677) is 70% less than the Central North Pacific green sea turtle nesting females, we consider 70% of the Central North Pacific green sea turtle abundance (184,870 individuals) a conservative minimum estimate of total abundance for this species.

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 species of green turtle at a relatively high risk of extinction in the foreseeable future (Seminoff et al. 2015).

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). 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). All life stages of green turtles are affected by habitat destruction in the neritic/oceanic zone resulting from coastal development and marine pollution.

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.

NMFS estimates that between 0 and 5% of the green sea turtles caught in the Hawaii SSLL fishery could be Central South Pacific green sea turtles (P. Dutton pers. comm. November 20, 2017). This means that the SSLL fishery may have captured one South Central Pacific green sea turtle since 2004. NMFS further estimates that between 0 and 24% of the green sea turtles caught in the Hawaii DSLL fishery may be Central South Pacific green sea turtles (P. Dutton pers.

comm. June 29, 2018) meaning the DSLL fishery captured between 0 and 16 Central South Pacific green sea turtles since 2004.

3.3.4.5 Southwest Pacific Green Sea Turtle

Distribution and Population Structure

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

Green turtle nesting is widely dispersed throughout the Southwest Pacific Ocean. Genetic data suggest that this "species" 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 occur in Australia's Great Barrier Reef World Heritage Area and eastern Torres Strait. Nesting also occurs on the Coral Sea Islands, New Caledonia, and Vanuatu, which are within the *Action Area*.

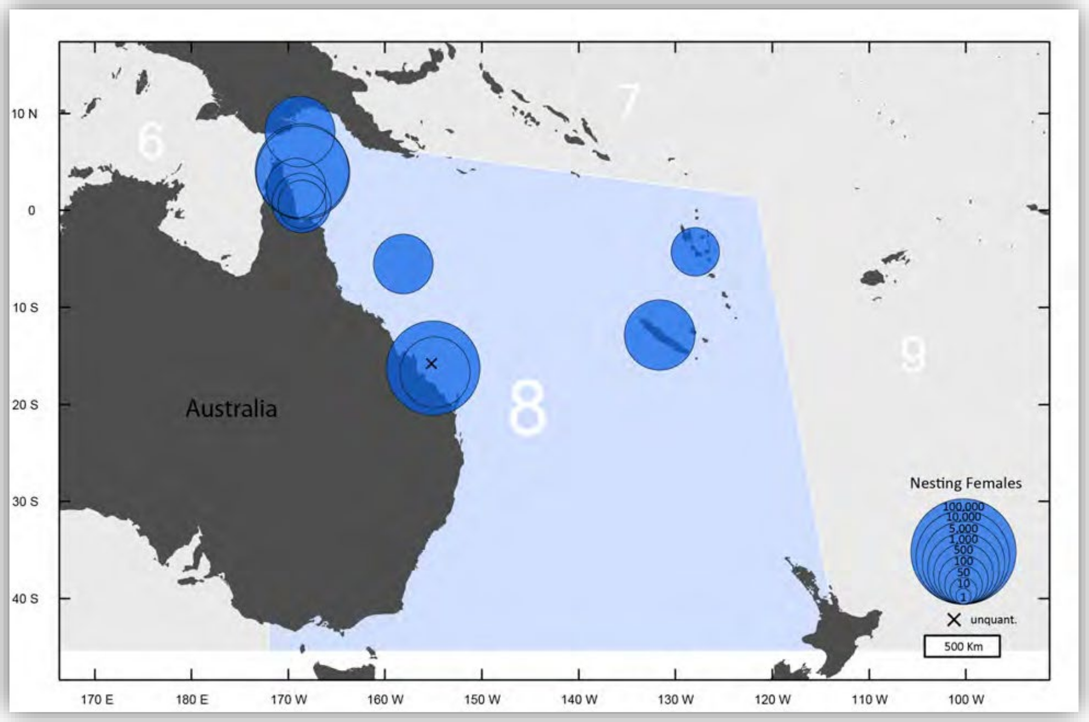


Figure 18. Nesting distribution of Southwest Pacific green sea turtles. Size of circles indicates nesting estimated nester abundance. Locations marked with '·' indicate nesting sites lacking abundance information.

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. 2008a; Limpus 2009). 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 female green turtles 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).

While there are no total abundance estimates for this species, we use the abundance estimate for the Central North Pacific green sea turtle as a proxy for our analyses. There are an estimated 3,846 Central North Pacific nesting green sea turtles, which is considerably lower than the tens of thousands Southwest Pacific green sea turtle (Figure 18). IUCN Redlist categorizes the status of the Central North Pacific green sea turtle as near threatened based on a current estimate that the population size is at levels that corresponds to 83% of pre-exploitation levels at approximately 265,600 turtles or more (Chaloupka and Balazs 2007; Pilcher et al. 2012). The historic population size is estimated to be approximately 320,000 individuals (Pilcher et al. 2012). Given the higher number of Southwest Pacific green sea turtle nesting females compared to Central North Pacific green sea turtles, we consider 265,600 a minimum estimate of total abundance for this species.

Population Dynamics

The population dynamics of this species are similar to that of the other green turtles, but these green turtles may produce higher proportions of females or females in this species 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 species is possible in the foreseeable future. If this occurs, it would have severe implications for the future of this species.

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. (2009), 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. 2009). All life stages of green turtles are affected by habitat destruction in the neritic/oceanic zone resulting from coastal development and marine pollution. 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 Native harvest 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 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).

NMFS estimates that between 0 and 8% of the green sea turtles caught in the Hawaii shallow-set longline fishery could be Southwest Pacific green sea turtles (P. Dutton pers. comm. November 20, 2017). This means that one turtle from this species may have been captured by this fishery since 2004. NMFS estimates that between 0 and 25% of the green sea turtles caught in the Hawaii deep-set longline fishery could be Southwest Pacific green sea turtles (P. Dutton pers. comm. June 29, 2018). This means that 0-17 turtles captured in the Hawaii deep-set longline fishery since 2004 were Southwest Pacific green sea turtles.

3.3.4.6 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 19).

The beaches with the highest abundance of nesting females in this species 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 turtles also nest on Guam and the Commonwealth of the Northern Mariana Islands (Seminoff et al. 2015). Nesting beaches in the Solomon Islands are within the *Action Area*, and nesting beaches in other areas such as the Federated States of

Micronesia and Palau are just outside of the *Action Area*, making it likely that green sea turtles from these nesting beaches migrate through the *Action Area.*

Green turtles from this species 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). In 1991, a turtle tagged in Yap 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).

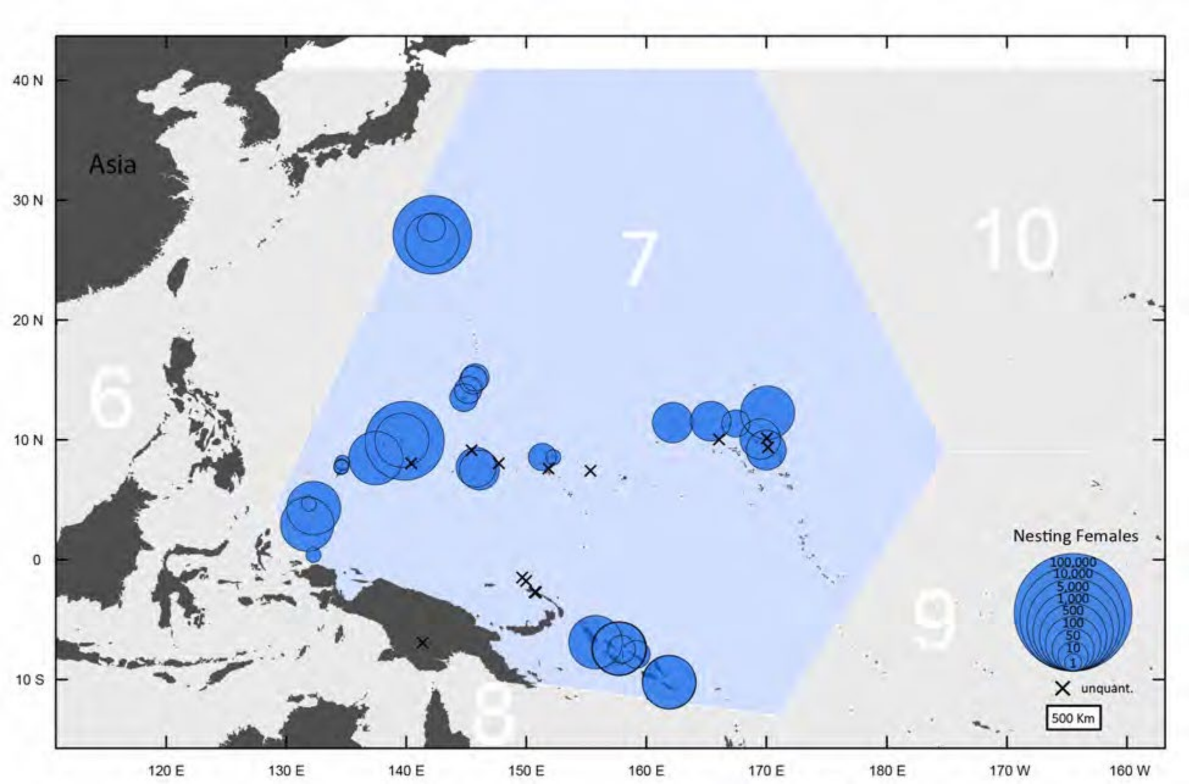


Figure 19. Nesting distribution of Central West Pacific green sea turtles (blue-shaded region). Size of circles indicates estimate nester abundance.

Status and Trends

Central West Pacific green sea turtles are listed as endangered. The species 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.

While there are no total abundance estimates for this species, we use the abundance estimate for the Central North Pacific green sea turtle as a proxy for our analyses. There are an estimated 3,846 Central North Pacific green sea turtle nesting females, which is lower than the estimated 6,518 Central West Pacific green sea turtle. IUCN Redlist categorizes the status of the Central North Pacific green sea turtle as near threatened based on a current estimate that the population size is at levels that corresponds to 83% of pre-exploitation levels at approximately 265,600 turtles or more (Chaloupka and Balazs 2007; Pilcher et al. 2012). The historic population size is estimated to be approximately 320,000 individuals (Pilcher et al. 2012). Given the higher number of Central West Pacific green sea turtle nesting females compared to Central North Pacific green sea turtles, we consider 265,600 a minimum estimate of total abundance for this species.

The limited available information suggests a nesting population decrease in some portions of the species 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). 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 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 that this population will fall below the trend reference or absolute abundance reference in 100 years approaches zero (Seminoff et al. 2015). The population has increased from a mean of approximately 100 females/year in the late 1970s/early 1980s to a mean of approximately 500 per year since 2000. Similarly, Chaloupka et al. (2008a) reports an estimated annual population growth rate of 6.8% per year for the Chichi-jima nesting site.

Population Dynamics

The population dynamics of this species 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 CCL in Palau to 104.5 cm curved carapace length (CCL) in Tinian, Commonwealth of the Northern Mariana Islands (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 species (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 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 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 species 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. All life stages of green turtles are affected by habitat destruction in the neritic/oceanic zone resulting from coastal development and marine pollution. Central West Pacific Ocean green turtles are incidentally captured in artisanal and commercial fisheries throughout the region. They are captured 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, 2014b, 2015, 2016, 2017, 2018).

NMFS estimates that between 0 and 8% of the green sea turtles caught in the Hawaii shallow-set longline fishery could be Central West Pacific green sea turtles (P. Dutton pers. comm. November 20, 2017). This means that the Hawaii shallow-set longline fishery may have captured one Central West Pacific green sea turtle since 2004. NMFS estimates that between 0 and 18% of the green sea turtles caught in the Hawaii deep-set longline fishery could be from the Central West Pacific species (P. Dutton pers. comm. June 29, 2018). This means that between 0 and 12 Central West Pacific green sea turtles were captured in the Hawaii deep-set longline fishery since 2004.

3.3.4.7 East Indian-West Pacific Green Sea Turtle

Distribution and Population Structure

East Indian-West Pacific green sea turtles use nesting beaches from northern Australia to southern Asia (Figure 20). Based on studies of 22 rookeries, 16 populations or subpopulations of this species 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 and are likely to occur in the *Action Area*.

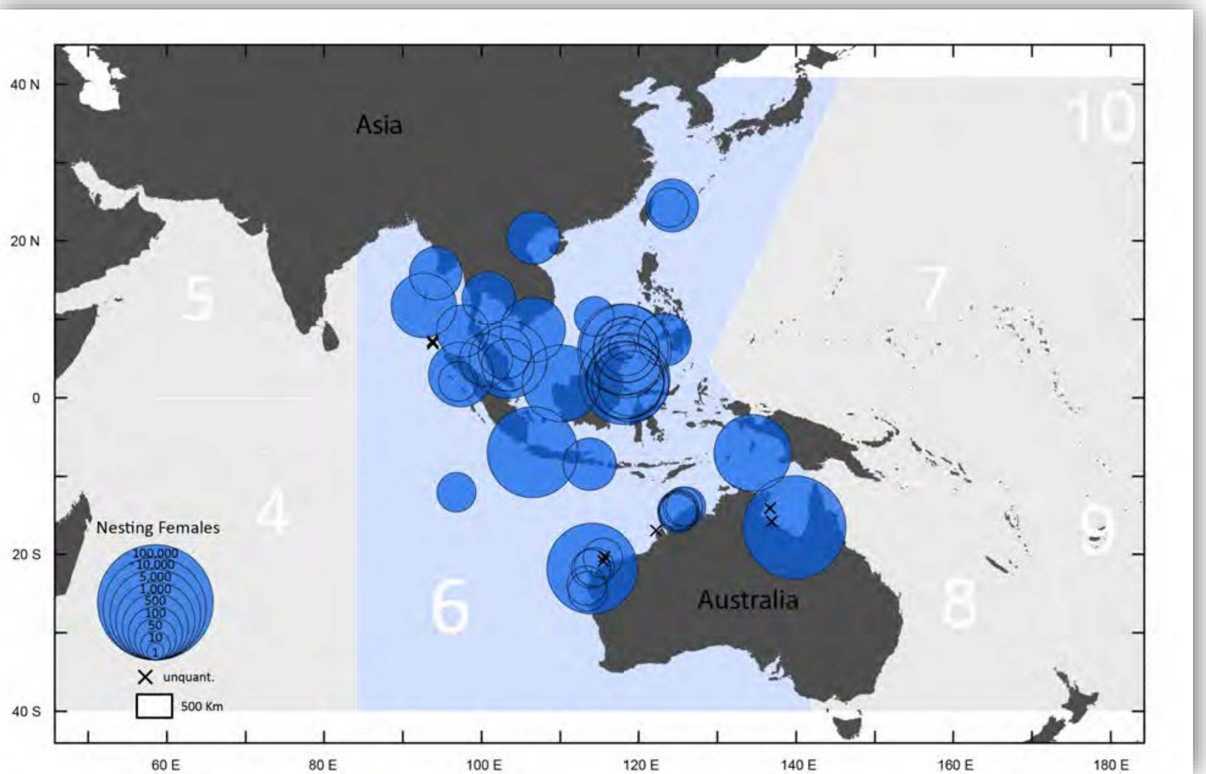


Figure 20. Nesting distribution of East Indian-West Pacific green sea turtle (blue-shaded area). Size of circles indicates estimated nester abundance. Locations marked with '·' indicate nesting sites lacking abundance information.

Status and Trends

East Indian/West Pacific green sea turtles are listed as threatened. The abundance of nesting females in this species is estimated at 77,009 females over 50 nesting sites. The largest nesting site is in northern Australia and supports about 25,000 nesting females. 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 species have relatively high probabilities of falling below 100 nesting females per year (Seminoff et al. 2015).

While there are no total abundance estimates for this species, we use the abundance estimate for the Central North Pacific green sea turtle as a proxy for our analyses. There are an estimated 3,846 Central North Pacific green sea turtle nesting females, which is considerably lower than the tens of thousands East Indian-West Pacific green sea turtle (Figure 20). IUCN Redlist categorizes the status of the Central North Pacific green sea turtle as near threatened based on a current estimate that the population size is at levels that corresponds to 83% of pre-exploitation levels at approximately 265,600 turtles or more (Chaloupka and Balazs 2007; Pilcher et al. 2012). The historic population size is estimated to be approximately 320,000 individuals (Pilcher et al. 2012). Given the higher number of East Indian-West Pacific green sea turtle nesting females compared to Central North Pacific green sea turtles, we consider 265,600 a minimum estimate of total abundance for this species.

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 species 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 species. 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 resulting from coastal development and marine pollution.

NMFS estimates that between 0 and 10% of the green sea turtles caught in the Hawaii SSLL fishery may be East Indian-West Pacific green sea turtles (P. Dutton pers. comm. November 20, 2017). This means that one East Indian-West Pacific green sea turtle may have been captured by the Hawaii SSLL fishery since 2004. NMFS estimates that between 0 and 28% of the green sea turtles caught in the Hawaii DSLL fishery may be East Indian-West Pacific green sea turtles (P.

Dutton pers. comm. June 29, 2018). This means that 0 and 19 green sea turtles captured by the Hawaii DSLL fishery since 2004 were likely East Indian-West Pacific green sea turtles.

3.3.4.8 Summary of the Status of the Green Sea Turtle

In this section of this biological opinion, we explained that Eastern 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 (Figure 21).

As previously mentioned in the green sea turtle status discussions, sea turtles face various threats throughout each stage of their respective life cycles. Our diagram of the species' population dynamics includes natural and anthropogenic threats that affect each life cycle stage (Figure 21). As you read the causal loop diagram, recall that the arrow represents the path between two variables and the sign associated with the arrow represents relationship between two connected variables (whether an increase in one variable results in an increase in the other or whether an increase in one variables results in a decrease in the other). We patterned this diagram based primarily on the dynamics of Central West Pacific and Central South Pacific green sea turtles as representative species likely to interact with the fishery. Additionally, we looked at the remaining species for any threats that may be applicable. However, as previously noted, caveats regarding the populations' trends apply to specific green sea turtle species. The Central West Pacific green sea turtle has some nesting locations which are increasing in abundance while other sites have unknown trends; however, not enough data are available on the Central South Pacific green sea turtle to identify an overall trend; and the East Indian-West Pacific green sea turtles have both increasing and decreasing abundance estimates depending on location. However, not enough information is available at this time to determine an overall trend for these particular species.

In our analysis of each green sea turtle species, we concluded most of these species 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 species appear to be increasing and that trend will tend to reinforce itself.

Our graphic illustrates the population behavior of green sea turtle adults (females), which is based on available nest count data. Because more is known about the behavior of females as opposed to males, nesting females serve as a surrogate for all of the life stages in the population. Natural threats were discussed holistically as these threats pertain to all populations of green sea turtles to varying degrees (81 FR 20057), and therefore, we illustrated some of these threats in our diagram (Figure 21). Anthropogenic threats can be direct, indirect, and may contain negative or positive associations between variables in relation to the stressor-response relationships illustrated in the casual loop. These threats are represented in our diagram by a path (arrow) that illustrates a negative relationship with the life stages they affect as represented by the negative sign at the terminus of the arrow where these threats meet the particular life stage of concern. A variety of conservation efforts have, and continue to occur, and these are represented with another arrow to the threats, and demonstrate a decrease in the threats as evidenced by the positive sign at the terminus of the arrow. Each threat to the species reduces to population to some degree, however the overall trend in the species' trajectory is positive.

While these green sea turtle species face 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 graphically represented by the diagram. Our causal loop diagram (Figure 21), which we constructed with a team of scientists, illustrates the interrelated nature of the different stages and threats and how these successive variables tend to cause or balance out impacts to the species. 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.

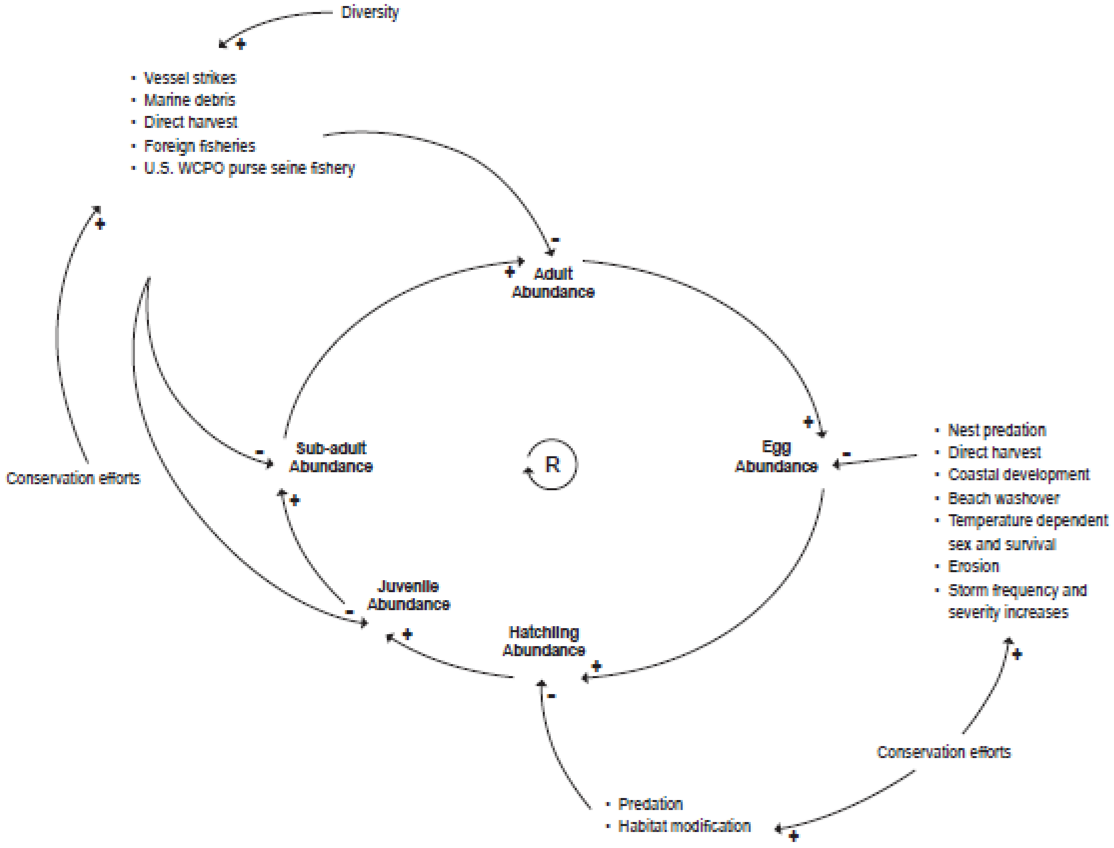


Figure 21. Green sea turtle casual loop diagram depicting various stressor-response relationships for each life cycle stage applicable to each species under consideration.

3.3.5 Hawksbill Sea Turtle

3.3.5.1 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. As with green sea turtles, 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, 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 taken 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 there are hawksbill sea turtle subpopulations 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 hawksbill sea turtles interacting with the United States WCPO purse seine fishery may be from, for the purpose of this assessment we consider all hawksbill sea turtle's interaction with the fishery to be from the West Pacific population (Banerjee et al. 2019).

3.3.5.2 Status and Trends

Hawksbill sea turtles were listed globally as endangered in 1970. Hawksbill sea turtle nesting information for nine primary locations within Oceania (excluding Hawaii) includes Great Barrier Reef, Papua New Guinea (PNG), Solomon Islands, Vanuatu, Fiji, the FSM, Republic of Palau, and the Samoan Islands (Independent Samoa and American Samoa). 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 for the last few years, with an overall downward trend (NMFS and FWS 2013a; Table 4). Much of the information on abundance and trends in Table 4 is based on anecdotal information; however, it represents the best available information on nesting numbers (NMFS and FWS 2013a).

Table 4. Summary of best currently available nesting information for hawksbill sea turtles in Oceania (NMFS and FWS 2013a).

Location	**Annual nesting females**	
	Estimate	**Trend**
Australia	4,000	Decreasing
Papua New Guinea	~500-1,000	Decreasing*
Solomon Islands	200-300	Increasing+
Vanuatu	>300	Unknown
Fiji	100-200	Decreasing*
Palau	15-25	Decreasing*
Federated States of Micronesia	~300	Decreasing*
Samoan Islands	>10-30	Decreasing*
Mariana Islands**	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).

There are currently no total population estimates for this species, however, 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). This estimate is based on the lower nester abundance level of 5,430 annual nesting females (NMFS and FWS 2013a) 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.

3.3.5.3 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).

3.3.5.4 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.

3.3.5.5 Threats to the Species

Anthropogenic climate change is likely affecting hawksbill sea turtles, although there are no published accounts or direct observations to date (NMFS 2016). 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 as cited in Witzell 1987).

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. All life stages of hawksbill sea turtles are affected by habitat destruction in the neritic/oceanic zone resulting from coastal development and marine pollution.

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. 2016), 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 increase the likelihood of boat strikes (NMFS and FWS 2007b, 2013a).

3.3.5.6 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).

3.3.5.7 Summary of the Status of the Hawksbill Sea Turtle

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 (Figure 22). Where we could, we included a graphic in our causal loop diagram to represent the behavior of the reference variables that influenced the behavior of that particular loop.

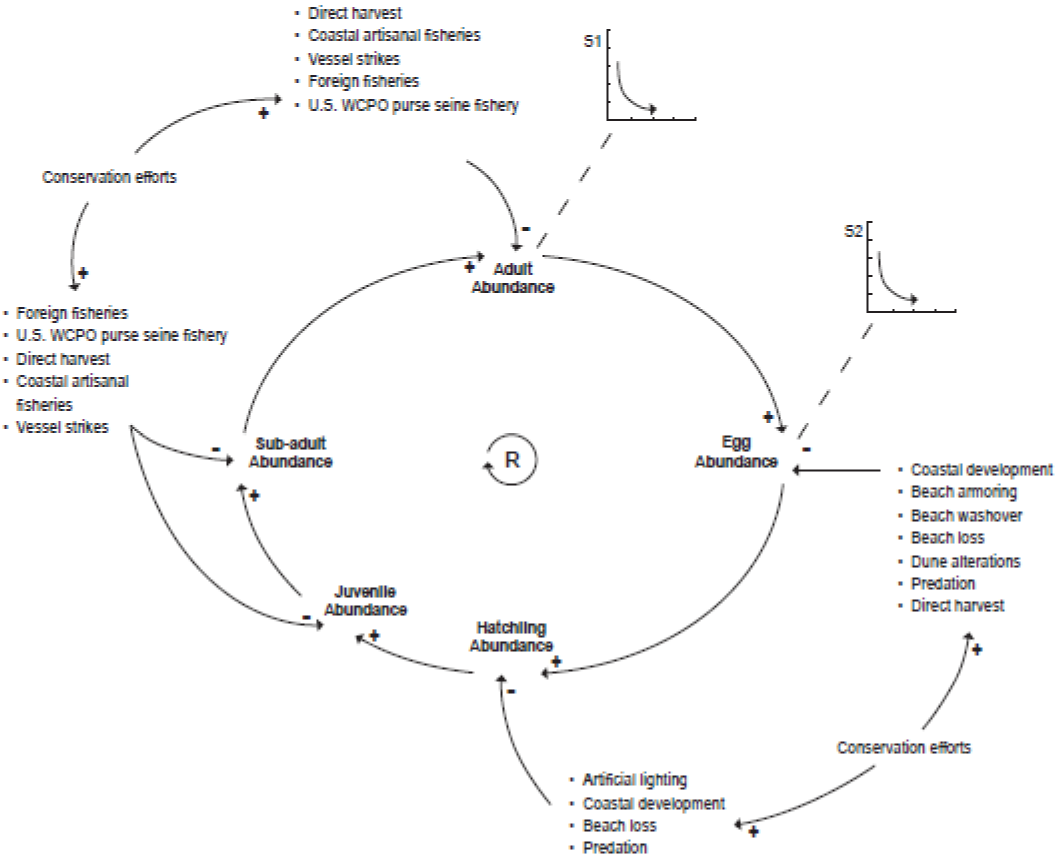


Figure 22. Hawksbill sea turtle casual loop diagram depicting various stressor-response relationships for each life cycle stage.

Our graphic illustrates the population behavior of hawksbill sea turtle adults (females), which is based on available nest count data. Because more is known about the behavior of females as opposed to males, nesting females serve as a surrogate for all of the life stages in the population. Threats can be direct, indirect, and may contain negative or positive associations between variables in relation to the stressor-response relationships illustrated in the casual loop. These threats are represented in our diagram by a path (arrow) that illustrates a negative relationship with the life stages they affect as represented by the negative sign at the terminus of the arrow where these threats meet the particular life stage of concern. A variety of conservation efforts have, and continue to occur, and these are represented with another arrow to the threats, and demonstrate a decrease in the threats as evidenced by the positive sign at the terminus of the arrow. Each threat to the species reduces to population to some degree, and while the conservation efforts mitigate these threats to a degree, the overall trend in the species' trajectory is negative.

Hawksbill sea turtles face both natural and anthropogenic threats at all life cycle stages, and currently, the number of recruits into the population does not appear to balance the loss from the various stressors as graphically represented by the diagram. Our causal loop diagram (Figure 22) illustrates the interrelated nature of the different stages and threats and how these successive variables tend to cause the negative trend apparent in Pacific hawksbill sea turtles.

3.3.6 Leatherback Sea Turtle

3.3.6.1 Distribution and Population Structure

Leatherback turtles are widely distributed throughout the oceans of the world. Leatherback sea turtles have the widest distribution of any sea turtle and occur from the equator to subpolar regions in both hemispheres with nesting on every continent except Europe and Antarctica, and several islands of the Caribbean and the Indo-Pacific (Eckert et al. 2012; NMFS and FWS 2013b; NMFS and FWS 2020). Leatherback sea turtles have been observed at sea between about 71º N to 47º S (Eckert et al. 2012). The marine distribution for Pacific leatherback turtles extends north into the Sea of Japan, northeast and east across the North Pacific to the west coast of North America (predominantly off Oregon and central 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 (Benson et al. 2011).

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 populations (35 FR 8491; Wallace et al. 2013b, Wallace et al. 2013c supplemental information; NMFS and FWS 2020). With the Western Pacific leatherback sea turtle population, we further distinguish between the nesting aggregations in Indonesia, Papua New Guinea, the Solomon Islands, and Vanuatu (Eckert et al. 2012; Tiwari et al. 2013). These nesting aggregations in the Western Pacific leatherback population are demographically-independent—the migration of individuals between nesting aggregations does not appear to be sufficient to prevent the aggregations from becoming extinct if their death rates exceed their birth rates—so we consider these aggregations as sub-populations. However, as we do not have sufficient data to ascertain the sub-population(s) origin of leatherback sea turtles interacting with the fishery, our analysis does not extend to subpopulations.

Only leatherback sea turtles from the West Pacific-based population are considered at risk of exposure to the United States WCPO purse seine fishery. Leatherback sea turtles from the East Pacific Ocean population are not considered likely to be captured in the United States WCPO purse seine fishery because: (1) there are very few captures of leatherback sea turtles in the United States WCPO purse seine fishery and all of the observed interactions occurred west of 169º W; 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* (Willis-Norton et al. 2015; NMFS 2017a) and (2) all of the turtles that have been captured by the ASLL fishery (overlapping *Action Areas*) have been from the west Pacific population based on genetic analysis. 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)

Two life history strategies are documented in the West Pacific Ocean population: winter boreal nesters and summer boreal nesters. The Papua Barat, Indonesia, nesting aggregation is composed of both life history strategies, and nesting is documented year round. Based on Benson et al. (2011), summer nesting aggregations migrate into the temperate North Pacific Ocean through the

Action Area, south of the *Action Area*, and into tropical waters of the South China Sea while winter nesting aggregations migrate into the southern hemisphere (see Figure 1 in Benson et al. 2011). What appears to be a latitudinal cline in the occurrence of life history strategies may however, be a result of limited samples sizes in tracking nesting females. Only the foraging grounds off central California have been extensively studied to date (Benson et al. 2020). There may be year-round nesting in other places in the West Pacific, including the Solomon Islands and Papua-New Guinea.

In addition, satellite tracking and stable isotope analysis suggests that West Pacific Ocean leatherbacks appear to form discrete foraging ‘‘units’’ (Benson et al. 2011; Seminoff et al. 2012). For the West Pacific population, seven ecoregions have been identified as important seasonal foraging areas: South China/Sulu and Sulawesi Seas, Indonesian Seas, East Australian Current Extension, Tasman Front, Kuroshio Extension, equatorial Eastern Pacific, and California Current Extension (Benson et al. 2011). Within these regions, leatherbacks spent long periods transiting between widely dispersed areas, which suggests movement between patches of high-density prey that aggregate leatherback sea turtles for periods of several months (Bailey et al. 2008, 2012b; Benson et al. 2020).

3.3.6.2 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 adult females and 260 nests), they classified the western Pacific leatherback turtle population as critically endangered (Tiwari et al. 2013; Wallace et al. 2013b).

The leatherback status review (NMFS and USFWS 2020) conservatively estimates adult female abundance at 1,277 individuals. This value is based only on nesting at Jamursba Medi and Wermon beaches in Papua Barat, Indonesia as these are the only beaches with long-term modeling. These two beaches likely represent between 50 and 75% of all nesting for this population (NMFS and USFWS 2020). To assist with analysis in the Hawaii-based shallow-set longline fishery biological opinion (NMFS 2019b), NMFS conducted a population viability analysis (PVA) on West Pacific leatherback sea turtles (Martin et al. 2020). They used the same data as the status review (NMFS and USFWS 2020) from the Jamursba Medi and Wermon index beaches, and used Bayesian models to impute missing data and to estimate the range of intrinsic population growth rates or *r*. Drawing from the resulting distribution of *r* values as well as the distributions of the nesting female population size at the end of the time series and a process error term, Martin et al. (2020) projected population trends 100 years into the future, conducting 10,000 simulations to capture the variability in projections. In the future projections, Martin et al. (2020) computed the proportion of simulations for which the projected number of annual nesters fell below (and remained below) 50%, 25%, and 12.5% of the estimated abundance of nesters at the end of the historic time series by the end of the 100-year projection. For the set of runs ending below a threshold, they calculated the mean, median, and 95% credible interval for the number of years until the population fell below the threshold. They also calculated the probability of the projected total reproductive females falling below each threshold at 5, 10, 25, 50, and 100 years in the future.

Results of the PVA model suggest that the adult female portion of the West Pacific leatherback sea turtle population is declining at a rate of 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 current abundance in about 13 years, and as few as five years (range 5-26 years, mean 12.7 years; Martin et al. 2020). PVA modeled estimates suggest the modeled population presently consists of about 790 adult female leatherback sea turtles (95% CI: 666-942) using the median values for nest counts. Using the lower 95% CI for nest counts resulted in a mean of 515 (95% CI 425-634) adult females.

The index of total nesting females in Jamursba Medi and Wermon (1,277 females) provided in the status review of the species (NMFS and USFWS 2020) was based on a simple calculation that does not provide confidence or credible intervals. While NMFS and USFWS (2020) determined that this index was a suitable representation of total nesting female abundance for their purposes (i.e., evaluating extinction risk), they acknowledged that the degree to which the index represents the actual abundance of nesting females is unknown. We consider the values from Martin et al. (2020) to be the best available estimates for abundance for two reasons. First, Martin et al. (2020) imputed missing data for months during which data were not collected, providing a more accurate estimate of total nesting. Second, their model evaluated variation due to natural causes (i.e., changes in nesting over time due to environmental or demographic factors) and observational error (i.e., imperfect data collection; Martin et al. 2020).

As noted above, these two nesting beaches represent approximately 50-75% of the total Western Pacific population. Assuming a 3:1 ratio of females to males and based on NMFS' PVA results for median nest counts, the total number of adult leatherback sea turtles in the West Pacific Ocean population would be 1,580 (95% CI: 1,332-1,884) if the index beaches represent 50% of the population and 1,317 (95% CI: 1,110-1,570) if the index beaches represent 75% of the population.

Assumptions of the Martin et al. (2020) PVA model include:

Imputation of missing data:

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. We used an autoregressive model with a lag of one month (AR1 model) where the relationship between the numbers of nests in two months is modeled by the Fourier series. We assume this model sufficiently captures the cyclical nature of nesting throughout the year.

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 Allee effects in the timeframe considered).

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

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

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

8. The remigration interval is constant through time.

Projections:

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

Jones et al. (2012) estimated that there were 6,199 (4,292 to 8,103) adult leatherback sea turtles in the Pacific in 2004. They assumed 85% of Pacific leatherback sea turtles were from the West Pacific, and that there was a 50% sex ratio, therefore, based on those numbers, there would have been approximately 2,600 (1,800-3,400) mature female leatherback sea turtles in the West Pacific Ocean population in 2004. Jones et al. (2012) also estimated that the total West Pacific leatherback sea turtle population size was approximately 250,000 (95% confidence interval 97,000 to 535,000). Martin et al. (2020) estimates that there were 790 (95% CI 666-942) adult females using the Jamursba Medi and Wermon index beaches in 2017. Assuming these beaches represent 75% of the total nesting for West Pacific leatherbacks suggests a total of 1,053 (888-1,256) adult female West Pacific leatherback sea turtles, representing a 51 to 64% decline from 2004 to 2017. We used these proportional changes in the estimates derived from the information contained in Jones et al. (2012) and more recent Martin et al. (2020) to estimate the current population size of the West Pacific Ocean leatherback sea turtle. The total West Pacific Ocean population is comprised of about 100,000 leatherback sea turtles but may range between 47,000 and 195,000 individuals.

3.3.6.3 Population Dynamics

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

The age at which leatherback sea turtle mature is uncertain. Estimates range widely between 9-15 years and 26-32 years depending on the data being analyzed (Dutton et al. 2005; Jones et al. 2011; Avens et al. 2020). Estimates based on the growth curves of turtles kept in controlled thermal and trophic conditions suggested size at maturity could be reached in 7 to 16 years (Jones et al. 2011). Average remigration intervals also vary, but range from 2.3 to 4 years and can vary with the foraging habitat used by adult females (Lontoh 2014). Clutch frequency ranges from 3 to 10 with a mean of 5.5 nests per female (Lontoh 2014). Clutch sizes also vary widely depending on geography and when the data were collected, but range from 40 to 120 eggs per nest at Jamursba Medi (Lontoh 2014).

This summary only provides a partial picture of the dynamics of leatherback turtle populations. The data available suggest that a hatchling that emerges in 2019 would require between 9 and 32 years to mature. However, empirical data on how long it takes for a hatchling to become a juvenile, a juvenile to become a sub-adult, and a sub-adult to mature. We also know very little about stage-specific survival rates, which would make it possible to assess the effects of removing juveniles or sub-adults from leatherback turtle populations. Nevertheless, the population dynamics suggest that the abundance of adult female turtles on nesting beaches integrates between one and three decades of demographic change and environmental influences.

3.3.6.4 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. 1988). Stable isotope analysis can complement satellite data of leatherback sea turtle movements and identify important foraging areas that reflect regional food webs (Seminoff et al. 2012).

3.3.6.5 Threats to the Species

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 (Tapilatu and Tiwari 2007; Bellagio Steering Committee 2008; PLAWG 2012; NMFS and FWS 2020). Over the long-term, climate change-related impacts will likely influence biological trajectories in the future on a century scale (Parmesan and Yohe 2003).

Natural factors, including the 2004 tsunami in the Indian Ocean (see detailed report by Hamann et al. 2006) and the tsunami that affected Japan in 2011, may have impacted leatherback nesting beach habitat through encroachment and erosion (2004 tsunami) or may have resulted in increased debris into leatherback marine habitat (e.g., impacting migratory routes and foraging hotspots).

Predation on eggs by feral pigs, feral dogs, and monitor lizards have been documents with feral pig predation being the most detrimental (NMFS and FWS 2020). Predation of nesting females by crocodiles has been documented at Jamursba Medi (NMFS and FWS 2020).

Major anthropogenic threats to the species are fisheries bycatch, direct harvest, alteration of nesting habitat, and predation, which are briefly described below (NOAA 2016). In addition, habitat changes attributed to changing environmental conditions (i.e., sand temperatures that result in mortality or changes in sex ratios, erosion), pollution and marine debris are also threats to this species (Tiwari et al. 2013; NOAA 2016).

The drivers of the decline of the West Pacific leatherback population—both anthropogenic (e.g. bycatch, egg harvest, exploitation of females) as well as environmental (e.g. lethal sand temperatures, predation, erosion) – have been described in detail (Eckert 1993; Bellagio Steering Committee 2008; Tapilatu and Tiwari 2007; Tapilatu et al. 2013; NMFS and FWS 2020). Egg harvest and exploitation of females have been minimized at the two most significant nesting beaches of Papua Barat, Indonesia, and the impact of environmental factors is being addressed through a science-based management and conservation programme. Fisheries bycatch is still considered the major obstacle to this population's recovery (Benson et al. 2011; Bailey et al. 2012a; Tapilatu et al. 2013; Wallace et al. 2013a; NMFS and FWS 2020).

Primary impacts to the West Pacific Ocean population in addition to United States commercial longline fisheries include: fishery interactions with international fleets within the Sulu Sulawesi and South China Seas and North Pacific Ocean (Roe et al. 2014), direct harvest of eggs and turtles, nest predation by feral animals (e.g., pigs and dogs), coastal development and village sprawl, coastal fishery impacts, beach erosion, low hatch success, marine debris entanglement and ingestion, and climate change (Benson et al. 2011; NMFS and FWS 2020). From the mid-1990s through 2001, the Hawaii longline fishery was estimated to capture about 110 leatherback sea turtles annually, with 35 of those considered mortalities (McCracken 2000; NMFS and FWS 2013b).

Since 2002 the Hawaii-based shallow-set longline and deep-set longline fisheries together have captured an estimated 271 leatherback sea turtles (105 observed in the shallow-set; 35 observed and 131 estimated in the deep-set) with an estimated 84 individuals dead as a result of capture (10 observed mortalities in the deep-set, the remaining estimated from post-interaction mortality rates based on injury at release). All of the leatherback sea turtles captured and sampled, except two, are from the West Pacific Ocean population (*n* = 60 for the shallow-set; *n* = 17 for the deep-set; Dutton pers. comm. 2018). Based on the lengths of leatherback turtles captured in the Hawaii-based shallow-set longline fishery (120 to 180 cm straight carapace length or SCL), that fishery interacts primarily with adult turtles. In contrast, leatherback turtles captured in the deep-set longline fishery have measured between 64 and 88 cm SCL, which suggests the fishery interacts with juvenile and subadult leatherback turtles.

Between 2006, when the observer program started in American Samoa, and 2018 the ASLL fishery has had an estimated 55 interactions with leatherback sea turtles, of which 38 were presumed mortalities based on the post-release mortality rates of Ryder et al. (2006) (NMFS 2019a). Six samples have been analyzed and all are from the West Pacific Ocean population (Dutton pers. comm. July 5, 2018). Of the ten observed in the fishery, half were between 57.5 SCL and 80.5 SCL, which are the juvenile and subadult stages, and the three larger are in the subadult to adult life stages and ranged from 106.7 cm- 150 cm SCL. The gender was not determined for any of the turtles captured.

The cumulative bycatch estimates for the United States California drift gillnet fishery (operating primarily in central California/Oregon waters) over 20-year period (1990 to 2009) was 104–242 leatherbacks (52–153 deaths; Martin et al. 2015). Genetic analyses indicated that almost all originated from the West Pacific population (NMFS SWFSC unpublished). In 2001, a Pacific Leatherback Conservation Area was implemented through regulations, which prohibited the California drift gillnet fishery from operating in central California, north to Oregon waters from August 15th through November 15th. Since the closure has been in place (nearly 20 years), only two leatherbacks have been observed captured, both released alive (NMFS-WCR Observer Program data). Genetic analysis indicated almost all of these turtles originated from the West Pacific Ocean population (Dutton et al. 2000).

In addition, nine fixed gear fisheries operate off the United States west coast, including the federally-managed sablefish pot fishery and the state-managed California Dungeness crab fishery. Since 2008, only one leatherback interaction has been documented in the sablefish fishery (NMFS 2013a). The California Dungeness crab fishery may be a new emerging threat with two documented leatherback interactions occurring in 2015 and 2016. Dungeness crab fishing effort is high, and the fishery has shifted into the Central California region, which overlaps somewhat with leatherback foraging habitat (S. Benson, NMFS, pers. comm. 2018).

Although the causes for decline of the West Pacific Ocean leatherback sea turtle populations are not entirely clear, it is likely the result of historic intensive egg and turtle harvest on nesting beaches and in foraging habitats, incidental capture of adults and juveniles in fisheries, and natural fluctuations due to changing environmental conditions that influence prey abundance and distribution (Wetherall et al. 1993; Sarti Martinez et al. 2007; Santidrian Tomillo et al. 2007, 2008; Wallace et al. 2010b; Saba et al. 2012; WWF 2018). High levels of bycatch in the pelagic drift gillnet fisheries, which operated in the Central North Pacific in the late 1980s and early 1990s and resulted in potentially thousands of leatherback turtle interactions cannot be discounted (Wetherall et al. 1993). Egg consumption by humans and domestic animals (e.g., dogs) persist on nesting beaches where protection is incomplete or projects do not exist (Tapilatu and Tiwari 2007; Urteaga et al. 2012; NMFS and FWS 2013b; Pilcher 2013; NMFS and FWS 2020). Direct harvest and fisheries bycatch are still considered major obstacles to population recovery (Wallace and Saba 2009; MTSG 2012; NMFS and FWS 2013b; NMFS and FWS 2020).

For nesting leatherbacks from the West Pacific 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).

The 2004 management measures introduced to the Hawaii shallow set longline fishery demonstrably reduced leatherback sea turtle interaction rates by 83% (Gilman et al. 2007; WPRFMC 2009). Since the Hawaii-based shallow-set longline fishery re-opened in 2004, 21 leatherback sea turtles have been estimated to have been killed in the fishery (NMFS 2019b). 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 (2001) regulations implemented a large Central California time/area closure that significantly reduced leatherback interactions in the California drift gillnet fishery by approximately 80%. NMFS has continued to refine management measures in this fishery and in 2013 NMFS exempted take of three interactions annually, or up to 10 leatherback interactions (or seven mortalities) over a five-year period (NMFS 2013b; NMFS and USFWS 2020). NMFS (2020; 85 FR 7246) management measures would include limits ("hard cap") on interactions with the California/Oregon drift gillnet fishery whereby the fishery would cease immediately if two leatherback turtle interactions occur in a two-year period.

There are around nine fixed gear fisheries operating off the United States west coast, which include the California state-managed Dungeness crab fishery and the federally managed sablefish pot fishery. Interactions in the sablefish pot/trap fishery is managed by NMFS (2012a); there was one documented interaction in 2008. The California Dungeness crab fishery might be a new emerging threat, which warrants additional information and investigation. Two documented interactions have occurred in 2015 and 2016, but fishing effort is high (dense) and the fishery has shifted into the Central California region which overlaps with leatherback habitat of the West Pacific Ocean population (although not 100% overlap over time and space; Benson pers. comm. from NMFS 2018g).

In Indonesia, the poaching of turtles and eggs continues, though egg harvest and exploitation of females has been minimized at Jamursba Medi and Wermon beaches due to the presence of monitoring programs and educational outreach (NMFS and FWS 2020). Before the monitoring programs, approximately 4 to 5 boats per week (from May to August) collected 10,000 to 15,000 eggs per boat at Jamursba Medi between 1980 and 1993 (Tapilatu et al. 2013). While the commercial egg harvest has been effectively eliminated since beach monitoring was established at Jamusrba Medi in 1993 (Hitipeuw et al. 2007), recent survey efforts suggest that most, if not all, sea turtle eggs (including leatherback turtles) are poached at other Bird's Head Peninsula beaches and sold in local markets (Tapilatu et al. 2017). Between 2016 and 2017, eight females were poached at Buru Island, Indonesia, (WWF 2018), and it is likely that three to five nesting females have been killed annually over the past two decades (J. Wang, NMFS, pers. comm., 2018). In 2017, 114 of 203 leatherback nests were harvested at Buru Island (WWF 2018). In 2018, due to education provided by the newly established WWF program on Buru Island, local community-based efforts in four villages now prohibit adult female and egg harvest. Indonesian laws prohibit the harvest of sea turtles and eggs, however there is very little enforcement of these laws in areas where monitoring programs to not exist (NMFS and FWS 2020)

In addition to the harvest of females and their nests, foraging leatherback turtles are harvested in the waters of the Kei Kecil Islands, Maluku Province, Indonesia. In this location, an indigenous and subsistence harvest of juvenile, subadult and adult turtles (52-204 cm CCL) occurs and has likely been a key feature of the local traditional culture for centuries (Compost 1980; Hitipeuw and Lawalata 2006, 2008; IOSEA 2013). Customary law ("*hak adat*") authorizes the ritual turtle hunt in the nine villages of the traditional kingdom of the Nafit people living within the Kei Kecil Islands. Suarez and Starbird (1995) brought attention to this hunt when they reported that approximately 200 animals were harpooned in just three months of 1994, with as many as 13 killed in one day. Over the past three decades, efforts to monitor harvest in these areas have been sporadic. Such efforts have resulted in estimates of up to 100 individuals killed per year (Suarez and Starbird 1996; Hitipeuw and Lawalata 2008). At one point, it was assumed that harvest pressure may have been on the decline and possibly no longer an issue (NMFS and FWS 2013b);

however, recent surveys indicate that harvest continues with conservative estimates of 431 mortalities over the past 8 years (53.9/yr.) and at least 103 leatherbacks harvested in 2017 (Hitipeuw and Lawalata 2008; WWF 2018). The size range of reported leatherback harvests ranges from 52-204 cm CCL, with 45% identified as females, 23% as males, and 32% unidentified; the report notes that better training is needed to determine gender so the information on gender should be used with caution (Hitipeuw and Lawalata 2006).

3.3.6.6 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 federally managed fisheries to monitor and 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, lowering nets deeper into the water column), and developing and promoting Sea Turtle Handling Guidelines (NMFS and FWS 2013b). For example, switching to large circle hooks and mackerel 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 at al. 2017). As part of its "Species in the Spotlight" initiative, which provides immediate, targeted efforts to halt declines and stabilize populations, focus resources within and outside of NOAA on the most at-risk species, 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).

Community-based conservation projects in Wermon and Jamursba Medi in Papua, Barat, PNG, Solomon Islands, and Vanuatu have been developed that monitor nesting and protect nests from harvest and predation, increasing the production of hatchlings from these nesting areas.

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 a resolution (CMM 2008-03) to mitigate the impacts on turtles from longline swordfish fisheries in the Western Central Pacific Ocean. In 2018, the WCPFC CMM 2008-03 was revised to CMM 2018-04 to mitigate the impacts of the purse seine fisheries on sea turtles.

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).

3.3.6.7 Summary of the Status of the Leatherback Sea Turtle

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 United States WCPO purse seine fishery will only interact with the West Pacific Ocean population. The West Pacific Ocean leatherback sea turtle is estimated to be declining at a rate of approximately 6% per year (Martin et al. 2020). Martin et al. (2020) estimate that there are approximately 790 adult females at the two primary nesting beaches in Indonesia, which represent approximately 50-75% of the population.

We constructed a diagram of the primary factors that affect the leatherback sea turtle's population dynamics (as illustrated in Figure 23), and used this as a point of discussion within our analytical team. Generally speaking, leatherback sea turtles face various threats throughout each stage of their respective life cycles. Our diagram of the species' population dynamics includes natural and anthropogenic threats that affect each life cycle stage. We patterned this diagram based on the apparent dynamics of the West Pacific Ocean population of the leatherback sea turtle. Those dynamics are consistent with the status and trend of the count data: leatherback sea turtles are declining and that decline will tend to reinforce itself.

Despite these conservation efforts, the overall trend of the leatherback sea turtle continues to decline. The Pacific Ocean populations are asymptotically declining populations. More than 25 years of conservation efforts have not yet reversed this trend. Our causal loop diagram, which we constructed with a team of scientists, illustrates the interrelated nature of the different stages and threats and how these successive variables tend to cause or reinforce the negative trend apparent in the leatherback sea turtle.

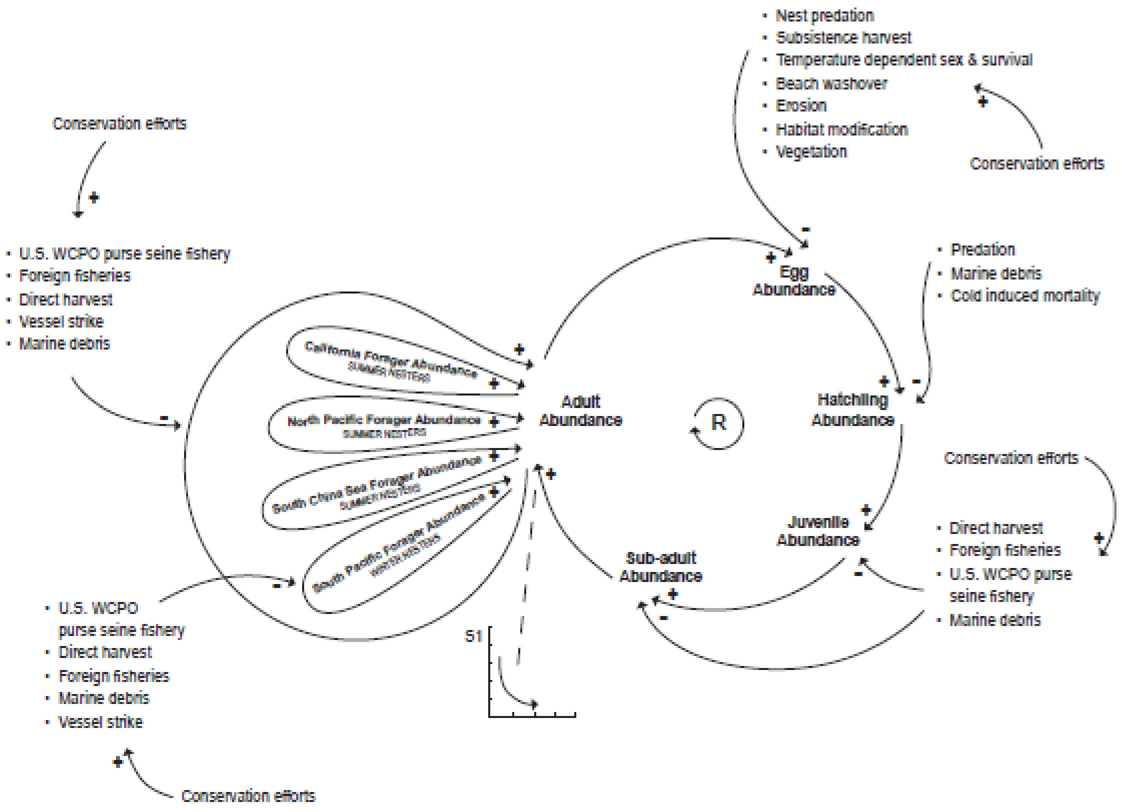


Figure 23. Leatherback sea turtle casual loop diagram depicting various stressor-response relationships for each life cycle stage.

3.3.7 South Pacific Loggerhead Sea Turtle

3.3.7.1 Distribution and Population Structure

South Pacific loggerhead sea turtles can be found throughout tropical to temperate waters in the South Pacific; however, their breeding grounds include a restricted number of sites. In the Pacific, extensive mtDNA studies show that the southern Pacific loggerhead populations are isolated from the northern Pacific populations (Hatase et al. 2002; Dutton 2007; Conant et al. 2009). They are the only species of loggerhead sea turtle in the *Action Area* (76 FR 58868).

All loggerhead sea turtles inhabiting the South Pacific Ocean are derived from beaches in Eastern Australia and a lesser-known number of beaches in southern New Caledonia (Limpus and Limpus 2003b; Limpus and Fien 2009). Nesting may also occur in Vanuatu, and Tokelau, however nesting in these locations is considered sporadic and only occurs in low densities (Limpus and Limpus 2003b; Boyle et al. 2009; Limpus and Fien 2009). There is no indication from genetic studies that the loggerhead sea turtles nesting in eastern Australia are distinct from those nesting in New Caledonia (Boyle et al. 2009), therefore we do not consider any subpopulations within the South Pacific loggerhead sea turtle in this assessment.

In Australia, there are both major and minor loggerhead sea turtle nesting rookeries (Limpus and Fien 2009), but long-term counts of nesting females are available for 7 index nesting sites that are all located in Queensland (Woongarra Coast, Heron Island, Wreck Island, Tyron Island, Lady Musgrave Island, Northwest Island and Wreck Rock beaches) and have been monitored from the 1970s to 2011 (Limpus et al. 2013). Additionally, there are two primary index foraging areas that have been monitored (via annual tag-capture-recapture sampling) in eastern Australia in the Southern Great Barrier Reef at Heron and Wistari Reef from 1984-1999 (Limpus and Limpus 2003b) and Moreton Bay (Limpus and Fien 2009). The size of the annual breeding population (females only) has been monitored at numerous rookeries in Australia since 1968 (Limpus and Limpus 2003b), and these data constitute the primary measure of the current status of the species.

3.3.7.2 Status and Trends

Loggerhead sea turtles were globally listed as threatened in 1978. In 2011, the global population was separated into nine DPS (which, once listed are treated as distinct species under the ESA and in this assessment), and the South Pacific loggerhead sea turtle was listed as endangered. Studies of South Pacific loggerhead sea turtles at their foraging areas in Eastern Australia provide some information on the status of non-breeding South Pacific loggerhead sea turtles. Chaloupka and Limpus (2001) determined that the resident South Pacific loggerhead sea turtle population on coral reefs of the southern Great Barrier Reef declined at three percent per year from 1985 to the late 1990s. The observed decline was hypothesized to be a result of recruitment failure, given few anthropogenic impacts and constant high annual survivorship measured at this foraging habitat (Chaloupka and Limpus 2001). Concurrently, a decline in new recruits was measured in foraging areas (Limpus and Limpus 2003a). This decline in annual nesting numbers for South Pacific loggerhead sea turtles in eastern Australian was primarily attributed to the mortality of turtles drowning in prawn trawls of eastern and northern Australia (Limpus and Reimer 1994).

Comparable nesting surveys have not been conducted in New Caledonia; however, information from pilot surveys conducted in 2005, combined with oral history information collected, suggest that there has been a decline in loggerhead sea turtle nesting (Limpus et al. 2006). Based on data from a pilot study, only 60 to 70 loggerhead sea turtles nested on the four surveyed New Caledonia beaches during the 2004–2005 nesting season (Limpus et al. 2006). Aerial surveys of New Caledonia nesting beaches from December 2006 and January 2008 estimate that 200 females nested during this period (World Wildlife Fund 2008 as cited in Conant et al. 2009)

We used the metrics described for fecundity and adult female population size (see Section 3.3.7.3 *Population Dynamics*) to estimate the total population size of the South Pacific loggerhead population. We estimated the abundance of this population for the purposes of our analysis, exclusive of hatchlings, to be 751,000 based on the following information: a total of 770 nesting females (Limpus et al. 2006, 2013), a remigration interval of 3.8 years (Limpus 2008a), average of 4 nests per year, with 125 eggs per nest, a nest survival rate of 0.95, a sex ratio of 1:1, first year survival of 0.675, small juvenile survival of 0.667, and large juvenile survival of 0.859, and lastly 27 years to reach maturity (metrics from Chaloupka 2003 unless otherwise noted). While we estimated 365,750 hatchlings are born in a year, due to the high rate of mortality, we used juveniles and adults for our analysis and is considered a minimum.

3.3.7.3 Population Dynamics

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

The age at which loggerhead sea turtle mature is uncertain. Avens et al. (2015) suggest a minimum age between 22.5 and 25 years and mean age between 36 and 39 years for Northwest Atlantic loggerhead sea turtles while Turner-Tomaszewicz et al. (2015) found a mean of 25 years for North Pacific loggerhead sea turtles. There are no age at maturity estimates specific the South Pacific loggerhead sea turtles. Remigration intervals vary, but average 3.8 years (range 0 to 10 years) in Australia (Limpus and Limpus 2003a). Females lay one to six clutches in a nesting year, with an average of 124 eggs per clutch. Adult female loggerhead sea turtles display high fidelity to both foraging areas and nesting beaches (Limpus and Limpus 2003a).

3.3.7.4 Diving and Social Behavior

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

3.3.7.5 Threats to the Species

Major shifts in oceanic currents is a natural threat to loggerheads in the South Pacific. The population is substantially impacted by periodic environmental perturbations such as the El Nino Southern Oscillation (ENSO). This 3- to 6-year cycle within the coupled ocean-atmosphere system of the tropical Pacific brings increased surface water temperatures and lower primary productivity, both of which have profound biological consequences (Chavez et al. 1999). Loggerheads are presumably adversely impacted by the lower food availability that often results from ENSO events, although data on this subject are lacking. Although ENSO may last for only short periods and thus not have a long-term effect on loggerheads in the region, recent studies by Chaloupka et al. (2008b) suggest that long-term increases in sea surface temperature within the South Pacific may influence the ability of the Australian nesting population to recover from historic population declines. They found an inverse relationship between sea surface temperature and annual nesting abundance.

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

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. Foxes, feral dogs and monitor lizards predate nests (Limpus et al. 2006; Limpus 2008a; Lei and Booth 2017). When they are in coastal or marine waters, loggerhead turtles are affected by a completely different set of human activities that include discharges of toxic chemicals and other pollutants into the marine ecosystem; underwater explosions; hopper dredging, offshore artificial lighting; entrainment or impingement in power plants; entanglement in marine debris; ingestion of marine debris; boat collisions; poaching, and interactions with commercial fisheries. Of these, interactions with fisheries represents a primary threat because of number of individuals that are captured and killed in fishing gear each year.

Loggerhead sea turtles are also captured and killed in commercial fisheries. In the Southeastern Pacific, significant bycatch has been reported in artisanal gillnet and longline shark and mahi fisheries operating out of Peru (Kelez et al. 2003; Alfaro-Shigueto et al. 2011) and, to a lesser extent, Chile (Donoso and Dutton 2006). In the southwestern Pacific, bottom-trawling gear has been a contributing factor to the decline in the eastern Australian loggerhead sea turtle population (Limpus and Reimer 1994). Loggerhead sea turtles have also been and are expected to continue to be captured and killed in the deep-set longline fisheries based out of American Samoa. No other United States fishery is anticipated to interact with South Pacific loggerhead sea turtles.

Studies of stranding records in Australia indicate that at least 65 sea turtles were killed annually at a result of vessel collisions between 1999 and 2002 (Hazel and Gyuris 2006). Loggerhead turtles were the second-most common species to be killed, and 72% were adults or sub-adults.

3.3.7.6 Conservation of the Species

Considerable effort has been made since the 1980s to document and reduce loggerhead sea turtle bycatch in Pacific Ocean fisheries, as this is the highest conservation priority for the species. Observer programs have been implemented in federally managed fisheries to collect bycatch data, and several strategies have been pursued to reduce both bycatch rates and post-hooking mortality. In Pacific Ocean fisheries these include developing gear solutions to prevent or reduce capture (e.g., circle hooks), implementing seasonal time-area closures to prevent fishing when turtles are congregated, modifying existing gear, and developing and promoting Sea Turtle Handling Guidelines (NOAA 2013a).

The conservation and recovery of loggerhead sea turtles is facilitated by a number of regulatory mechanisms at international, regional, national, and local levels, such as the IUCN FAO Technical Consultation on Sea Turtle-Fishery Interactions, the Inter-American Convention for the Protection and Conservation of Sea Turtles (IAC), CITES, and others. As an Appendix I species under CITES. 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.

3.3.7.7 Summary of the Status of the South Pacific Loggerhead Sea Turtle

In this section of this biological opinion, we explained that the South Pacific loggerhead sea turtle is endangered, and that the species' trend appears to be decreasing in abundance. We used our knowledge of the species' demography and population ecology to capture the primary factors that appear to determine the loggerhead sea turtle's population dynamics (Figure 24). Where we could, we included a graphic in our causal loop diagram to represent the behavior of the reference variables (data) that influenced the behavior of that particular loop.

As previously mentioned in the leatherback status discussion, sea turtles face various threats throughout each stage of their respective life cycles. Our diagram of the species' population dynamics includes natural and anthropogenic threats that affect each life cycle stage. We patterned this diagram based on the apparent dynamics of the of the South Pacific loggerhead sea turtle, however we included additional threats which may be applicable to the species based on the literature review. Those dynamics are consistent with the status and trend of the count data: South Pacific loggerhead sea turtles appear to be decreasing and that trend will tend to reinforce itself.

While this species faces both natural and anthropogenic threats at all life cycle stages, the number of recruits into the population does not appear to balance the loss from the various stressors as graphically represented by the diagram. Our causal loop diagram, which we constructed with a team of scientists, illustrates the interrelated nature of the different stages and threats and how these successive variables tend to cause or balance out to create the negative trend apparent in the South Pacific loggerhead sea turtle.

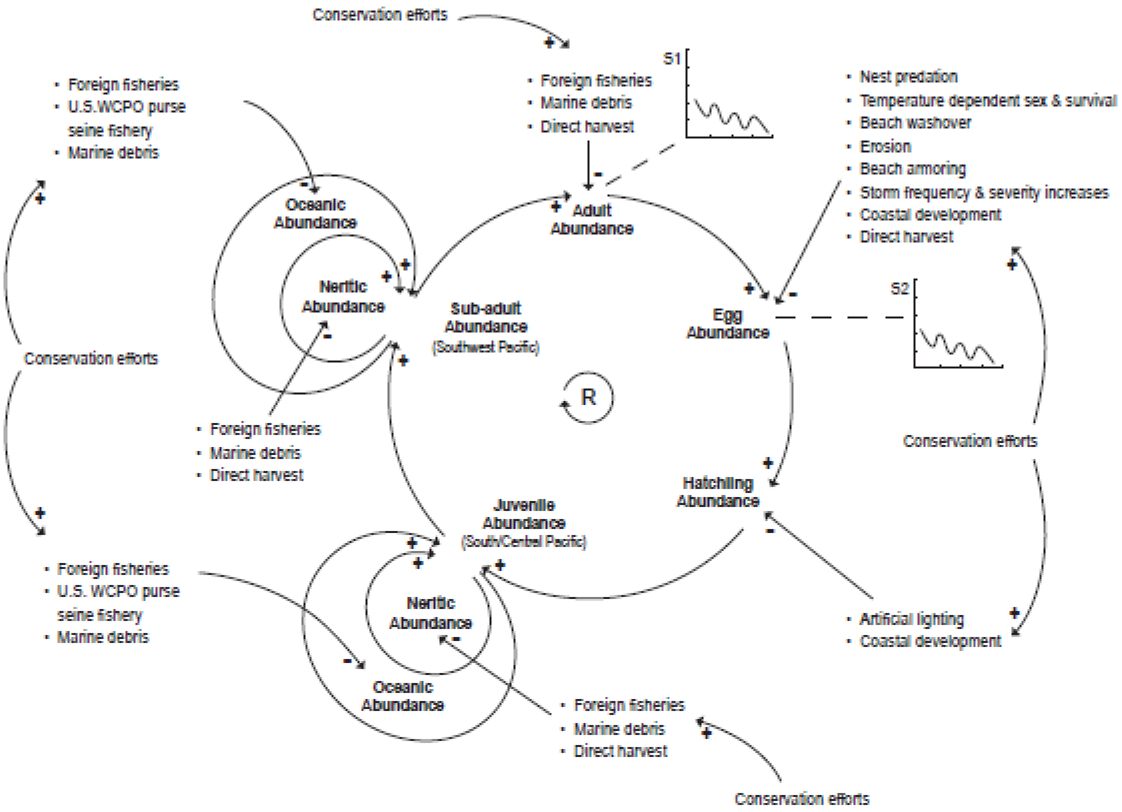


Figure 24. South Pacific loggerhead sea turtle casual loop diagram depicting various stressor-response relationships for each life cycle stage.

3.3.8 Olive Ridley Sea Turtle

3.3.8.1 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 north 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 unpublished data) suggesting that sea turtles from both populations migrate as far as the American Samoa EEZ and therefore both populations are likely to occur in the *Action Area*.

3.3.8.2 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). In-water estimates of East Pacific olive ridley sea turtles are 1,390,000 (95% CI: 1,150,000 to 1,620,000) but this estimate includes individuals from both the endangered and threatened populations (Eguchi et al. 2007).

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).

In Nicaragua, two arribada beaches still exist (Ruiz 1994): Playa Chacocente (located in the Chacocente Wildlife Refuge) and Playa La Flor (located in a private wildlife refuge; NMFS and FWS 2014). Population trends for Playa Chacocente are unknown. The nesting population at Playa La Flor is thought to be depleted but stable (Abreu-Grobois and Plotkin 2008). Overall, there are at least 1,000,000 nesting females in the threatened population in the East Pacific and therefore we consider this a minimum population estimate.

In the western Pacific, information on the size of olive ridley nesting aggregations are limited (NMFS and FWS 2014). In Indonesia, olive ridley sea turtles nest on beaches in the West Papua Province, 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). We digitized Figure 1 from (Kurniawan and Gitayana 2020; Figure 18 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 population is approximately 4,400. On Jamursba-Medi beach, 77 olive ridley sea turtle nests were documented from May to October 1999, 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). Extensive hunting and egg collection, in addition to rapid rural and urban development, have reduced nesting activities in Indonesia (Tapilatu et al. 2017). In eastern Java, olive ridley sea turtle nesting was documented from 1992-1996 that ranged from 101 to 169 nests. In Malaysia, olive ridley sea turtles nest on the eastern and western coasts; however, nesting has declined rapidly in the past decade. The highest density of nesting was once reported in Terrenganu, Malaysia, which once yielded 2,400 nests, but the populations were virtually extirpated by 1999 due to long-term over-harvest of eggs (NMFS and FWS 2014). In Australia, olive ridley sea turtle nesting is scattered throughout northern Australia, with a few thousand females nesting annually (Limpus 2008b). 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 2008b; NMFS and FWS 2014). There is no evidence to suggest that the current nesting numbers in Australia are the remnant of a population that has declined substantially within historical times (Limpus 2008b). Olive ridleys generally nest every other year (van Buskirk and Crowder 1994), therefore we estimate a minimum 6,000 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 12,000. Combining this nesting population with the Alas Purwo National Park, East Java, Indonesia nesting population suggests a minimum adult population size of 16,400 for Indo-West Pacific olive ridley sea turtles.

The once large nesting populations of olive ridley sea turtles that occurred in peninsular Malaysia and Thailand have been decimated through long term over-harvest of eggs (Limpus 2008b). The species nests in low numbers at many sites in Indonesia and is only rarely encountered nesting in the Republic of the Philippines or Papua New Guinea (Limpus 2008b).

In the Indian Ocean, arribadas occur in northeastern India in the Indian State of Odisha (formerly known as Orissa), at Gahirmatha and Ryshikulya, have estimates exceeding 700,000 turtles nesting per arribada (NMFS and FWS 2014). A number of other locations in western and eastern India are also described as sites of potential solitary nesting activity, but nesting activity is unquantified at these locations (NMFS and FWS 2014). Survey effort on Indian beaches has fluctuated over the years and methods used to census nesting populations have also changed. As a result, reported trends and abundance numbers may be somewhat speculative and potentially unreliable. The most reliable abundance estimate for Gahirmatha during the 1999 arribada was approximately 180,000 nesting females, with long-term data indicating the population may be in decline (NMFS and FWS 2014). During the 2012 nesting season, an estimated 100,000 olive ridley sea turtles laid eggs in Orissa compared to 250,000 in 2011 (IOSEA 2013). Lower numbers of eggs are often laid following a good year of nesting. Yet this arribada (that often occurs in February) was delayed about a month, raising concerns about the influence of climate change, storms (such as the effects of Cyclone Thane that struck the Bay of Bengal December 30, 2011), fishing activity, or coastal erosion (IOSEA 2013). In contrast, there are no known arribadas of any size in the Western Pacific, and apparently, only a few hundred nests scattered across Indonesia, Thailand and Australia (Limpus 2008b). Data are not available to analyze trends (NMFS and FWS 2014).

3.3.8.3 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). For females, 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).

Olive ridley sea turtles commonly nest in successive 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).

3.3.8.4 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).

3.3.8.5 Threats to the Species

Impacts from climate change, especially due to global warming, are likely to become more apparent in future years (Intergovernmental Panel on Climate Change (IPCC) 2007; NMFS and FWS 2014). Based on the available information, climate change is an anthropogenic factor that 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 from non-United States fisheries 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). An estimated 75,000 turtles were killed each year for over two decades until 1990 when the fishery closed (Aridjis 1990; NMFS and FWS 2014). Impacts from the Mexican turtle fishery were so dramatic they have been widely referred to in the literature as population collapses, crashes, or extinctions (NMFS and FWS 2014). 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). This fishery killed several hundred thousand olive ridleys during this time (Green and Ortiz-Crespo 1982; NMFS and FWS 2014).

The nationwide ban on commercial harvest of sea turtles in Mexico, enacted in 1990, has improved the situation for the olive ridley. Surveys of important olive ridley nesting beaches in Mexico indicate increasing numbers of nesting females in recent years (Marquez et al. 1996; Cornelius et al. 2007). Annual nesting at the principal beach, Escobilla Beach, Oaxaca, Mexico, averaged 138,000 nests prior to the ban, and since the ban on harvest in 1990, annual nesting has increased to an average of 525,000 nests (Salazar et al. in press). 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 2005 and 2018 there were 686 olive ridley interactions in the Hawaii DSLL fishery and from this the estimated mortality is 650 (NMFS 2018b). 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 incidental take statement for up to 2 anticipated olive ridley interactions and 1 anticipated estimated mortality every five years (NMFS 2013a). 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).

3.3.8.6 Conservation

The conservation and recovery of olive ridley sea turtles are 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.

3.3.8.7 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 (Figure 25).

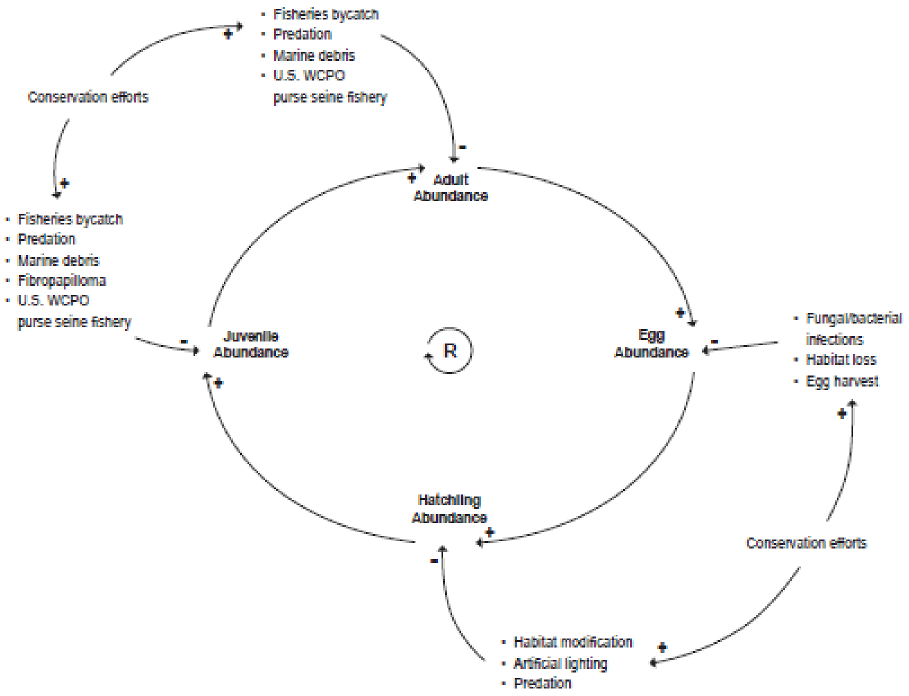


Figure 25. Olive ridley sea turtle causal loop diagram depicting various stressor-response relationships for each life cycle stage.

As previously mentioned in the leatherback status discussion, sea turtles face various threats throughout each stage of their respective life cycles. Our diagram of the species' population dynamics includes natural and anthropogenic threats that affect each life cycle stage. We patterned this diagram based on the apparent dynamics of the of the olive ridley sea turtle, however we included additional threats which may be applicable to the species based on the literature review. Those dynamics are consistent with the status and trend of the count data: olive ridley sea turtles appear to be increasing slightly and that trend will tend to reinforce itself.

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 graphically represented by the diagram. Our causal loop diagram, which we constructed with a team of scientists, illustrates the interrelated nature of the different stages and threats and how these successive variables tend to cause or balance out to create the positive trend apparent in the olive ridley sea turtle.

3.3.9 Oceanic Whitetip Shark

3.3.9.1 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 WCPO, 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; 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 26).

Only two studies have been conducted on the genetics and population structure of the oceanic whitetip shark, which suggest there may be some genetic differentiation between various populations (Camargo et al. 2016; Ruck 2016). 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). However, 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 was discovered within ocean basins. Ruck (2016) did detect weak but significant differentiation between the Atlantic and Indo-Pacific Ocean populations. An additional analysis of the sample 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).

Tagging studies have also 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 be philopatric (Howey-Jordon et al. 2013; Tolotti et al. 2015; Young and Carlson 2020) however it is not clear if this is a result of females exhibiting site fidelity to pupping areas or if the species has an underlying subpopulation structure (Young and Carlson 2020).

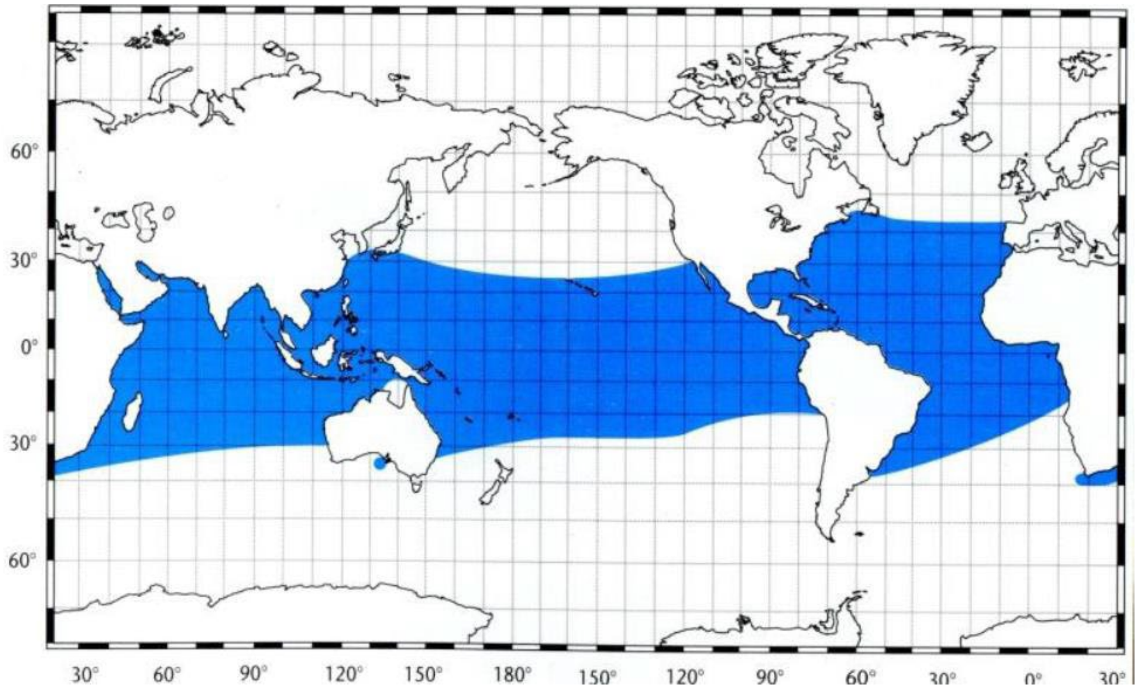


Figure 26. Geographical distribution of the oceanic whitetip shark (Last and Stevens 2009).

3.3.9.2 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).

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. 2011; Brodziak et al. 2013; Rice et al. 2015; Tremblay-Boyer et al. 2019). Rice and Harley (2012) estimated the 2010 total biomass at 7,295 metric tons. FAO (2012) suggest this represents approximately 200,000 individuals assuming and average individual body weight of 80.4 pounds (36.5 kilograms). Rice and Harley (2012) concluded that the oceanic whitetip shark is not only experiencing overfishing in the Western and Central Pacific, but the population is currently in an overfished state. Tremblay-Boyer et al. (2019) updated the stock assessment of Rice and Harley (2012). They concluded that total biomass in 2010 was 19,740 metric tons and that biomass declined to 9,641 metric tons by 2016. This resulted in a greater spawning biomass used by Tremblay-Boyer et al. (2019) for the analysis of WCPO oceanic whitetip sharks than that which was used in Rice et al. (2015). Using the same assumptions as FAO (2012), this biomass would equate to approximately 264,318 individuals. Similar to Rice and Harley (2012), Tremblay-Boyer et al. (2019) also concluded that the population is overfished and continues to undergo overfishing while using a higher spawning biomass estimate and a wider range of variables than originally considered in Rice and Harley (2012). However, Tremblay-Boyer et al. (2019) also note that the rate of overfishing has declined since the WCPFC adopted CMM 2011-04 in 2013, prohibiting the retention of oceanic whitetip sharks, in whole or in part and requiring the release of any oceanic whitetip that is caught as soon as possible after the shark is brought alongside 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. Young et al. (2017) believe CPUE may have stabilized in the last five years at a depressed state in the Hawaii-based longline fisheries. The significant declining trends observed in all available abundance indices (e.g. standardized CPUE, biomass, and median size) of oceanic whitetip sharks occurred as a result of increased fishing effort in the longline fisheries, with lesser impacts from targeted longline fishing and purse-seining (Young et al. 2017).

As noted above in Section 3.3.9.1– *Distribution and Population Structure*, it is possible that oceanic whitetip sharks are philopatric; therefore, the decreases in biomass may have resulted in localized depletions resulting in a loss of genetic diversity as well as abundance.

3.3.9.3 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 maximum age may be 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).

3.3.9.4 Diving and Social Behavior

Oceanic whitetip sharks are generally associated with mixed surface layers where temperatures typically remain greater than 20°C, but they can occur at depths of about 150 m 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 (formations or clusters of individuals which have gathered which may or may not have distinct demographic characteristics) 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.

3.3.9.5 Threats to the Species

The primary threat to oceanic whitetip sharks worldwide is incidental bycatch in commercial fisheries, including both United States and foreign 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. Impacts to the species from fisheries (United States and foreign) that overlap the *Action Area* will be discussed in the *Environmental Baseline*, as appropriate.

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

The most significant threats to the species result from the combined effect of fisheries bycatch and exploitation for the fin trade. Currently, bycatch-related mortality in longline fisheries is considered the primary driver for the population declines (Clarke et al. 2011; Rice and Harley 2012; Young et al. 2017), with purse seine fisheries being secondary sources of mortality. In addition to bycatch-related mortality, the oceanic whitetip shark is a preferred species for retention because its large fins obtain a high price in the Asian fin market, and comprises approximately 2% of the global fin trade (Clarke et al. 2006a). This high value and demand for oceanic whitetip fins incentivizes the retention and subsequent 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. As a result, oceanic whitetip biomass has declined by 88% since 1995 (Tremblay-Boyer et al. 2019). Currently, the population is overfished and overfishing is still occurring throughout much of the species' range (Rice and Harley 2012; Tremblay-Boyer et al. 2019; 85 CFR 46588). As a result, 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 (see the casual loop diagram in the *Status of Listed Resources*).

United States fisheries in the Pacific that capture oceanic whitetip sharks include the Hawaii SSLL and DSLL, and America Samoa longline fisheries. The Hawaii SSLL is estimated to interact with 102 oceanic whitetip sharks in a given year (95th percentile; NMFS 2019). The Hawaii DSLL is estimated to interact with 3,185 oceanic whitetip sharks annually (McCracken 2018; NMFS 2018b). The ASLL fishery is discussed in the *Environmental Baseline*, as that fishery overlaps the *Action Area*. No interactions have been noted with oceanic whitetip sharks in any West Coast HMS FMP fishery to date (C. Villafana and C. Fahy pers. comm. to J. Rudolph; March 7, 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 would limit their ability to recover from over-exploitation.

3.3.9.6 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 Australia, Cook Islands, Micronesia New Zealand, Palau, Republic of the Marshall Islands and Tokelau, as well as by the IATTC and the WCPFC. These finning bans range from requiring fins remain attached to the body to allowing fishermen to remove shark fins provided that the weight of the fins does not exceed 5% of the total weight of shark carcasses landed or found onboard. The WCPFC has implemented several conservation and management measures for sharks with the following objectives (Clarke 2013): (1) promote full utilization and reduce waste of sharks by controlling finning (perhaps as a means to indirectly reduce fishing mortality for sharks); (2) increase the number of sharks that are released alive (in order to reduce shark mortality); and (3) increase the amount of scientific data that is collected for use in shark stock assessments. Also, specific to oceanic whitetip sharks, CMM 2011-04 prohibits WCPFC vessels from retaining onboard, transshipping, storing on a fishing vessel, or landing any oceanic whitetip shark, in whole or in part, in the fisheries covered by the Convention.

3.3.9.7 Summary of the Status

In this section of this biological opinion, we explained that the oceanic whitetip shark is threatened, and that the species' population trend appears to be decreasing. 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 (Figure 27).

Based on the best scientific and commercial data available, we have determined the oceanic whitetip shark population appears to be decreasing at significant rates. The species is likely to become endangered within the foreseeable future throughout all or a significant portion of its range (Young et al. 2017). Overutilization, including fisheries bycatch and inadequate regulatory mechanisms are the most concerning threats that may contribute to the extinction risk of the species.

As a result of fishing mortality, oceanic whitetip biomass declined by 86% in the Western and Central Pacific. Currently, the population is overfished and overfishing is still occurring (Rice and Harley 2012; Tremblay-Boyer et al. 2019). As a result, 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, which are represented by S1 and S2 within our causal loop diagram. Similar results between analyses of SPC observer data from the larger Western and Central Pacific and the observer data from the Hawaii-based pelagic longline fishery suggest that the population 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). The significant declining trends observed in all available abundance indices (e.g., standardized CPUE, biomass, and median size) of oceanic whitetip sharks occurred as a result of increased fishing effort in the longline fisheries, with lesser impacts from targeted longline fishing and purse-seining (Rice and Harley 2012; Tremblay-Boyer et al. 2019).

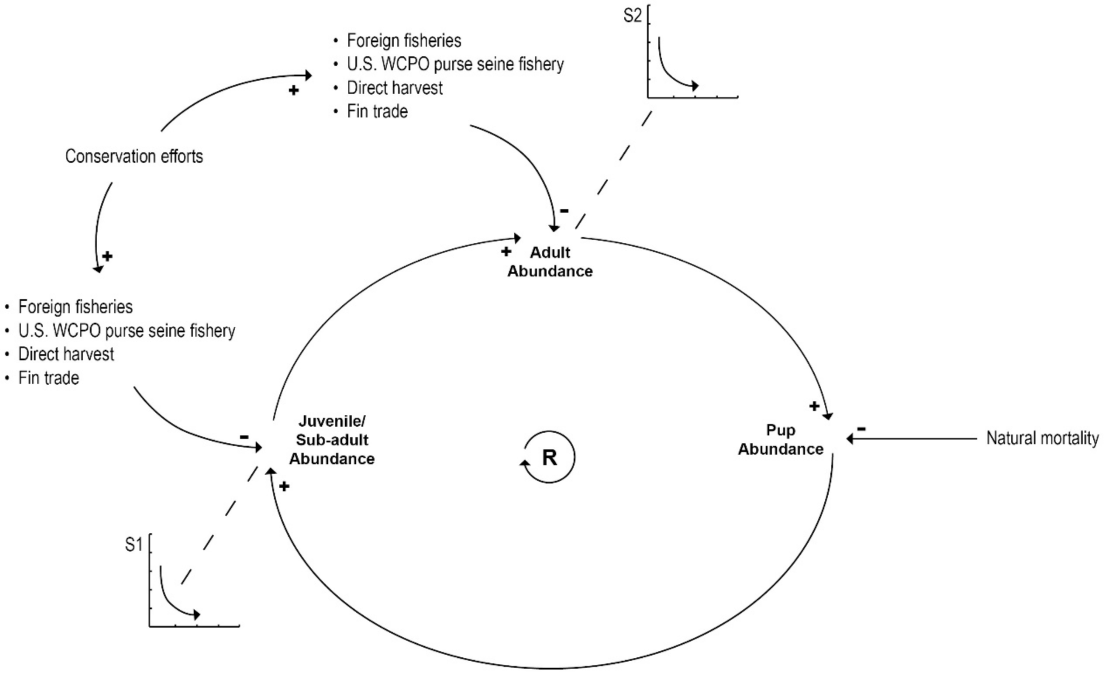


Figure 27. Oceanic whitetip shark causal loop diagram depicting various stressor-response relationships for each life cycle stage.

3.3.10 Indo-West Pacific Scalloped Hammerhead Shark

3.3.10.1 Distribution and Population Structure

Scalloped hammerhead sharks 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 species (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.

Both juveniles 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 permanent resident populations, like those in the East China Sea (Compagno 1984). There is evidence of size segregation in areas off the coast of Australia (Noriega et al. 2011) as well as in schools around Wolf Island in the Galapagos (Hearn et al. 2010). In the Gulf of California, Klimley (1985 as cited in Miller et al. 2014) reported highly polarized and aggressive schools, with females predominating and competing for positions at the center of schools. Adult scalloped hammerhead sharks are well known to aggregate over offshore seamounts and those near islands.

These sharks have been observed making migrations along continental margins as well as between oceanic islands in tropical waters (Kohler and Turner 2001; Duncan and Holland 2006; Bessudo et al. 2011; Diemer et al. 2011). Tagging studies reveal the tendency for scalloped hammerhead sharks to aggregate around and travel to and from core areas or “hot spots” within locations (Holland et al. 1993; Duncan and Holland 2006; Hearn et al. 2010; Bessudo et al. 2011), however they are also capable of traveling long distances (1941 km, Bessudo et al. 2011; 1671 km, Kohler and Turner 2001; Hearn et al. 2010; 629 km, Diemer et al. 2011).

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.

3.3.10.2 Status and Trends

Indo-west Pacific scalloped hammerhead sharks are listed as threatened because of overharvesting. Although range-wide trends in the abundance of this species are unknown, CPUE data suggest that local populations throughout the range of the species have declined significantly. For example, the hammerhead population in Australia's northwest marine region has been estimated to have declined between 58- 76%. Similarly, catch rates of scalloped hammerhead sharks 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 as cited in Miller et al. 2014; CITES 2010). CPUE data from the northern Australian shark fishery indicate declines of 58-76% in hammerhead abundance in Australia's northwest marine region from 1996-2005 (Heupel and McAuley 2007 as cited in Miller et al. 2014). Data from protective shark meshing programs off beaches in New South Wales (NSW) and Queensland also suggest significant declines in hammerhead populations 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). Similarly, data from the Queensland shark control program indicate declines of around 79% in hammerhead shark abundance between the years of 1986 and 2010, with *S. lewini* abundance fluctuating over the years but showing a recent decline of 63% between 2005 and 2010 (QLD DEEDI 2011 as cited in Miller et al. 2014).

Current effective population sizes 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 (Table 5; Miller et al. 2014).

Table 5. Estimates of current effective population size (Ne) of scalloped hammerhead sharks. (NMFS 2014a, Adapted from estimates in Duncan et al. 2006).

Ocean Basin	**Population**	**Sample Size (n)**	**Ne (5.7 year generation time)**	**Ne (16.7 year generation time)**
Pacific	Baja	44	22,000,000	7,600,000
	Pac. Panama	8	62,000,000	2,000,000
	Hawaii	44	3,200	1,100

Ocean Basin	**Population**	**Sample Size (n)**	**Ne (5.7 year generation time)**	**Ne (16.7 year generation time)**
	Philippines	15	64,000	22,000
	Taiwan	20	15,600,000	5,200,000
	E. Australia	32	70,000	24,000
Indian	W. Australia	26	6,800	22,000
	Seychelles	12	15,200	54,004
	S. Africa	25	18,000	60,010
Atlantic	W. Africa	6	300,000	100,000
	East Coast U. S.	16	36,000,000	12,000,000
All	Total	271	280,000	94,000

In order to estimate the effective population size in the Indo-West Pacific scalloped hammerhead shark from 280,000; the percentage was calculated from the areas highlighted in gray of the total listed in the table. These areas are considered part of the Indo-West pacific scalloped hammerhead's range represent about 12% of the effective population size in the table. Therefore, for purposes of this analysis the effective population size with a 5.7 generation time is 33,600 (280,000*12%) and for the 16.7-year generation time the estimate is 11,280 (94,000*12%).

Pacoureau et al. (2020) indicates a 67% decline from 1970 to 2018 equating to a 2.31% decline per year. However, Figure 5 of Pacoureau et al. (2020) suggests populations in the South Pacific and Indian Oceans (i.e. Indo West Pacific scalloped hammerheads) have stabilized at a depressed level.

3.3.10.3 Population Dynamics

Similar to the other elasmobranchs included in this biological opinion, scalloped hammerhead sharks 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).

The population size and trend of Indo-west Pacific scalloped hammerhead sharks is unknown. However, as described above we assume that Indo-west Pacific scalloped hammerhead sharks represent about 12% of the effective population size of all scalloped hammerhead sharks. Depending on whether the generation time is assumed to be 16.7 years or 5.7 years, the effective population size of Indo-west Pacific scalloped hammerhead sharks is estimated at 11,280 or 33,600, respectively (NMFS 2014a).

3.3.10.4 Diving and Social Behavior

Both juveniles 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 permanent resident populations, 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 Kāne'ohe Bay in Oahu, Hawaii, coastal waters off Oaxaca, Mexico, 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).

3.3.10.5 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. NMFS (2013) noted that significant catches of scalloped hammerheads have and continue to go unrecorded or reported in many countries outside the United States. Furthermore, NMFS (2013), 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 shark species in general, and thus species-specific population trends for scalloped hammerheads are not readily available (NMFS 2013). 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). Bottom longline fisheries for example have between 92 to 94% at vessel mortality for hammerhead shark species including scalloped hammerhead sharks, and does not include post release mortality (Morgan and Burgress 2007). Considering purse seine fisheries, while Hutchinson’s (2015) study focused on silky sharks, the study showed that sharks confined in the sack portion of the net just prior to loading suffered much higher mortality with only a 6.67% chance of survival after brailing. This highlights the consequences of restricting the movement of hammerhead shark species given their respiratory mode (i.e. obligate ram ventilation). Compared to other chondrichthyans, scalloped hammerhead sharks appear to sustain a higher level of fishing mortality (Miller et al. 2014). Miller et al. (2014) further ranked high at-vessel mortality as the most serious threat to the species.

Catches of Indo-west Pacific scalloped hammerhead sharks from non-United States fisheries have decreased since reaching a maximum of 798 tonnes in 2002 (see Figure 2 in NMFS 2013). According to shark fin traders, hammerhead sharks are one of the sources for the best quality fin needles for consumption and fetch a high commercial value in the Asian shark fin trade (Abercrombie et al. 2005). In Hong Kong, the world’s largest fin trade market, scalloped hammerhead, and smooth hammerhead sharks are found under the “Chun chi” market category, the second most traded fin category in the market (Clarke et al. 2006a). Applying a Bayesian statistical method to the Hong Kong shark fin trade data, Clarke et al. (2006b) estimated that between 1 and 3 million hammerhead sharks, with an equivalent biomass of 60 – 70 thousand metric tonnes, are traded per year.

United States fisheries appear to have less influence on this species status when compared to foreign fisheries. Fisheries in Alaska and California, and the Hawaii SSLL do not overlap the species range. However, two United States fisheries interact with the Indo-West Pacific scalloped hammerhead shark other than the WCPO purse seine fishery in the Pacific. The Hawaii-based DSLL fishery interacts with an estimated four Indo-west scalloped hammerheads per year (McCracken 2018; NMFS 2018b). While the Hawaii DSLL fishery overlaps the range of the species, it does not overlap the *Action Area*. Fishery interactions from the ASLL fishery will be discussed in the *Environmental Baseline* as that fishery occurs within the *Action Area*. Thus, the greatest influence on the decline of this species is from non- United States fisheries throughout the species range in the western Pacific.

3.3.10.6 Conservation

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

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

3.3.10.7 Summary of the Status

In this section of this biological opinion, we explained that the Indo-West Pacific scalloped hammerhead shark is threatened, and that the species' trend appears to be decreasing. 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 (Figure 28).

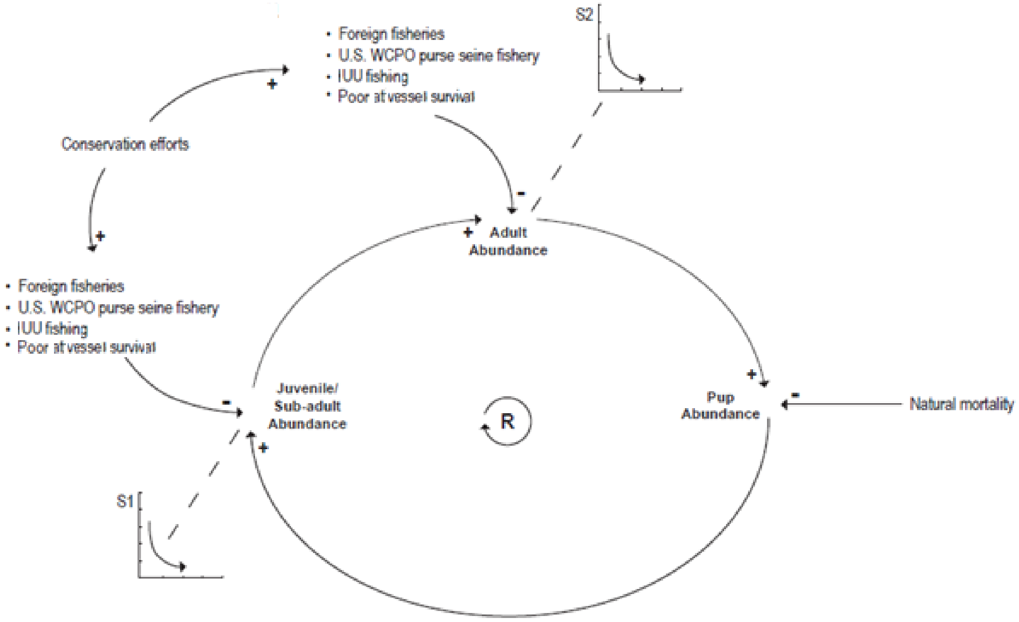


Figure 28. Indo-West Pacific scalloped hammerhead shark causal loop diagram depicting various stressor-response relationships for each life cycle stage.

Based on the best scientific and commercial data available we have determined the Indo-West Pacific scalloped hammerhead shark population appears to be decreasing at significant rates. The species is likely to become endangered within the foreseeable future throughout all or a significant portion of its range (Miller et al. 2014). Evidence of heavy fishing pressure by industrial/commercial and artisanal fisheries, and reports of significant illegal, unreported and unregulated fishing, especially off the coast of Australia, have likely led to overutilization coupled with inadequate regulatory mechanisms are the most concerning threats that may contribute to the extinction risk of the species.

As a result of this fishing mortality, the Indo-West Pacific scalloped hammerhead shark population is declining, which is represented by S1 and S2 within our causal loop diagram. The threats from fisheries are represented in our diagram by a path (arrow) that illustrates a negative relationship with the life stages they affect as represented by the negative sign at the terminus of the arrow where these threats meet the particular life stage of concern. A variety of conservation efforts have, and continue to occur, and these are represented with another arrow to the threats, and demonstrate a decrease in the threats as evidenced by the positive sign at the terminus of the arrow. Each threat to the species reduces to population to some degree, and while the conservation efforts mitigate these threats to a degree, the overall trend in the species' trajectory is negative. Our causal loop diagram (Figure 28), which we constructed with a team of scientists, illustrates the interrelated nature of the different stages and threats and how these successive variables tend to cause the negative trend apparent in Indo-West Pacific scalloped hammerhead sharks.

3.3.11 Giant Manta Ray

3.3.11.1 Distribution and Population Structure

The giant manta ray is globally distributed 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, 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 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 there are distinct morphological differences between the two species, they can be difficult to distinguish without adequate training and identification keys (Stevens et al. 2018). Therefore, correct identification to the species level is likely an issue in fisheries observer data.

Area of occupancy for giant manta rays was estimated from observations and expert opinion by Lawson et al. (2017; Figure 29). This map is not a complete representation of the species occurrence and does not depict giant manta rays in much of the WCPO 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 is likely distributed throughout this area (see Figure 63 in the *Effects of the Action* section of this biological opinion and Figures 23 and 24 in Tremblay-Boyer and Brouwer 2016).

The environmental variables that drive or are correlated with giant manta ray movements in the ocean are largely unknown (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 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. Feeding aggregations of giant manta ray have been observed off the Yucatan Peninsula of Mexico as large as 125 individuals in a single location (Trujillo-Cordova et al. 2020).

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 29).

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, though to be viable the abundance of each subpopulation likely needs to be at least 1,000 individuals (see Section 3.3.11.2; *Status and Trends*). 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. 2016; Moazzam 2018).

Several studies have tracked individual giant manta rays and provide information on the spatial extent of giant manta ray populations. Stewart et al. (2016a) studied four subpopulations of giant manta ray using genetics, stable isotopes, and satellite tags. They found that these subpopulations appeared to be discrete with no evidence of movement between them. The home ranges for three of these subpopulations (all of which are outside of the *Action Area*), defined as the areas where tagged animals were expected to spend 95% of their time encompassed areas of 79,293 km^2 (Raja Ampat, Indonesia), 70,926 km^2 (Revillagigedo Islands, Mexico), and 66,680 km^2 (Bahia de Banderas, Mexico). They suggest that their findings indicate that giant manta rays form discrete subpopulations that exhibit a high degree of residency. Stewart et al. (2016a) state that this does not preclude occasional long-distance migrations, but that these migrations are likely rare and do not generate substantial gene flow or immigration of individuals into these subpopulations. One instance of a long-distance migration has been noted in the literature. Hearn et al. (2014) tracked nine giant manta rays at Isla de la Plata, Ecuador. Eight of the nine tagged giant manta rays remained in an area of 162,500 km^2, while the ninth traveled a straight-line distance of 1,500 km to the Galapagos Islands, however, Stewart and Hearn later believed it may have been from a floating tag (J. Stewart pers. comm. to J. Rudolph, October 7, 2020).

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). 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. 2016 [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.

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.

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

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

Given the current understanding of giant manta ray population structure, for the remainder of this biological opinion, we will use the terms 'giant manta ray' or 'species' to refer to the giant manta ray as they were listed, the term 'population' to refer to the Indo-Pacific population as a whole, and 'subpopulation' to refer to independent subunits considered in this biological opinion. We note that for some of the study areas identified in Table 6 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 29. 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).

3.3.11.2 Status and Trends

NMFS listed giant manta rays globally as threatened in 2018. The IUCN lists them as vulnerable (the category that immediately precedes endangered in the IUCN classification system), with a decreasing population trend. Although the number of regional subpopulations is unknown, the sizes of those identified as regional subpopulations tend to be small, ranging from 600 to 1,875 with one recorded as containing more than 2,400 recorded individuals (CITES 2013; Marshall et al. 2018; Beale et al. 2019; Table 6). CITES (2013) highlights three giant manta ray subpopulations that have been studied and population estimates provided, and counts for more than ten aggregations (Table 6). 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. (2013a), Heinrichs et al. (2011), Lewis et al. (2015), and Fernando and Stevens (2011). The most comprehensive of these is O'Malley et al. (2013) that presents an overview of the economic value of manta ray watching tourism. They highlight 23 sites globally, and within the *Action Area* of the United States WCPO purse seine fishery, these areas include nine sites: Indonesia, Papua New Guinea, Federated States of Micronesia, Palau, Solomon Islands, Kiribati, New Caledonia, Fiji and French Polynesia. Overall, giant manta ray subpopulations appear to be regionally distinct (Lewis et al. 2015; Stewart et al. 2016a; Moazzam 2018; Beale et al. 2019) and may have distinct home ranges (Stewart et al. 2016a).

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 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); with many of these studies having been conducted for the last 10–20 years (Table 6). A study of Japan-wide photographic records confirmed that the known main aggregation in Ogasawara Islands (42 known individuals during 1995–1998 study) represents a part of a fairly isolated population (Kashiwagi et al. 2010). A mark-recapture population study in southern Mozambique over five years from 2003 to 2008 estimated the local population during that time to be 600 individuals (Marshall 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 6 may not be indicative of population size.

The number of individually identified giant manta ray for each studied aggregation ranges from less than 50 in regions with low survey effort or infrequent sightings to more than 1,000 in some regions with targeted, long-term studies. However, ongoing research including mark-recapture analyses suggests that typical subpopulation abundances are more likely in the low thousands (e.g. Beale et al. 2019) and in rare cases may exceed 10,000 in areas with extremely high productivity (pers. comm. Joshua Stewart, Manta Trust to A. Garrett 2021). Of the 12 studied subpopulations identified in Table 6, statistical analyses of sightings/photo-identification data to estimate total population size has only been conducted for three of them. For Raja Ampat, CITES (2013) indicated that there were 72 identified individuals. After additional research and an analysis of resightings data, Beale et al. (2019) estimated the total population size to be approximately 1,875 individuals. Isla de la Plata, Ecuador had approximately 650 identified individuals reported in CITES (2013), in this case, Burgess (2017) conducted further analyses and estimates the total population size to be 2,464 individuals. Similarly, for the Republic of Maldives, as of 2013, 63 individuals had been identified (CITES 2013), Nicholson-Jack (2020) reported 378, and further study indicates a more than 10-fold increase over the initial number of identified individuals (n = 716; J. Stewart pers. comm. to A. Garrett citing S. Hilbourne pers. comm. 2021; Table 6). Thus, while some subpopulations may have been reduced to very small population sizes due to fisheries (direct harvest or bycatch), in general, viable 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 has 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 2019c). 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.

Table 6. Numbers of recorded individuals and subpopulation estimates of giant manta ray at identified locations originally adapted from CITES (2013a) and updated with supplementary references as specified.

Location	**Recorded Individuals**	**Subpopulation Estimate**	**Reference**
Mozambique	180 - 254	600	Marshall et al. (2009) and pers. comm. cited in CITES (2013); MantaMatcher (2016)
Egypt	60	-	Marine Megafauna (2011) as cited in CITES (2013)
Republic of Maldives	716	-	J. Stewart pers. comm. to A. 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 2011as 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)

3.3.11.3 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 Phillipine 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. 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, so increased mortality of those life stages are also important.

3.3.11.4 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).

3.3.11.5 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). 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 (Marshall and Conradie 2014). Utami and Marlow (2018) report a 97% decrease in landings after the ban was implemented. However, as the researchers do not provide abundance estimates, we cannot ascertain the condition of the giant manta ray population in this area to determine the effectiveness of the measure other than the noted decrease in landings in Lamakera. Conversely, if the population has been overexploited, there may not be mantas to land.

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, and the Pacific and Indian Oceans.

Giant manta rays have been identified in United States bycatch data for the Hawaii DSLL and SSLL fisheries. Observer data (20% coverage) for the Hawaii DSLL fishery recorded 56 interactions with giant manta rays from 2004-2018 with an estimated 218 interactions (NMFS unpublished data). Observer data (100% coverage) for the Hawaii SSLL fishery recorded 21 interactions with giant manta rays from 2004-2018 (NMFS 2019b).

3.3.11.6 Conservation

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

Internationally, the giant manta ray is protected in the Maldives, Philippines, Mexico, Brazil, Ecuador, Yap, Indonesia, Western Australia, and New Zealand (Miller and Klimovich 2017). These protections range from restrictions on knowingly capturing or killing rays, to bans on exportation of ray species and their body parts to established Marine Protection Areas of known giant manta ray aggregations. However, many of these restrictions are difficult and rarely enforced, specifically in Indonesia where the restriction has notably increased the price of manta ray products, significantly increasing the current income of current resident fishermen (Marshall and Conradie 2014).

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 Megafuna Foundation, the Manta Pacific Research Foundation and MantaWatch. Others are driven by the countries whose economies largely depend on manta ray tourism (Erdmann 2014). 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 WCPFCIA in January 2021. CMM 2019-05 prohibits vessels from targeted fishing or intentional setting on mobulid rays; from retaining on board, transshipping, or landing any part or whole carcass of mobulid rays; fishing vessels must promptly release animals alive and unharmed that will result in the least possible harm to the individuals captured.

3.3.11.7 Summary of the Status

In this section of this biological opinion, we explained that although the giant manta ray occurs circumglobally the species is typically documented in small regional subpopulations. While some subpopulations may have been reduced to very small population sizes due to fisheries (direct harvest or bycatch), in general, viable 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 information is needed to understand the population structure of this species, and the role that small aggregations play in the species persistence. This species is one of the least understood of the marine mega vertebrates. Many of the studied giant manta ray subpopulations have declined significantly in areas subject to fishing (Marshall et al. 2018). Fisheries catch and bycatch has 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).

Our diagram of the species' population dynamics includes natural and anthropogenic threats that affect each life cycle stage. As you read the causal loop diagram in Figure 30, recall that the arrow represents the path between two variables and the sign associated with the arrow represents the relationship between two connected variables (whether an increase in one variable results in an increase in the other or whether an increase in one variable results in a decrease in the other). A variety of protections and conservation efforts have, and continue to occur, and these are represented with another arrow to the threats and demonstrate a decrease in the threats as evidenced by the positive sign at the terminus of the arrow. We patterned this diagram based on the apparent dynamics of the global giant manta ray.

As mentioned above, in the early stages of development as an embryo, the giant manta ray is susceptible to toxins that may be passively transferred from its mother through milk production (Lyons et al. 2013). Species like the giant manta ray with delayed sexual maturity increase their potential to accumulate toxins and therefore, are expected to offload higher levels of contaminants to their offspring. Once the giant manta ray grows beyond a neonate, it is vulnerable to the same threats throughout its juvenile and adult life stages. 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, have little potential to withstand high and sustained levels of fishing exploitation (Hoenig and Gruber 1990; Stevens et al. 2000; Couturier et al. 2012; Dulvy et al. 2014). Furthermore, subpopulations may or may not have observed skewed sex ratios. Meaning threats to each subpopulation may be unique. Despite the best efforts of protections and conservation measures, the overall trend of the giant manta ray continues to decline.

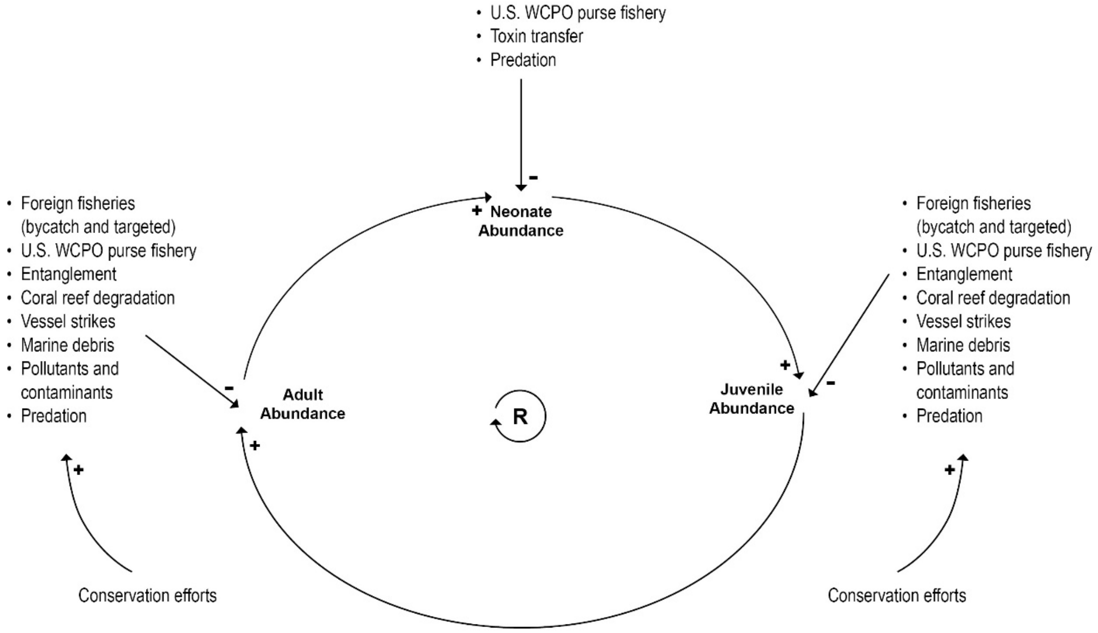


Figure 30. Giant manta ray casual loop diagram depicting various stressor-response relationships for each life cycle stage.

4 ENVIRONMENTAL BASELINE

By regulation, the *Environmental Baseline* refers to the condition of the listed species or its designated critical habitat in the *Action Area*, without the consequences to the listed species or designated critical habitat caused by the proposed action. The environmental baseline includes the past and present impacts of all Federal, State, or private actions and other human activities in the *Action Area*, the 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. 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 Consultation Handbook further clarifies that the environmental baseline is “an analysis of the effects of past and ongoing human and natural factors leading to the current status of the species, its habitat (including designated critical habitat), and ecosystem, within the *Action Area*” (FWS and NMFS 1998). 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 16 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 ASLL fisheries and other fisheries that occur within the same *Action Area* as the United States WCPO purse seine fishery (NMFS 2004, 2006, 2010a; McCracken 2019). 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 white tip sharks (Young et al. 2017), the status review report for leatherback sea turtles (NMFS and FWS 2020), and the other sources as cited in subsequent subsections.

4.1 Global Climate Change

Global annually averaged surface air temperature has increased by about 1.8 ºF (1.0 ºC) over the last 115 years (1901 to 2016) (USGCRP 2017). This period is now the warmest in the history of modern civilization. It is extremely likely that human activities, especially emissions of greenhouse gases, are the dominant cause of the observed warming since the mid-20th century. For the warming over the last century, there is no convincing alternative explanation supported by the extent of the observational evidence (USGCRP 2017). These global trends are expected to continue over climate timescales. The magnitude of climate change beyond the next few decades will depend primarily on the amount of greenhouse gases (especially carbon dioxide) emitted globally. Without major reductions in emissions, the increase in annual average global temperature relative to preindustrial times could reach 9 ºF (5 ºC) or more by the end of this century (USGCRP 2017). With significant reductions in emissions, the increase in annual average global temperature could be limited to 3.6 ºF (2 ºC) or less (USGCRP 2017). The global atmospheric carbon dioxide concentration has now passed 400 parts per million, a level that last occurred about three million years ago, when both global average temperature and sea level were significantly higher than today. There is broad consensus that the further and the faster the earth warms, the greater the risk of potentially large and irreversible negative impacts (USGCRP 2017).

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

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

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

Marine species ranges are expected to shift as they align their distributions to match their physiological tolerances under changing environmental conditions (Doney et al. 2012). Climate-related shifts in range and distribution have already been observed in some marine mammal populations (Silber et al. 2017). Marine mammal species often exhibit strong dependence on or fidelity to particular habitat types, oceanographic features, and migration routes (Sequeira et al. 2018). Specialized diets, restricted ranges, or reliance on specific substrates or sites makes many marine mammal populations particularly vulnerable to climate change (Silber et al. 2017). Marine mammals with restricted distributions linked to water temperature may be exposed to range restriction (Learmonth et al. 2006; Isaac 2009). MacLeod (2009) estimated that, based upon expected shifts in water temperature, 88% of cetaceans would be affected by climate change, 47% would be negatively affected, and 21% would be put at risk of extinction. Hazen et al. (2012) examined top predator distribution and diversity in the Pacific Ocean in light of rising sea surface temperatures using a database of electronic tags and output from a global climate model. Hazen et al. (2012) predicted up to a 35% change in core habitat area for some key marine predators in the Pacific Ocean, with some species predicted to experience gains in available core habitat and some predicted to experience losses. 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 marine mammal and sea turtle foraging success as well as sea turtle reproductive periodicity (Kashner et al. 2011; Pike 2014).

Significant impacts to marine mammals and sea turtles from ocean acidification may be indirectly tied to foraging opportunities resulting from ecosystem changes (Busch et al. 2013; Haigh et al. 2015; Chan et al. 2017). Nearshore waters off California have already shown a persistent drop in pH from the global ocean mean pH of 8.1 to as low as 7.43 (Chan et al. 2017). The distribution, abundance and migration of baleen whales reflects the distribution, abundance and movements of dense prey patches (e.g., copepods, euphausiids or krill, amphipods, and shrimp), which have in turn been linked to oceanographic features affected by climate change (Learmonth et al. 2006). Ocean acidification may cause a shift in phytoplankton community composition and biochemical composition that can impact the transfer of essential nutrients to predators that eat plankton (Bermudez et al. 2016). Increased ocean acidification may also have serious impacts on fish development and behavior (Raven et al. 2005), including sensory functions (Bignami et al. 2013) and fish larvae behavior that could impact fish populations (Munday et al. 2009) and piscivorous ESA-listed species that rely on those populations for food.

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 spatiotemporal 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).

Some species, like sharks, that primarily occupy pelagic waters would likely be less affected by climate change than other listed species like turtles that have a pelagic, nearshore and terrestrial life history stages. Even so, not all elasmobranchs have similar habitat requirements. Chin et al. (2010) conducted a climate vulnerability ranking of the relative risk that climate change poses to sharks and rays on Austria's Great Barrier Reef. The risk ranking suggests that species like the giant manta ray, oceanic whitetip and scalloped hammerhead sharks (bathyal and pelagic species) are generally less vulnerable to changes in habitat attributes that are often associated with climate change like sea-level rise, severe weather, and freshwater input. The exception in this category of pelagic and bathyal species are the plankton feeders like giant manta ray, which were ranked as having a moderate vulnerability to changes in temperature and ocean circulation (Chin et al. 2010). Since giant manta rays frequently rely on coral reef habitat for important life history functions (e.g., feeding, cleaning) and are planktivores, we would expect that they would be sensitive to environmental changes that affect primary and secondary productivity (Brainard et al. 2011; Guinder and Molinero 2013). A recent study on giant manta rays by Beale et al. (2019) examined the distribution of nearly 600 giant manta rays relative to a few environmental variables over a six-year period, and concluded that the habitat attributes of temperature, ocean circulation, and water column stratification (also attributes of climate variability) can have a profound effect on zooplankton distribution and consequently, giant manta ray distribution. In the event, attributes of climate variability, like water temperatures cause a shift in zooplankton, giant manta ray may need to expand their foraging grounds, which would increase their energetic demands and may in turn reduce their access to other important habitats like breeding grounds and cleaning stations. Consequently, we would expect large scale changes in climate may negatively impact the fitness of the giant mantas by altering access and distribution of their food, and could hinder their ability to reduce parasitic loads and dead tissue at cleaning stations, 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.

4.2 Commercial Whaling/Subsistence Hunting/Human Consumption

4.2.1 Marine Mammals

As described in the *Status of Listed Resources* sections for fin, sei and sperm whales, the status and current abundances of large whales were driven by whaling that resulted in substantial population declines. Whaling occurred mainly from 1904 to 1975 for fin whales (IWC 1990), 1910 to 1975 (Horwood 1987) for sei whales, and 1800 and 1987 for sperm whales (Best 1976 as cited in Carretta et al. 2018a; Ohsumi 1980; Clapham et al. 1997; Brownell et al. 1998 as cited in Carretta et al. 2018a; Clapham and Ivashchenko 2016). There is no active whaling or subsistence use of whales considered in this biological opinion in the *Action Area*.

Between 1910 and 1975, harvest of 152,233 sei whales occurred in commercial whaling activities in the Southern Hemisphere (Horwood 1987). About 63,500 sei whales were killed during commercial whaling in the North Pacific between 1910 and 1987 (Committee for Whaling Statistics 1942; Scheffer and Slipp 1948; Rice 1977; Barlow et al. 1997). Japan has continued whaling for sei whales under the auspices of scientific whaling and captured 592 from 1988-2009 in the northwestern Pacific (IWC 2010), and 1,698 sei whales under their special permit in the North Pacific between 2001-2018 (IWC 2021).

Between 1800 and 1909, about 60,842 sperm whales were estimated harvested in the North Pacific (Best 1976 as cited in Carretta et al. 2018a). Thirteen sperm whales were harvested by shore-based whaling stations in California between 1919 and 1926 (Clapham et al. 1997). The reported harvest of North Pacific sperm whales by commercial whalers between 1947 and 1987 totaled 258,000 (C. Allison pers. comm. as cited in Carretta et al. 2017b). Ohsumi (1980) lists an additional 28,198 sperm whales harvested mainly in coastal whaling operations from 1910-1946. Based on the massive under-reporting of Soviet catches, Brownell et al. (1998 as cited in Carretta et al. 2018a) estimated that the Soviet pelagic whaling fleet harvested about 89,000 additional whales between 1949 and 1979. The Japanese coastal operations also underreported catches by an unknown amount (Clapham and Ivashchenko 2016). Thus, a total of at least

436,000 sperm whales were harvested between 1800 and the end of commercial whaling for this species in 1987. Between 2000 and 2016, 56 sperm whales were caught by Japan under the auspices of scientific research permits (IWC 2010).

From 1904 to 1975, there were 703,693 fin whales killed in Antarctic whaling operations (IWC 1990).

Historic whaling plays an integral role in the current status of these species within the Pacific. The most severe consequence being that population size and structure were affected due to overharvesting. Considering whales have low reproductive rates and produce single calves over a long time frames, are long lived, and travel across large expanses traversing multiple jurisdictions, these impacts will have lasting effects for the species. These populations may not repopulate historic geographic ranges where extirpated. Additionally, recovery is further depressed by incidental anthropogenic mortality (i.e. ship strikes, fishery interactions, etc.), indirect effects (i.e. competition for food resources, habitat degradation, etc.) and illegal direct harvests (Balance 2012).

4.2.2 Sea Turtles

Direct harvest of the sea turtles considered in this biological opinion still occurs in some nations within the *Action Area*.

4.2.2.1 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 Commonwealth of the Northern Mariana Islands (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 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.

4.2.2.2 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 largest market globally. 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).

4.2.2.3 Leatherback Sea Turtles

Leatherback sea turtle egg collection occurs in many countries around the world (NMFS and FWS 2020). 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; NMFS and FWS 2020). Historically both adult females and eggs were harvested at the Bird’s Head Peninsula in Indonesia, this harvest has been minimized at the index beaches of Jamursba Medi and Wermon but continues at other nesting beaches on the peninsula (Hitipeuw et al. 2007; Tapilatu et al. 2013; NMFS and FWS 2020). At Buru Island, Indonesia, it is likely that three to five nesting females have been harvested each year for more than 20 years (J. Wang, NMFS, pers. comm., 2018 as cited in NMFS and FWS 2020).

4.2.2.4 South Pacific Loggerhead Sea Turtles

Directed harvest for loggerheads still occurs in many and is a serious and continuing threat to loggerhead recovery (NMFS and FWS 1998c). Legislation in Australia outlaws the harvesting of loggerheads by indigenous peoples (Limpus et al. 2006). Despite national laws, in many areas the poaching of eggs and hunting of adult and juvenile turtles is still a problem, and Limpus and Fien (2009) suggested that the harvest rate of loggerheads by indigenous hunters, both within Australia and in neighboring countries, is approximately 40 turtles per year. Preliminary studies suggest that local harvesting in New Caledonia constitutes about 5% of the nesting population (Limpus et al. 2006). Loggerheads also are consumed after being captured incidentally in high-seas fisheries of the Southeastern Pacific (Alfaro-Shigueto et al. 2006), and occasionally may be the product of illegal trade throughout the region.

4.2.2.5 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).

Summary

The directed harvest of sea turtles and eggs continues to be a threat to the persistence of sea turtles species considered in this biological opinion and may have impacts on population growth rates that are similar to or, for some populations, potentially greater than impacts from fishery bycatch. For example, in CNMI, 32% of nesters have been poached over the last decade, from a total nesting population of 6-18 females, and that the illegal and continued harvest of sea turtles on Saipan has reduced the annual population growth rate from 11.4 to 7.4% (Summers et al. 2018a). The primary threat to turtles in the CNMI is human induced, with 79% of strandings attributed to poaching (Summers et al. 2018b).

4.2.3 Elasmobranchs

As noted above there are not many directed fisheries for sharks, however, the demand for shark products, including liver oil, hides, meat, teeth, jaws, and especially fins, has resulted in opportunistic retention of sharks caught as bycatch in other non-domestic fisheries as opposed to releasing them. 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 were further enhanced in response to a stock assessment report indicating that oceanic whitetip sharks are overfished (Rice and Harley 2012). The WCPFC adopted CMM 2011-04 (most recently revised in CMM 2019-04), prohibiting the retention of oceanic whitetip sharks, in whole or in part, for vessels flying the flags of and vessels under charter arrangements to participating commission members, non-members, and territories in the fisheries covered by WCPFC. The measure further requires the release of any oceanic whitetip 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.

Similar to sharks, giant manta rays are valued their meat, but in addition for their 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 rakers, 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 Lamakera, 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 Lamakera in large numbers in the 1990s (Barnes 2005).

Although fishing for manta rays was banned within the Indonesian EEZ in February 2014, manta rays were still being caught and processed at Lamakera as of May 2014, 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 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 four 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.

Human consumption has played a pivotal role in current global shark and ray populations not only in the Pacific but worldwide. Recently, the USFWS seized a historic 1,400 lbs of shark fins from South America destined for Asia in Miami in January of 2020 (Guzman and Ries 2020). But was upstaged in May 2020 by the seizure of 26 mt of shark fins in Hong Kong (Lo 2020). While the news reports most were from thresher and silky sharks, these events highlight the pressure illegal harvests place on elasmobranchs. Illegal overharvesting pushes these species closer to extinction.

Similar conservation measures were established by the WCPFPC for giant manta rays. CMM 2019-05 prohibits vessels from targeted fishing or intentional setting on mobulid rays; from retaining on board, transshipping, or landing any part or whole carcass of mobulid rays; fishing vessels must promptly release animals alive and unharmed that will result in the least possible harm to the individuals captured. Furthermore, purse seine vessel's must, at the point of landing or transshipment, surrender whole mobulid rays to the responsible governmental authorities, or other competent authority, or discard them where possible if found incidentally caught and landed. CMM 2019-05 further states that mobulid rays that are surrendered in this manner may not be sold or bartered but may be donated for purposes of domestic human consumption. Proper mitigation, identification, handling and releasing techniques are noted and vessels are encouraged keep all necessary equipment onboard for the safe release of mobulid rays. Lastly, the CMM encourages at-vessel and post-release mortality metrics be researched while also allowing for the collection of biological samples by observers within their Convention Area.

4.3 Fisheries

A key impact to all species considered in this biological opinion within the *Action Area* are from United States and international fishing fleets other than the United States WCPO purse seine fishery. The primary fisheries that occur in the *Action Area* and that have measurable effect on the endangered and threatened species considered in this biological opinion are fisheries for highly migratory species. These primarily involve fisheries for tuna (albacore, bigeye, skipjack and yellowfin), swordfish, and marlin and primarily involve longline and purse seine gear. Fisheries that use both types of gear have occurred in the western and central Pacific Ocean since the 1950s (for longline fisheries; 1980s for purse seine fisheries), but fisheries managers have only recently collected systematic data that can be used to assess the impacts of these fisheries on endangered and threatened species in the WCPO generally and the *Action Area*. Nevertheless, the early stages of these fisheries probably had measurable effect on the abundance and dynamics of populations of imperiled species in the *Action Area*. Given the longevity and long generation times of the endangered and threatened species that occur in the *Action Area*, those impacts are likely to still influence the dynamics of populations of these species. In this section we review the available information for the level of impact of these fisheries on the ESA-listed species considered in this biological opinion.

4.3.1 United States WCPO Purse Seine Fishery Historical Effort

Since 2006, the size of the United States WCPO purse seine fleet increased markedly mainly based on newly constructed vessels entering the fishery (Hamilton et al. 2011). This upturn in fleet size is reflected in the sharp increase in the United States purse seine catch in the WCPO. From 2007 to 2009, the tuna catch more than tripled (Hamilton et al. 2011) do. While the increase in the size of the United States WCPO purse seine fleet was mirrored by an increase in fishing effort as described in the BE during this period (NMFS 2017a), recently, there has been a downward trend in effort is since 2015 (NMFS IFD suppl. memo dated August 19, 2019). In 2015, 38 vessels were active in the fishery and 33 active in 2018. This fishery is limited to a maximum of 40 vessels by the South Pacific Tuna Treaty (SPTT). As noted in the *Consultation History*, on April 2, 2021, NMFS IFD presented NMFS PRD with a memorandum documenting its changes to the proposed action, including a reduction in expected level of fishing effort to no more than 3,100 sets per year with an expectation that 1,581 of those being FAD sets.

In recent years (from 2009 to the present), regulations to implement decisions of the Commission limit the total number of fishing days that United States purse seine vessels may collectively spend each year in the Effort Limit Area for Purse Seine (ELAPS), the area between the latitudes of 20° N. and 20° S. on the high seas and in the United States EEZ in the WCPFC Convention Area. The 2015 limit in the ELAPS of 1,828 fishing days was reached in June 2015, and the fishery was closed in the ELAPS from June 15, 2015 through December 31, 2015. The 2016 limit in the ELAPS of 1,828 fishing days was expected to be reached by September 2, 2016, so the fishing was closed in the ELAPS from September 2, 2016 through December 31, 2016 (81 FR 58410). Additionally, the fishery was closed from September 18, 2018 to December 31, 2018 as a result of reaching the 2018 limit (83 FR 45849) and from October 9, 2019-November 29, 2019 and from December 9-31, 2019, due to the limit in 2019 (84 FR 65690).

Historically, past impacts by the United States WCPO purse seine fishery to the species considered in this biological opinion include the following. The fishery has captured 86 ESA-listed whale species (observed: 45 sei, 5 sperm, and 4 fin whales) with an additional 32 unidentified marine mammals from 2008 to 2018. It's estimated that a total of 116 (95% CI: [100, 172]) marine mammals have been caught by the fishery during this time.

In those same years, 297 records of sea turtles were observed captured by the fishery including 97 green, 8 leatherback, 68 loggerhead, 57 olive ridley, 59 hawksbill, and 8 unidentified sea turtles. It is estimated a total of 508 (95% CI: [444, 606]) turtles were captured.

Furthermore, 680 sets have interacted with oceanic whitetip sharks resulting in the capture of 1,330 observed individuals between those same years (2008 and 2018). It's estimated the fishery caught a total of 2,284 (95% CI: [1,983, 2,569]) oceanic whitetip sharks during that time.

A total of 14 Indo-West Pacific scalloped hammerhead sharks were caught and positively identified in the United States WCPO purse seine fishery. However, NMFS IFD estimates a total of 41 (95% CI: [31, 51]) Indo-West Pacific scalloped hammerhead sharks were captured between 2008 and 2018 using the Bayesian model approach (NMFS 2019c). Lastly, NMFS estimates that 5,330 (95% CI: [4,254, 6,258]) giant manta rays were captured in the United States WCPO purse seine fishery between 2008 and 2018. However, see Section 5.2.1.3 for a discussion on the likelihood of misidentification for this species, in that most of the captured individuals identified as giant manta ray were likely a different species; hence the estimate of total captures calculated from the observer data is considered to be an overestimate of actual captures.

The effect of this continuing action on ESA-listed species is addressed in the *Effects of the Action* section of this biological opinion in greater detail. In this section, we present an overview of the types and effort level for these fleets that overlap in geographical space with the fishery. We address the specific impacts of these fisheries to each species group below.

4.3.2 United States Fisheries Managed under the Pelagics Fisheries Ecosystem Plan

Fisheries managed under the Pelagics FEP that occur in the *Action Area* include the American Samoa-based pelagic longline and troll fisheries (fisheries managed under the Pelagics FEP that occur outside the *Action Area* include the shallow-set and deep-set Hawaii longline fisheries; and the troll, handline, and pole-and-line fisheries in Hawaii, Guam, and CNMI). Previously, the 2006 biological opinion considered the DSLL fishery in the *Environmental Baseline* (NMFS 2006). However, the two fisheries no longer overlap in geographical space. The last fishing set conducted by the DSLL fishery in waters north of Palmyra was in 2015 (NMFS unpublished observer data 2020). Furthermore, NMFS Sustainable Fisheries Division (SFD) confirmed this information and did not consider the fisheries to overlap in their recent biological evaluation of the DSLL fishery (NMFS 2018b). Additionally, the PRIA pelagic troll fisheries do not have an ITS for any species considered in this opinion.

There is also an American Samoa bottomfish fishery that occurs in the *Action Area* (McCracken 2019). These fisheries are described and analyzed in detail in the 2008 biological opinion on the Bottomfish fisheries of the MHI, Guam, American Samoa, and the CNMI (NMFS 2008a), an informal consultation dated April 9, 2015 in which NMFS determined the bottomfish fisheries are not likely to adversely affect the Indo-West Pacific scalloped hammerhead shark and ESA-listed reef-building corals, including *Acropora globiceps*, *A. jacquelineae*, *A. retusa*, *A. speciosa*, *Euphyllia paradivisa*, and *Isopora crateriformis* (NMFS 2015b), a 2015 biological opinion on the ASLL fishery (NMFS 2015a), a 2009 biological opinion on the pelagic troll and handline fisheries, as managed under the FEP (NMFS 2009), and a 2004 biological opinion on all the fisheries managed under the Pelagics FEP (NMFS 2004). This section attempts to consolidate the information from these sources for each species group.

4.3.2.1 Marine Mammals

ITS' have not been provided to any marine mammal species in United States fisheries operating within the *Action Area* based on our searches conducted during the course of this consultation (e.g. ECO searches, NMFS records maintained by our office or the NOAA Institutional Repository, etc.). The number of interactions and corresponding mortality attributed to marine mammals in United States fisheries operating in the Pacific but outside of the *Action Area* are detailed in the relevant *Status of Listed Resources* sections.

4.3.2.2 Sea Turtles

ITS' have been provided for all sea turtle species considered in this biological opinion for United States fisheries managed under the Pelagic FEP (although not all populations in all fisheries).

The best available information comes from the observer program for the ASLL fishery and we summarize total captures and mortalities for each species from 2012 to the second quarter of 2019 for each species below.

Green sea turtles

In total, 18 turtles have been observed with 108 estimated interactions in the ASLL fishery from 2012 to the second quarter of 2019 (McCracken 2019). This includes all green sea turtle species, which may occur in the *Action Area*. Genetic data collected from green sea turtles captured by the ASLL provide information on the proportions of individual green sea turtle populations represented in the bycatch. Using this information, McCracken (2019) estimates that 6 Central South Pacific, 12 Southwest Pacific, 22 East Pacific, 35 Central West Pacific and 11 East Indian-West Pacific green sea turtles will be caught annually. However, a larger period was considered (from 2006 to 2018) in the genetic analysis of green sea turtle samples for the ASLL fishery. Therefore, the data may present higher interaction numbers than those presented in McCracken (2019).

Hawksbill sea turtle

Observers begun monitoring this fishery in 2006, and there had been no recorded interactions with hawksbill sea turtles in this fishery until 2016. In total, from 2016 to the second quarter of 2019, three hawksbill sea turtles have been observed captured in the ASLL fishery and all three were dead. Each turtle was externally hooked; two in the side of the mouth and one in the front flipper, and all three were juveniles. The estimated number of interactions and estimated future interactions were provided by McCracken (2019) and consider data from 2016 to 2018. Using McCracken's (2019) estimates, we anticipate that up to 40 hawksbill sea turtles may be captured by the ASLL every three years, and all captures would likely result in mortalities.

Leatherback sea turtle

Between 2006 and the second quarter of 2019, the ASLL is estimated to have had 68 interactions (39 mortalities) with leatherback sea turtles (McCracken 2019). Six genetic samples have been analyzed from this fishery and all were from the West Pacific Ocean population (P. Dutton pers. comm. July 5, 2018). Of the 12 observed leatherback interactions in the fishery, eight were between 50 SCL and 100 SCL, which are juveniles, and the four larger are in the subadult to adult life stages and ranged from 122 cm to 152 cm SCL. Gender was determined for two of the juveniles in which one was a female and the other a male. Using this information, McCracken (2019) estimates 69 leatherback sea turtles will be captured in three years, and of those, 45 will die.

Loggerhead sea turtle

No loggerhead sea turtle interactions have been documented in United States fisheries that overlap with the *Action Area* between 2006 and the second quarter of 2019 (McCracken 2019; NMFS unpublished data).

Olive ridley sea turtle

In total, 13 olive ridley turtles have been observed captured in the ASLL fishery from 2012 to the second quarter of 2019. During this period, eight died from their interactions (NMFS unpublished data). It is estimated that 70 interactions have occurred in total with this species (McCracken 2019). One turtle was released injured without obtaining any measurements. Six of the olive ridley sea turtles were juveniles based on carapace length while the remaining seven were classified as sub-adults or adults. McCracken (2019) estimated that up to 61 olive ridley sea turtles will be captured by the ASLL every three years, with 40 anticipated mortalities.

4.3.2.3 Elasmobranchs

All three species/populations of sharks and rays considered in this biological opinion are also captured in the ASLL fishery which is managed under the Pelagics FEP. The observed, estimated, and anticipated annual captures of oceanic whitetip sharks, Indo-West Pacific scalloped hammerhead sharks, and giant manta rays that occurred from 2010 - 2017 in the ASLL fishery are shown in Table 7. The consultation for the ASLL fishery is underway and the anticipated number of captures from the BE (NMFS 2019a) is given in Table 8.

Table 7. Summary of the mean (95% confidence interval) of the annual number observed captures of oceanic whitetip sharks, scalloped hammerhead sharks, and giant manta rays and the estimated annual number of interactions based on observer coverage in the ASLL fishery from 2010-2017 (McCracken 2019).

Species	Observed Interactions	Estimated Interactions
Oceanic Whitetip Shark	115 (82 – 147)	628 (418 – 837)
Scalloped Hammerhead Shark	1 (0 – 2)	5 (2 – 9)
Giant Manta Ray	2 (1 – 2)	8 (1 – 15)

Table 8. Anticipated annual interactions of elasmobranchs at the mean and 95th percentile for the ASLL (McCracken 2019)*.

Species	Mean	95th percentile
Giant Manta Ray	9	28
Manta/Mobula ray	5	16
Unidentified ray	8	24
Oceanic whitetip shark	620	1,110
Indo-West Pacific scalloped hammerhead shark	5	15
IWP unidentified hammerhead shark	18	41

*Values reported by McCracken (2019) may be slightly different as they have been rounded for the purposes of this table in order to properly account for the individual animal.

4.3.3 Non-United States WCPO Longline Fisheries

There were roughly 5.5 to 7.5 million hooks deployed annually by the international longline fleet operating in the WCPO between 2008 and 2015 (WCPFC 2019b). 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. 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 tuna, bigeye tuna, albacore, and swordfish. The non-United States WCPO longline fishery as a whole was observed at rates between 1.8-3.0% from 2013–2017. In contrast, the United States fleet has, maintained coverage at approximately 20% for the ASLL annually since 2010.

Peatman et al. (2018b), summarize observer data from 2003 to 2017 to estimate the catch and catch composition of the longline fisheries of the WCPO. 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 WCPFC Convention Area (Figure 31) 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). The data from the Hawaii deep and shallow set longline fisheries is included in this summarized observer data report even though those fisheries do not overlap the *Action Area*. Therefore, the cells highlighted in Figure 31 near Hawaii should be ignored as they do not overlap the United States WCPO purse seine fishery and are discussed in the *Status of the Listed Resources* as appropriate.

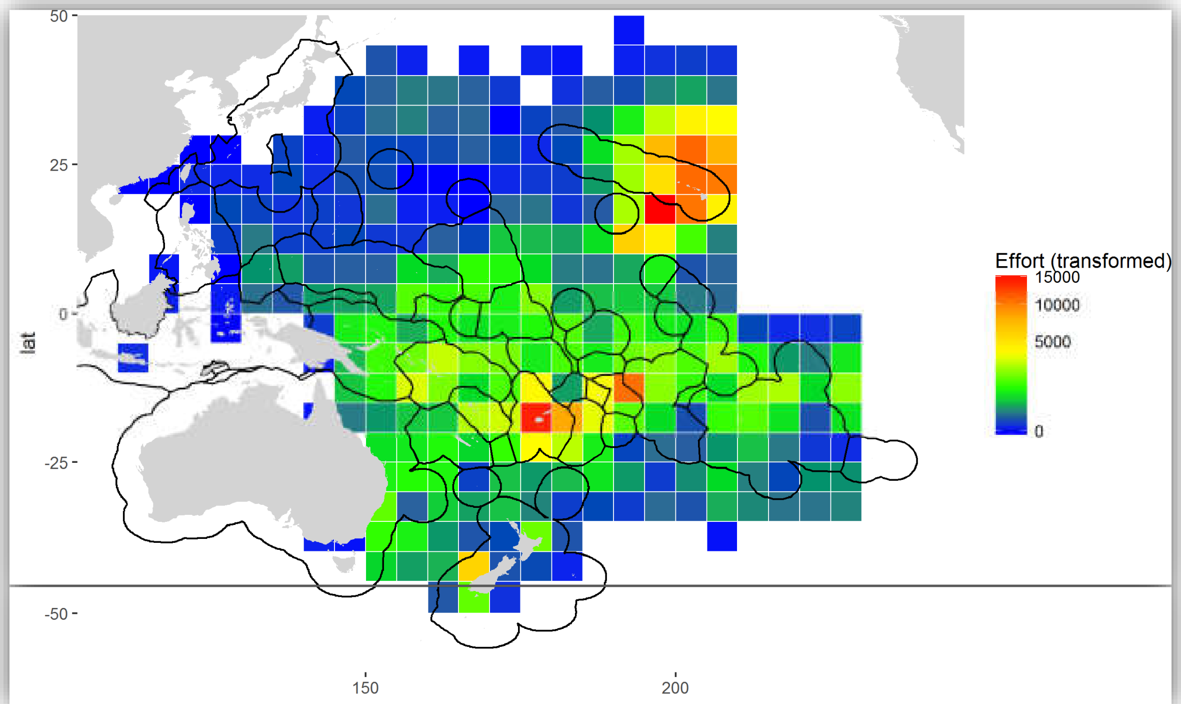


Figure 31. Observed effort in number of hooks (square root transformed) for longliners between 2003 and 2017 in the WCPFC convention area (Peatman 2018b).

4.3.3.1 Marine Mammals

Large whales are generally not impacted by longline fishing gear, with exception of bait removal and catch depredation by toothed whales. Sperm whales are known to depredate bait and catch from longline hooks. Based on a literature review, Hamer et al. (2012) found that false killer whales and pilot whales were primarily responsible for depredation at lower latitudes, while killer whales and sperm whales are the main species involved at higher latitudes. Depredation poses risks to cetaceans from ingesting the hook, which can become lodged in the mouth throat or stomach and lead to infection (Hamer et al. 2012). Some hooked animals may not be able to surface to breathe, although this is not likely a problem for large sperm whales. Sperm whales are in the *Action Area* and overlap with longline fisheries; hence, longline fisheries likely adversely affect the fitness of some sperm whales, but there are not sufficient data to describe the significance of this effect on the species chances of survival and recovery. Overall it is unlikely that ESA-listed marine mammal interactions with longline fishing gear are impacting their overall abundance or reproductive output.

4.3.3.2 Sea Turtles

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 9 (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 9. 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 10 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 10. 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 11. This information is not exclusive of United States fishery data.

Table 11. Average and (95% confidence interval) of the annual numbers of sea turtles reported captured/killed from 2013 to 2018 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**
Green Sea Turtle	33 (30 – 37)	16 (14 – 18)	1,373 (1,314 - 1,432)	668 (549 – 788)
Hawksbill Sea Turtle	15 (9 – 20)	6 (5 – 8)	581 (419 – 743)	251 (200 – 303)
Leatherback Sea Turtle	18 (13 – 22)	1 (0 – 2)	754 (502 – 1006)	62 (9 – 115)
Loggerhead Sea Turtle	53 (25 – 80)	10 (8 – 11)	2,193 (961 – 3,425)	392 (315 – 470)
Olive Ridley Sea Turtle	102 (45 – 159)	80 (27 – 132)	3,998 (1,958 – 6,038)	3,115 (1,234 – 4,996)
Percent Observer Coverage (2013-2018): 2.4% (2.1 – 2.8%)				

4.3.3.3 Elasmobranchs

In the WCPO, annual reports provided to the Commission from the member countries lack species-specific data for many of the elasmobranchs. Peatman et al. (2018b) modeled median shark and ray catch estimates for some species for multiple regions of the WCPFC using longline observer data. Tremblay-Boyer et al. (2019) present an updated model of bycatch and targeted catch of oceanic whitetip sharks in the WCPO longline fisheries from 1995 to 2016. Their estimates include the entire WCPFC convention area, which extends from 30°N to 30°S; the east to west boundaries are similar to the *Action Area*. Table 12 displays statistics relevant to the oceanic whitetip shark between 10°S and 10°N, and South of 10°S from 2003 to 2017 which includes a portion of the purse seine fisheries' *Action Area* and is summarized from Table 30 of Peatman et al. (2018b). All of 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; Tremblay-Boyer et al. 2019). As previously noted, observer coverage in United States fishing fleets in the Pacific is higher, ranging from 20 to 100%. 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 13).

Table 12. Mean and (95% confidence interval) of the annual median oceanic whitetip shark catch estimates between 10°S and 10°N, and South of 10°S in the WCPO longline fisheries from 2003 to 2017. Includes both United States and foreign data (Peatman et al. 2018b).

Oceanic Whitetip Shark (reported in '000 individuals)		
Year	**10°S to 10°N**	**South of 10°S**
2003 - 2017	43,873 (32,247 – 55,500)	16,349 (12,461 – 20,236)
Totals	658,100	245,230

Table 13. 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 2020).

Species	**Observed Interactions**	**Observed Mortalities**	**Estimated Interactions**	**Estimated Mortalities**
Oceanic Whitetip Shark	592 (538 – 647)	146 (120 – 171)	24,851 (20,877-28,824)	6,194 (4,653 – 7,735)
Scalloped Hammerhead Shark	14 (4 – 23)	5 (1 – 10)	568 (171 – 964)	230 (43 – 417)
Hammerhead Shark (spp. not identified)	7 (5 – 10)	1 (1 – 2)	307 (198 – 416)	56 (18 – 94)
Percent Observer Coverage (2013-2018): 2.4% (2.1 – 2.8%)				

Rays are currently not considered key bycatch species, and therefore have not been included in the Western Central Pacific Fisheries Commission bycatch summaries discussed above. However, their bycatch rates were summarized by Tremblay-Boyer and Brouwer (2016). 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 WCPO 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 32), suggesting a decline in abundance. From 2012 to 2015, 348 giant manta rays were observed captured by WCPO longline fisheries (Tremblay-Boyer and Brouwer 2016). The mean observer coverage for this fishery from 2014 to 2018 was 2.4% (Table 13) and a direct expansion of this observer rate suggests 348 observed giant manta ray captures would equal 14,500 estimated captures from 2012 to 2015, or 3,625 captures per year. The number of giant manta rays that would have died from their capture is unknown.

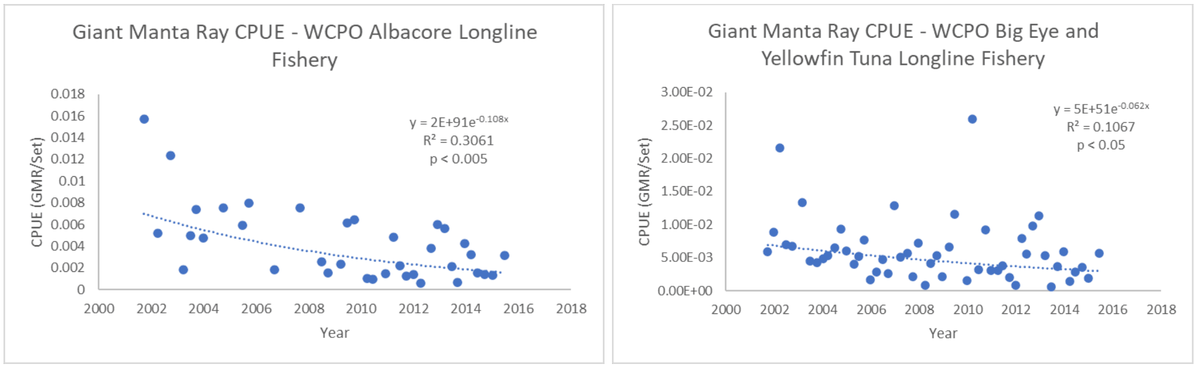


Figure 32. Data from Tremblay-Boyer and Brouwer (2016) for the WCPO longline fisheries. Dashed lines represent exponential curve fits describing the mean percent per year change in CPUE (value in the exponent of the fit equations). The regressions were significant for both of the fisheries at the p=0.05 level.

Peatman et al. (2018b) estimate the total number of manta and mobula ray interactions documented by observers from 2003-2017 to be 1,800. Of these, very few mobulids were retained, the majority captured were discarded. In Figure 33 below, of the mobulids discarded with known conditions (alive or dead), 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.

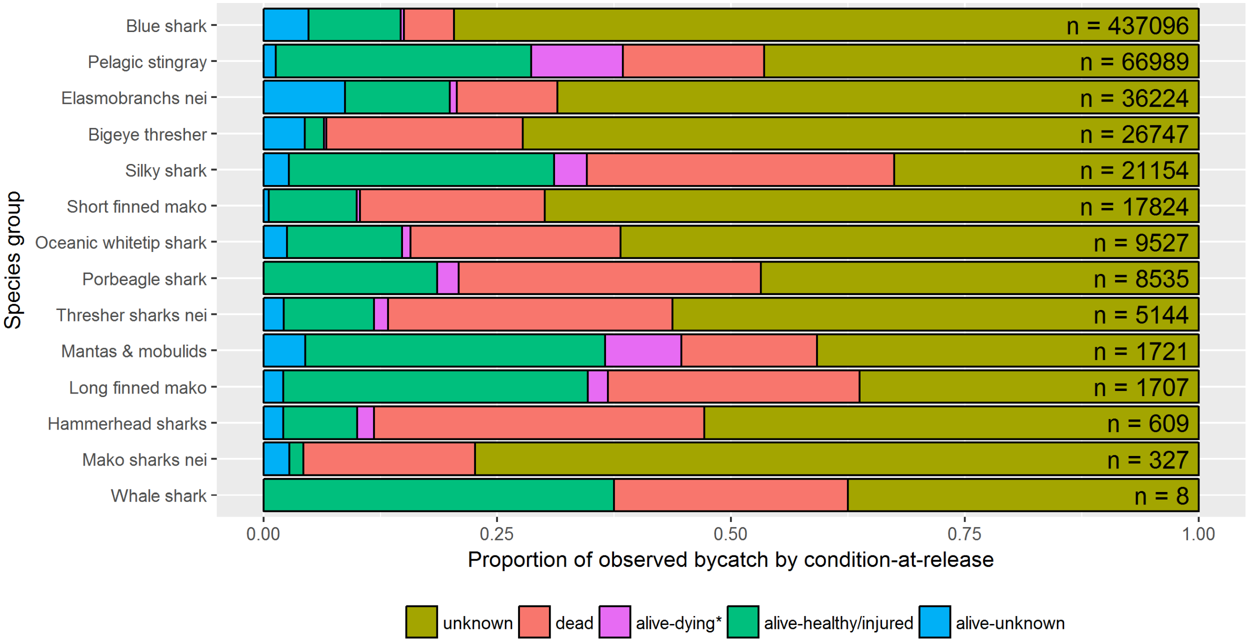


Figure 33. Recorded condition at release of observed sharks and rays catch by species/species group, as a proportion of total observed catch (number of fish) for the species/species group in the longline fisheries. The number of records is provided for each species/group. Note – alive-dying* are individuals that were alive but considered unlikely to survive (Peatman et al. 2018b).

4.3.4 Non-United States WCPO Purse Seine Fisheries

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 (WCPFC 2019b). 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, since 2010, received 100% coverage, although not all data are available (see discussion in the *Effects Analysis* section). Much of the best scientific and commercial data available in the literature includes United States data as they are regionally scaled analyses.

4.3.4.1 Marine Mammals

Large whales interact with purse seine fisheries; however, interactions are not always identified to species. Until 2012, it was a common practice for purse seiners to set on baleen whales, as schools of tuna are often associated with them (Secretariat of the Pacific Community Oceanic Fisheries Programme 2012). In 2012, the WCPFC adopted CMM 2011-2003, prohibiting setting on tuna schools associated with cetaceans in the high seas and in the EEZs of the WCPFC Convention Area (encompassing the EEZs within the *Action Area*); the measure went into effect in January 2013. Secretariat of the Pacific Community- Oceanic Fisheries Programme (2012) assessed logbook and observer data from 2007-2010 for WCPO purse seine fisheries for interactions with cetaceans. They found that any interaction with toothed whales was mainly incidental and not targeted; no sperm whale interactions were recorded in that timeframe. The interactions with baleen whales, however, did appear to be deliberate. In 2010, the rates were 0.2% for logbooks and 1.6% for observer data. We used the public domain bycatch data served by the WCPFC to estimate the numbers of annual interactions with ESA-listed whales from 2013 to 2018 (Table 14). These values are inclusive of the United States WCPO purse seine fishery that is being analyzed in the opinion. For the large whales considered in this biological opinion, interactions with purse seine fishery gear generally results in the animal being released unharmed, though in some cases minor injuries are incurred from net abrasions. Overall, it is unlikely that ESA-listed marine mammal interactions with purse seine fishing gear are impacting their overall abundance or reproductive output.

Table 14. Estimated ESA-listed whale interactions with the WCPO purse seine fishery (inclusive of the United States WCPO purse seine fishery) from 2013 to 2018. Estimated interactions were calculated using the Expansion Factor ([100/Observer Coverage] x [Observed Interactions] (WCPFC 2020).

Species	**Average (95% confidence interval) Annual Observed Interactions**	**Average (95% confidence interval) Annual Estimated Interactions**
Fin Whales	3 (0 – 6)	1 (0 - 5)
Sei Whales	42 (33 – 52)	68 (53 - 84)
Sperm Whales	4 (2 – 6)	6 (3 - 9)
Percent Observer Coverage (2013-2018): 63% (56 – 71%)		

4.3.4.2 Sea Turtles

There are fewer analyses of sea turtle bycatch available for the 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
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 2013-2017; 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**
Green Sea Turtle	47 (27 – 66)	1 (0 – 1)	75 (49 – 100)	1 (0 – 2)
Hawksbill Sea Turtle	30 (13 – 46)	0 (0 – 1)	47 (23 – 70)	0 (0 – 1)
Leatherback Sea Turtle	5 (3 – 7)	0 (0 – 1)	8 (6 – 11)	0 (0 – 1)
Loggerhead Sea Turtle	33 (20 – 45)	0 (0 – 1)	52 (35 – 69)	1 (0 – 1)
Olive Ridley Sea Turtle	39 (30 – 48)	1 (0 – 3)	63 (52 – 74)	2 (0 – 4)
Percent Observer Coverage (2013-2018): 61.4% (53.6 – 69.1%)				

4.3.4.3 Elasmobranchs

For the oceanic whitetip shark, more recent data conflicts with the expectation of Hall and Roman (2017) and the ERA team. Approximately 1.6% of sharks and rays caught in the WCPO purse seine fisheries from 2010 to 2016 were oceanic whitetip sharks; since 2009 the bycatch levels have been relatively stable after a decline starting in 2003 (Figure 34; Peatman et al. 2017). From 2009 to 2016, the average median number of oceanic whitetip shark captures per year was 493 (range: 469 to 5,201 individuals; Peatman et al. 2017). As previously noted, this data reviews all fisheries in the WCPFC's convention area and does include United States data.

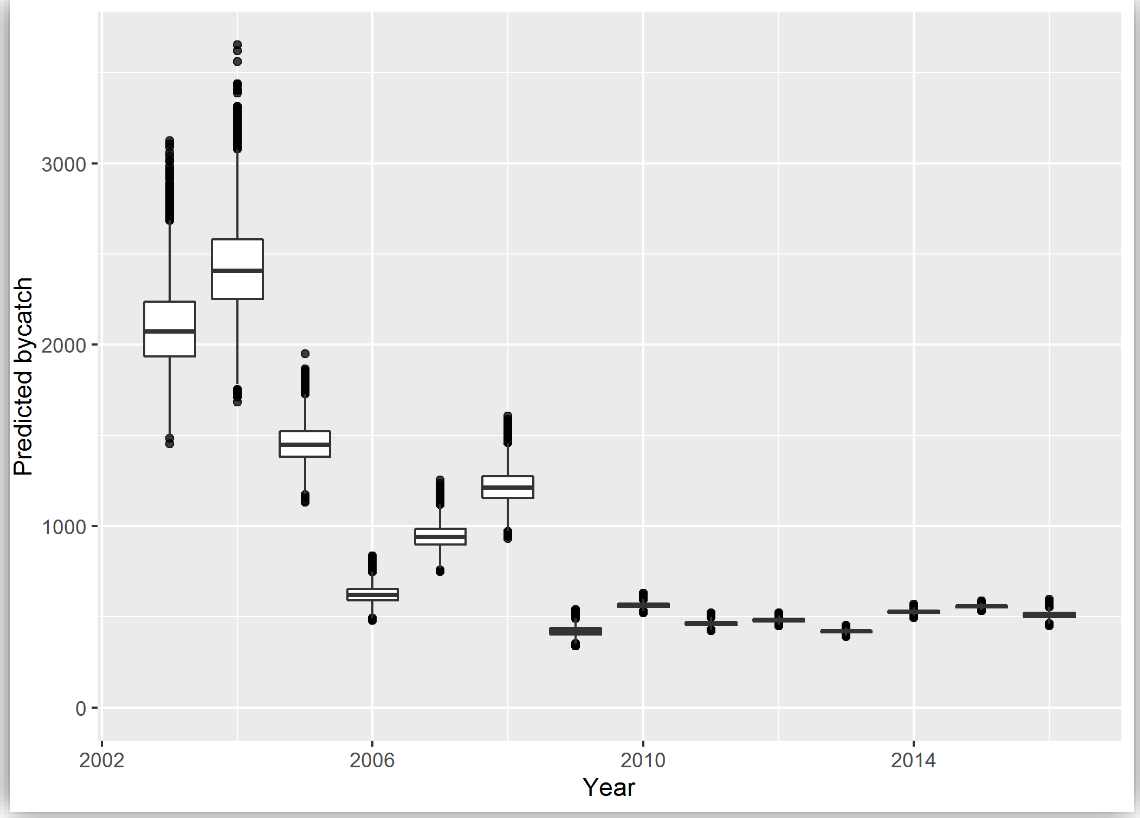


Figure 34. Predicted total annual oceanic whitetip bycatch (numbers) by year for large-scale purse seine fleets (Peatman et al. 2017).

However, Peatman et al. (2017) provided modeled estimations 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 for this period (*n* = 1,822), for the large-scale purse seine fleet, are provided in (Peatman et al. 2017). An update was provided by Peatman et al. (2018a) for the 2017 fishing season and reported 721 sharks, resulting in total median bycatch estimate of 13,882 from 2003 to 2017. As Peatman et al. (2018a) discusses, observer coverage was only 20% and the bycatch estimates should be considered preliminary. However, at this time, this is considered the best scientific data available for this fishery and Region. Furthermore, as previously noted, observer coverage in the United States fleet is 100%.

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 17). 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 17. 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**
Scalloped Hammerhead Shark	14 (5 – 24)	0 (0 – 0)	23 (9 – 37)	0 (0 – 0)
Hammerhead Shark (spp not identified)	3 (0 – 5)	0 (0 – 0)	1 (0 – 3)	0 (0 – 0)
Percent Observer Coverage (2013-2018): 2.4% (2.1 – 2.8%)				

Giant manta rays were not commonly identified in the WCPO purse seine fisheries until about 2011 (NMFS 2019c) and there is limited information on their capture rates. Tremblay-Boyer and Brouwer (2016) estimated CPUE rates for non-key shark and ray species including giant manta rays. From 2005 to 2016 they found increasing trends in CPUE for giant manta rays but Tremblay-Boyer and Brouwer (2016) caution that these trends represent increases in reporting the species rather than increasing abundance. The average interaction rate from 2011 to 2016 (eliminating earlier years with low recording rates) based on the information presented in Tremblay-Boyer and Brouwer (2016) is 0.042 giant manta ray per set. The mean number of WCPO sets per year from 2013 to 2018 is 58,902 (WCPFC 2020). Combining this information, we estimate that approximately 2,474 (58,902*0.042) individuals identified as giant manta ray may have been captured in the WCPO fisheries each year from 2013 to 2018. As noted previously and discussed in detail in Section 5.2.1.3, there is likely a substantial misidentification rate for giant manta ray in this fishery, therefore this number is likely high. Additionally, based on how the data is amalgamated by the WCPFC, United States data is presumed to be present although not specifically identified as such. From 2013 to 2018, the United States WCPO purse seine fishery captured an average of 221 giant manta ray each year (exclusive of the unidentified mobulid category as reported in Table 53; again we caveat this with the likelihood of this estimate being high due to misidentification). In spite of the likely misidentification of giant manta ray, assuming the misidentification rate is similar throughout the WCPO purse seine fleets, we estimate that the United States WCPO purse seine fishery captures approximately 9% ([221/2474] x100) of the giant manta ray captured in the Region annually from 2013 to 2018, which is consistent with the portion of the fishing applicable the United States which is approximately 13% based on the number of sets per year from 2013 to 2018.

When considering estimated interactions with unidentified records prorated for the species (see Table 53), this suggests the United States fleet could be responsible for upwards of 12% ([298/2474]x100) of the entire fleet's annual interactions, over that six-year period (2013 to 2018). This is using the average annual number of interactions over that six-year time span (*n*= 298). However, we do not know how many giant manta rays might have inadvertently been recorded as an unidentified ray in the greater WCPO PS fleet. We expect that our estimate may be high; however, we do not have enough information available to be sure. In any case, our analysis suggests the United States fleet likely captured at least 9% of the giant manta rays annually in this fishery with the available information, but we do not know to what extent this metric may be under or over estimated without additional data from the greater WCPO PS fleet.

4.3.5 Fish Aggregating Devices (FADs) used in Purse Seine Fisheries

As discussed in the *Description of the Proposed Action* Section, purse seine sets are categorized either as “unassociated” or as “associated” when sets are made on FADs. These are man-made (artificial) drifting FADs (dFADs), or they can be sets made on natural objects such as logs, dead whales, or other floating debris (Hare et al. 2015). NMFS IFD is not aware of any anchored FADs (aFADs) being used in this fleet. 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. 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 believed to enhance tuna aggregation. The depth of those appendages can vary, from 10 to 120 m in depth, depending on fleet, ocean and season, with a tendency for that depth to increase in recent years (Pilling et al. 2018).

In the *Description of the Proposed Action* section, we discuss the history of CMMs issued by the WCPFC for FAD use by all commission members, cooperating non-members, and participating territories, and that NMFS subsequently issued rules to implement these measures in the United States WCPO purse seine fishery. One purpose of the conservation measures is to reduce entanglements of marine life. The measure adopted at the end of 2018 (CMM 2018-01), to be implemented by January 2020, requires that FADs be non-entangling and comply with the following:

- The floating or raft part (flat or rolled structure) of the FAD can be covered or not. To the extent possible the use of mesh net should be avoided. If the FAD is covered with mesh net, it must have a stretched mesh size less than 7 cm (2.5 inches) and the mesh net must be well wrapped around the whole raft so that there is no netting hanging below the FAD when it is deployed.
- The design of the underwater or hanging part (tail) of the FAD should avoid the use of mesh net. If mesh net is used, it must have a stretched mesh size of less than 7 cm (2.5 inches) or tied tightly in bundles or “sausages” with enough weight at the end to keep the netting taut down in the water column. Alternatively, a single weighted panel (less than 7 cm (2.5 inches) stretched mesh size net or solid sheet such as canvas or nylon) can be used.

The measure also encourages CCMs to promote the use of natural or biodegradable materials in the construction of FADs and for the WCPFC Scientific Committee to continue research on the use of non-entangling and biodegradable material. We have limited data on the impacts of entanglement in FADs for the United States WCPO purse seine fishery and this biological opinion only considers the impacts on sea turtles based on very limited observer data. However, in general, we anticipate that this CMM will reduce unobserved entanglement injuries and mortalities of sharks and sea turtles.

Abandoned FADs contribute to the marine debris problem within pelagic and coastal ocean habitats (Consoli et al. 2020; Sinopoli et al. 2020). While abandoned non-entangling FADs may not pose an entanglement risk, they are still a source of pollution that can degrade coastal habitats used as foraging areas by sea turtles (Musick and Limpus 1996) and breeding areas by sharks and rays (e.g. Duncan and Holland 2006). It has been shown that FADs with GPS transmitters have grounded in nearshore environments and can cause damage (State of Hawaii DLNR 2014; Balderson and Martin 2015; Maufroy et al. 2016, 2018; Davies et al. 2017). However, we do not have any tracking data for FADs lost by this fishery and do not know the final resting place of any lost FADs. Consoli et al. (2020) suggests several mechanisms for preventing or mitigating the effects of FADs becoming marine debris, these include: use of biodegradable materials, using instrumented buoys to find and remove unwanted FADS (Grande et al. 2018), and payment for retrieved gear.

4.3.5.1 Marine Mammals

There have been no observations or reports of ESA-listed whales associated with FADs in the non- United States WCPO purse seine fisheries. It is possible that the nets associated with FADs can pose an entanglement risk, however, we do not have enough information to assess if the use of FADs impacts the marine mammal populations considered in this biological opinion. Therefore, we do not expect their overall abundance or reproductive outputs to be affected by non-United States WCPO purse seine fisheries.

4.3.5.2 Sea Turtles

Man-made FADs are designed to maximize the potential catch of purse seine target species (Pilling et al. 2018). While enhancing the probability of catching tuna, the use of FADs may impact the rates of sea turtle bycatch. Molony (2005) found a significant difference in sea turtle capture rates among set types. Fewer than expected captures occurred on unassociated sets, baitfish-associated sets, and dFAD sets. More than expected captures occurred on log sets, aFAD sets, and whale sets. The exception was for green sea turtles, for which more than 60% of captures occurred in unassociated sets. Similarly, SPC-OFP (2001) found the highest CPUE rate of sea turtles with logs and aFADS, but catch rates were lower for dFADs than for unassociated sets. Bourjea et al. (2014) found higher catch per set rates for sea turtles on dFAD sets compared to unassociated sets in the Atlantic and Indian Ocean purse seine fisheries. In the WCPO purse seine fisheries, most fleets, including the United States, exclusively use dFADs (Itano et al. 2004; Lopez et al. 2017), hence we anticipate lower sea turtle interaction rates on FAD sets compared to unassociated sets.

The new regulation limiting FADs to 350 per vessel (83 FR 33851) may also reduce sea turtle interaction rates, however Escalle et al. (2018) estimated that less than 16% of vessels purse seining in the WCPO currently have more than 350 FADs per year.

FADs also pose an entanglement risk to sea turtles (FAO 2010; Pilling et al. 2018). Pilling et al. (2018) report that in 2015, six sea turtle interactions were noted as entanglements with FAD netting, with three of those being mortalities in the WCPO. Bourjea et al. (2014) examined data for floating object observations in the Atlantic and Indian Oceans between 2003 and 2011. In total, 354 sea turtles were recorded on 14,124 floating objects. The most common species in the Indian Ocean was the olive ridley sea turtle, followed by hawksbill and green sea turtles; leatherback sea turtle sightings were rare. Of the observed sea turtles, 21% were entangled dead, 47% were entangled alive and released, and 32% were untangled. Entanglement and drowning with both the surface and the submerged netting is likely for sea turtles, but this risk has not been quantified.

4.3.5.3 Elasmobranchs

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). However, there are no estimates in the literature for oceanic whitetip shark interaction rates with FADs, 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. There is one record of an oceanic whitetip shark entangled in a FAD in the WCPO purse seine fishery in 2018 (NMFS unpublished data 2019). However, more information is needed regarding these types of events. Shark and ray bycatch may also be higher on FAD sets compared to unassociated sets (Leroy et al. 2013).

In purse seine sets, *Manta* spp. are 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). Hence, we anticipate that the use of FADs has a limited impact on giant manta ray populations.

4.4 Vessel Strikes

Marine habitats occupied by ESA-listed species under NMFS' jurisdiction often feature both heavy commercial and recreational vessel traffic. 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.

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. Five of the eleven confirmed species identified are found in the WCPO *Action Area* including blue, fin, humpback, sei, and sperm whales (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). Far fewer reports exist of strikes to blue (eight records), and sei whales (three records) (Jensen and Silber 2003). 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). However, this is not unexpected given the low likelihood of a collision being reported.

Van Waerebeek et al. (2006) on the other hand reported 248 whale strikes in the Southern Hemisphere of which 170 were attributed to large whale species (southern right, blue, sei, fin, Bryde's and humpbacks). Records included those previously reported by Jensen and Silber (2003). However, we conclude as with the data from Jensen and Silber (2003) quantification of the number of incidents specifically within the *Action Area*, is unknown.

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 sea 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).

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. (2008c) reported that 2.5% of green sea 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 researchers 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 2018d, 2019b), we do not have this level of detail for the *Action Area* but we estimate that there are fewer vessel strikes in the ports most frequency used by the United States WCPO purse seine fishery (i.e. Pago Pago and the Republic of the Marshall Islands) because those ports are smaller, with less vessel activity than the port of Honolulu and surrounding areas, and have observers onboard. There are fewer turtles in proximity to these ports as well. NMFS estimates that 250 green sea turtles are struck by vessels annually around Hawaii with a mortality of 95-100% (NMFS 2018d); we estimate that the turtles killed by vessel strikes in the *Action Area* is less than the 250 green sea turtles struck around Hawaii. This data provides a good comparison as it quantifies the number of sea turtles being struck in heavy maritime use areas which are unlike the ports used by the vessels in this fishery and represents the best source of data available for applicable vessel strike metrics. The less frequently-used ports identified in Table 1 have very low traffic attributable to the United States WCPO purse seine fishery that would not substantially increase the vessel traffic to the ports or the risk of vessel strike to sea turtles.

No turtles are known to have been struck in the ASLL fishery which occurs in a portion of the *Action Area*. The number of vessels participating in the ASLL fishery has fluctuated, albeit 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. As a result, the risk of collision with an ASLL fishery vessel is considered unlikely. Furthermore, NMFS previously determined vessel strikes would have insignificant effects to sea turtles in ASLL fleet (NMFS 2015a).

Manta ray aggregation sites are sometimes in areas of high maritime traffic, and therefore giant manta rays are at potential risk of being struck and killed by boats (Marshall et al. 2011; Graham et al. 2012). Internet searches also reveal photographs of mantas species including giant manta rays, 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; Pate and Marshall 2019), potentially similar to the impacts of shark or orca attacks. Stewart et al. (pers. comm. to J. Rudolph, 11 May 2020 and October 7, 2020) has determined a 19% injury rate which includes both vessel strikes and fishery interactions in the nearshore Mexican population of giant manta rays (identified in Stewart et al. (2016a)), and a 1.5% injury rate for the offshore population (unpublished data). The nearshore population has an overall injury rate of approximately 30%, with 11% of those from natural causes (i.e. shark bites) and the offshore had a total observed injury rate of 3% of the population with approximately 1.5% from natural causes (Stewart pers. comm. to J. Rudolph, October 7, 2020). However, there is very little quantitative information on the frequency of vessel strikes on giant manta rays. Additionally, Pate and Marshall (2019) documented rapid healing rates for significant propeller strikes between 15 and 40 days. Thus, injuries may go undetected during brief visual encounters.

We found no information to indicate that vessel strikes are an issue for oceanic whitetip sharks or Indo-West Pacific scalloped hammerhead sharks after a careful review of available literature.

4.5 Pollution

Many different types of pollution can adversely affect ESA-listed 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 ESA-listed resources.

4.5.1 Oil Pollution

Oil released into the marine environment contains aromatic organic chemicals known to be toxic to a variety of marine life (Yender et al. 2002). Oil spills can impact wildlife directly through three primary pathways: (1) ingestion—when animals swallow oil particles directly or consume prey items that have been exposed to oil; (2) absorption—when animals come into direct contact with oil; and (3) inhalation—when animals breath volatile organics released from oil or from "dispersants" applied by response teams in an effort to increase the rate of degradation of the oil in seawater.

Direct exposure to oil can cause acute damage including skin, eye, and respiratory irritation, reduced respiration, burns to mucous membranes such as the mouth and eyes, diarrhea, gastrointestinal ulcers and bleeding, poor digestion, anemia, reduced immune response, damage to kidneys or liver, cessation of salt gland function, reproductive failure, and death (NOAA 2003, 2010). Nearshore spills or large offshore spills that reach shore can oil beaches on which sea turtles lay their eggs, causing birth defects or mortality in the nests (NOAA 2003, 2010). Disruption of other essential behaviors, such as breeding, communication, and feeding may also occur.

The loss of invertebrate communities due to oiling or oil toxicity could also decrease prey availability for hawksbill, and loggerhead sea turtles (NOAA 2003). Sea turtle species which commonly forage on crustaceans and mollusks may be vulnerable to oil ingestion due to oil adhering to the shells of these prey and the tendency for these organisms to bioaccumulate toxins found in oil (NOAA 2003). Seagrass beds may be particularly susceptible to oiling as oil contacts grass blades and sticks to them, hampering photosynthesis and gas exchange (Wolfe et al. 1988). If spill cleanup is attempted, mechanical damage to seagrass can result in further injury and long-term scarring. Loss of seagrass due to oiling would be important to green sea turtles, as this is a significant component of their diets (NOAA 2003). Sea turtles are also known to ingest and attempt to ingest tar balls, which can block their digestive systems, impairing foraging or digestion and potentially causing death (NOAA 2003).

Oceanic whitetip sharks, Indo-West Pacific scalloped hammerhead sharks, and giant manta rays can be exposed to oil and its associated chemical components either through ingestion of prey or when contaminated water travels across the surface of their gills. Sampling of sharks exposed to oil during Deep Water Horizon found physiological signs of elevated polycyclic aromatic hydrocarbons (PAHs) exposure but showed no evidence for chromosomal or higher-level impacts to sharks (Heithaus et al. 2014). However, some shark species exhibited greater effects of PAH exposure to oil, likely due to remaining in the area over longer periods than other species (Walker 2011). Kibria and Haroon (2015) and Lee et al. (2015) provided an extensive literature review of pollutant bioacculmation in sharks and described a range of effects from cardiac and birth defects to infertility, endocrine disruption and immune system. Cardiac development was also shown to be effected in other fish species (tuna; Incardona et al. 2014).

Cetaceans exposed to oil inhale contaminated air, aspirate liquid oil, ingest contaminated suspended sediments or prey, and swim through contaminated water. Respiratory impacts were the most obvious adverse health effect noted, but disruption of the hypothalamicpituitary-adrenal axis, impaired stress response and compromised immune function were also noted in the literature (National Academies of Sciences and Medicine 2016 as cited in NMFS 2018d).

Spills and discharges may consist of toxic hydrocarbon-based chemicals such as fuel oils, gasoline, lubricants, or other toxicants. The primary pathway for turtle, shark, and giant manta ray exposure to toxins is likely from land-based sources and on-shore or near-shore sources of chronic discharges. Major oceanic spills are rare but acute events. Federal laws and regulations strictly regulate the discharge of oil, garbage, waste, plastics, and hazardous substances into ocean waters under a variety of acts, including the Clean Water Act, as amended by the Oil Pollution Act of 1990, the Act to Prevent Pollution from Ships, MARPOL 1973/1978, and the Ocean Dumping Act. Violations of these laws may result in severe civil penalties, criminal fines, and imprisonment

4.5.2 Marine Debris

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 POPs) 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 unattended for decades (ghost fishing), killing large numbers of marine animals through entanglement. As noted above, FADs have also become a growing concern. A recent study estimated that over 20,000 FADs were deployed in the region in 2018 and that 51.8% were classified as lost; 10.1% were retrieved; 6.7% were beached; 15.4% were sunk, stolen, or had a malfunctioning buoy; and 10.4% were deactivated by the fishing company and left drifting (Escalle et al. 2019). These FADs continue to aggregate fish for an unknown period of time until they decompose, sink, or ground.

Marine debris is a significant concern for sea turtles, fish, and marine mammals. The initial 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). Some sea turtles may eat plastic because it closely resembles jellyfish, a common natural prey item (Schuyler 2013). 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 dead stranded green sea 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, and 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. Also, based on this model, olive ridley turtles are the most at-risk species, while green, loggerhead, and leatherback turtles were also found to be at a high and increasing risk from plastic ingestion (Schuyler 2013). 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 (Schuyler et al. 2015). 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 1990; Lutcavage et al. 1997; Laist et al. 1999). Turtles are particularly vulnerable to ghost nets due to their tendency to use floating objects for shelter and as foraging stations (Kiessling 2003; Dagorn et al. 2013).

Marine mammals are also particularly susceptible to the threats associated with marine debris and many cases of ingestion and entanglement have been reported around the world (Poeta et al. 2017). Baulch and Perry (2014) found that the proportion of cetacean species ingesting debris or becoming entangled in debris is increasing. Based on stranding data, they found that recorded rates of ingestion have increased by a factor of 1.9 and rates of entanglement have increased by a factor of 6.5 over the last 40 years (1970-2010). Ingestion of marine debris can also have fatal consequences for large whales. In 2008, two male sperm whales stranded along the northern California coast with large amounts of fishing net scraps, rope, and other plastic debris in their stomachs. One animal had a ruptured stomach, the other was emaciated, and gastric impaction was suspected as the cause of both deaths (Jacobsen et al. 2010).

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

Marine debris may entangle or be ingested by oceanic whitetip sharks (Compango 1984), leading to injury or possibly starvation, and derelict fishing gear may cause entanglement and possibly drowning. However, data are not available to estimate the number of oceanic whitetip mortalities resulting from marine debris in the *Action Area*. Furthermore, we did not find any evidence of interactions of this nature (with marine debris and the Indo-west Pacific scalloped hammerhead shark) in the available literature. However, they could be entangled or ingested as noted in other shark species.

4.5.3 Anthropogenic Sound

The ESA-listed species that occur in the *Action Area* are regularly exposed to multiple sources of anthropogenic sounds. Anthropogenic noises 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 noise at any one place and time. These noises include transportation, dredging, construction; oil, gas, and mineral exploration in offshore areas; geophysical (seismic) surveys; sonars; explosions; and ocean research activities (Richardson et al. 1995). The species considered in this opinion may be impacted by anthropogenic sound in various ways. It can produce direct physical harm or may illicit behavioral responses including, but not limited to, cessation of feeding, resting, or social interactions, changes in habitat to avoid areas of higher sound levels, or changes in diving behavior (MMC 2007).

Anthropogenic sound in the *Action Area* is generated by commercial and recreational ships, aircraft, sonar, ocean research activities, dredging, construction, offshore mineral exploration, military activities, seismic surveys, and other human activities (NRC 2003b). These activities occur within the *Action Area* to varying degrees throughout the year. 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 noise are becoming both more pervasive and more powerful, increasing both oceanic background sound levels and peak intensity levels (Hildebrand 2004).

This section is divided into subsections addressing the impacts to listed species in the *Action Area* from the following major sources of anthropogenic sound sources: vessels and commercial shipping; seismic surveys; military training and testing activities; and active sonar.

4.5.3.1 Vessel sound and commercial shipping

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

4.5.3.2 Seismic Surveys

Offshore seismic surveys involve the use of high energy sound sources operated in the water column to probe below the seafloor. There are two major categories of seismic surveys: (1) deep seismic surveys which include ocean bottom, vertical seismic profile or borehole, 2-dimensional, 3-dimensional, 4-dimensional and wide azimuth surveys; and (2) high resolution surveys. Deep seismic survey acoustic sources consist of airgun arrays while receiver arrays consist of hydrophones or geophones encased in plastic tubing called streamers. When an airgun array fires an acoustic energy pulse is emitted and reflected or refracted back from the seafloor. These reflected/refracted acoustic signals create pressure fluctuations, which are detected and recorded by the streamers. Seismic airguns generate intense low-frequency sound pressure waves capable of penetrating the seafloor and are fired repetitively at intervals of 10 to 20 seconds for extended periods (NRC 2003a). Most of the energy from airguns is directed vertically downward, but significant sound emission also extends horizontally. Peak SPLs from airguns usually reach 235 to 240 decibels at dominant frequencies of five to 300 Hz (NRC 2003a). High-resolution surveys collect data on surface and near-surface geology used to identify archaeological sites, potential shallow geologic and manmade hazards for engineering, and site planning for bottom-founded structures.

High-resolution surveys may use airguns but also use other sound sources such as sub-bottom profilers (at 2.5-7 kHz), echosounders (single-beam at 12-240 kHz; multibeam at 50-400 kHz), boomers (at 300-3,000 Hz), sparkers (at 50-4,000 Hz), compressed high intensity radar pulse sub-bottom profiler (at 424 kHz), pingers (at 2 kHz), and side-scan sonars (16-1,500 kHz). These sound sources are typically powered either mechanically or electromagnetically. 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). McCauley et al. (2000) conducted trials with caged sea turtles and an approaching-departing single air gun to gauge behavioral responses of green and loggerhead sea turtles. Their findings showed behavioral responses to an approaching airgun array at 166 dB re: one micro Pascal rms and avoidance around 175 dB re: 1 micro Pascal rms. From measurements of a seismic vessel operating 3-dimensional airgun arrays in 100 to 120 m water depth this corresponds to behavioral changes at around two kilometers and avoidance around one kilometer.

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 (Casper et al. 2012). Particle motion is presumably the only sound stimulus that can be detected by elasmobranchs (Casper et al. 2012). Given their assumed hearing range, elasmobranchs are anticipated to be able to detect the low frequency sound from an airgun array if exposed. However, the duration and intensity of low-frequency acoustic stressors and the implementation of conservation measures will likely minimize the effect this stressor has on elasmobranchs. Furthermore, although some elasmobranchs have been known to respond to anthropogenic sound, in general 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).

Popper et al. (2014) concluded that the relative risk of fishes with no swim bladders exhibiting a behavioral response to low-frequency active sonar was low, regardless of the distance from the sound source. The authors did not find any data on masking by sonar in fishes, but concluded that if it were to occur, masking will result in a narrow range of frequencies being masked (Popper et al. 2014). Popper et al. (2014) also concluded that the risk of mortality, mortal injury, or recoverable injury for fish with no swim bladders exposed to low frequency active sonar was low, regardless of the distance from the sound source.

The precise expected response of ESA-listed elasmobranchs to low-frequency acoustic energy is not completely understood due to a lack of sufficient experimental and observational data for these species. However, given the signal type and level of exposure to the low frequency signals used in seismic survey activities, we do not expect adverse effects (including significant behavioral adjustments, TTS, PTS, injury, or mortality). The most likely response of ESA-listed elasmobranch exposed to seismic survey activities, 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 noise on elasmobranchs in the *Action Area*.

Exposure of marine mammals to very strong sound pulses can result in physical effects, such as changes to sensory hairs in the auditory system, which may temporarily or permanently impair hearing. Threshold shift depends upon the duration, frequency, sound pressure, and rise time of the sound. TTS results in a temporary hearing change (Finneran and Schlundt 2013), and can last minutes to days and full recovery is expected. However, a study on mice has shown that although full hearing can be regained from TTS (i.e., the sensory cells actually receiving sound are normal), damage can still occur to nerves of the cochlear nerve leading to delayed but permanent hearing damage (Kujawa and Liberman 2009). At higher received levels, particularly in frequency ranges where animals are more sensitive, PTS can occur, meaning lost auditory sensitivity is unrecoverable. These conditions can result either from a single pulse or from the accumulated effects of multiple pulses, in which case each pulse need not be as loud as a single pulse to have the same accumulated effect.

TTS and PTS are generally specific to the frequencies over which exposure occurs but can extend to a half-octave above or below the center frequency of the source in tonal exposures (less evident in broadband noise such as the sound sources associated with the proposed action) (Schlundt et al. 2000; Kastak 2005; Ketten 2012). For TTS, full recovery of the hearing loss (to the pre-exposure threshold) has been determined from studies of marine mammals, and this recovery occurs within minutes to hours for the small amounts of TTS that have been experimentally induced (Nachtigall et al. 2004; Finneran et al. 2005). The recovery time is related to the exposure duration, sound exposure level, and the magnitude of the threshold shift, with larger threshold shifts and longer exposure durations requiring longer recovery times (Finneran et al. 2005; Mooney et al. 2009a, 2009b). For an animal to experience a large threshold shift, it would have to approach close to the sonar source or remain near the sound source for an extended period.

Interference, or masking, occurs when a sound is a similar frequency and similar to or louder than the sound an animal is trying to hear (Francis and Barber 2013). Masking can interfere with an individual’s ability to gather acoustic information about its environment, such as predators, prey, conspecifics, and other environmental cues (Marshall 1995 as cited in NFMS 2018d). This can result in loss of environmental cues of predatory risk, mating opportunity, or foraging options (Francis and Barber 2013).

There is frequency overlap between air gun sounds and vocalizations of ESA-listed whales, particularly baleen whales but also sperm whales. Seismic surveys could mask whale calls at some of the lower frequencies. This could affect communication between individuals, affect their ability to receive information from their environment, or affect sperm whale echolocation (Evans 1998 as cited in NFMS 2018d; NMFS 2006h as cited in NFMS 2018d). Most of the energy of sperm whales clicks is concentrated at 2 to 4 kHz and 10 to 16 kHz, and though the findings by Madsen et al. (2006) suggest frequencies of seismic pulses can overlap this range, the strongest spectrum levels of air guns are below 200 Hz. Any masking that might occur would likely be temporary because seismic sources are not continuous and the seismic vessel would continue to transit through the area.

Given the disparity between sperm whale echolocation and communication-related sounds with the dominant frequencies for seismic surveys, masking is not likely to be significant for sperm whales (NMFS 2006h as cited in NFMS 2018d). Overlap of the dominant low frequencies of air gun pulses with low frequency baleen whale calls would be expected to pose a somewhat greater risk of masking.

We expect the greatest response to air gun sounds in terms of number of responses and overall impact to be in the form of changes in behavior. Listed individuals may briefly respond to underwater sound by slightly changing their behavior or relocating a short distance, in which case the effects are unlikely to have fitness consequences for the individual and would not likely be significant at the population level. Displacement from important feeding or breeding areas over a prolonged period would come at higher costs to individual animals and ultimately the populations that they represent. Marine mammal responses to anthropogenic sound vary by sound source and its frequency and severity or intensity, as well as species, state of maturity, prior exposure, current activity, reproductive state, time of day, and other factors (Ellison et al. 2012); this is reflected in a variety of aquatic, aerial, and terrestrial animal responses to anthropogenic noise that may ultimately have fitness consequences (Francis and Barber 2013).

Whales continue calling while seismic surveys are operating locally (Richardson et al. 1986; McDonald et al. 1995; Greene Jr et al. 1999; Madsen et al. 2002; Tyack et al. 2003; Nieukirk et al. 2004; Jochens et al. 2006; Smultea et al. 2013). Some fin, and sperm whales stopped calling for short and long periods apparently in response to air guns (Bowles et al. 1994; McDonald et al. 1995). Fin whales (presumably adult males) engaged in singing in the Mediterranean Sea moved out of the area of a seismic survey while air guns were operational as well as for at least a week thereafter (Castellote et al. 2012). Sperm whales, at least under some conditions, may be particularly sensitive to air gun sounds, as they have been documented to cease calling in association with air guns being fired hundreds of kilometers away (Bowles et al. 1994). Other studies have found no response by sperm whales to received air gun sound levels up to 146 dB re 1 µPap-p (McCall Howard 1999 as cited in NFMS 2018d; Madsen et al. 2002). Some exposed individuals may cease calling, however this would be considered temporary.

There are numerous studies of the responses of some baleen whale to air guns. Although responses to lower-amplitude sounds are known, most studies seem to support a threshold of approximately 160 dBrms re 1 µPa as the received sound level to cause behavioral responses other than vocalization changes (Richardson et al. 1995; NMFS 2013b). Activity of individuals seems to influence response (Robertson et al. 2013), as feeding individuals respond less than mother/calf pairs and migrating individuals (Malme et al. 1984; Richardson et al. 1995 as cited in NMFS 2013b; Miller et al. 1999 as cited in NMFS 2013c; Richardson et al. 1999; Miller et al. 2005 as cited in NMFS 2013b; Harris et al. 2007 as cited in NMFS 2013b). Surface duration decreased markedly during seismic sound exposure, especially while individuals were engaged in traveling or non-calf social interactions (Robertson et al. 2013).

Observational data are sparse for specific baleen whale life histories (breeding and feeding grounds) in response to air guns. Available data support a general avoidance response. Some fin and sei whale sighting data indicate similar sighting rates during seismic versus non-seismic periods, but sightings tended to be further away and individuals remained underwater longer (Stone 2003; Stone and Tasker 2006). Other studies have found at least small differences in sighting rates (lower during seismic activities) as well as whales being more distant during seismic operations (Moulton and Miller 2005; Moulton et al. 2006a as cited in NFMS 2018d; Moulton et al. 2006b as cited in NFMS 2018d). When spotted at the average sighting distance, individuals would have likely been exposed to approximately 169 dB re 1 µParms (Moulton and Miller 2005).

Sperm whale response to air guns has thus far included mild behavioral disturbance (temporarily disrupted foraging, avoidance, cessation of vocal behavior) or no reaction. Several studies have found Atlantic sperm whales to show little or no response (Stone 2003; Moulton and Miller 2005; Madsen et al. 2006; Moulton et al. 2006a as cited in NFMS 2018d; Stone and Tasker 2006; Weir 2008; Miller et al. 2009). Detailed study of Gulf of Mexico sperm whales suggests some alteration in foraging from less than 130 to 162 dB re 1 μPap–p, although other behavioral reactions were not noted by several authors (Gordon et al. 2004; Gordon et al. 2006 as cited in NMFS 2018c; Jochens et al. 2006; Madsen et al. 2006; Winsor and Mate 2006 as cited in NMFS 2018c). This has been contradicted by other studies, which found avoidance reactions by sperm whales in the Gulf of Mexico in response to seismic ensonification (Mate et al. 1994; Jochens et al. 2006). Johnson and Miller (2002 as cited in NFMS 2018d) noted possible avoidance at received sound levels of 137 dB re 1 μPa. Other anthropogenic sounds, such as pingers and sonars, disrupt behavior and vocal patterns (Watkins and Schevill 1975 as cited in NFMS 2018d; Watkins and Tyack 1991; Goold 1999 as cited in NFMS 2018d). Miller et al. (2009 as cited in NFMS 2018d) found sperm whales to be generally unresponsive to air gun exposure in the Gulf of Mexico, with possible but inconsistent responses that included delayed foraging and altered vocal behavior. Displacement from the area was not observed. Winsor et al. (2017) did not find a nonrandom distribution of satellite-tagged sperm whales at and beyond 5 km from seismic air gun arrays, suggesting individuals were not displaced or move away from the array at and beyond these distances in the Gulf of Mexico (Winsor et al. 2017). However, no tagged whales within 5 km were available to assess potential displacement within 5 km (Winsor et al. 2017). The lack of response by this species may in part be due to its higher range of hearing sensitivity and the low-frequency (generally less than 188 Hz) pulses produced by seismic air guns (Richardson et al. 1995 as cited in NMFS 2013b). Sperm whales are exposed to considerable energy above 500 Hz during the course of seismic surveys (Goold and Fish 1999), so even though this species generally hears at higher frequencies, this does not mean that it cannot hear air gun sounds. Breitzke et al. (2008) found that source levels were approximately 30 dB re 1 μPa lower at 1 kHz and 60 dB re 1 μPa lower at 80 kHz compared to dominant frequencies during a seismic source calibration. Adverse effects to whales likely occur within the *Action Area* from anthropogenic noises, we do not have sufficient information to describe the significance of this stressor to listed whales as many countries traverse these waters and may conduct scientific studies of this nature at any time. To date, NMFS has not exempted the take of ESA-listed species under consideration in this biological opinion within the *Action Area* for seismic survey activities.

4.5.3.3 Active Sonar

Sonar systems use acoustic energy to locate objects beneath the ocean surface. They are designed to provide information about objects within the water column (e.g. schools of fish), at the sea bottom (e.g. locating shipwrecks), or within the sediment (e.g. describing sediment composition; Hildebrand 2004). Active sonar emits high-intensity acoustic energy, usually in the form of a focused beam, and receives reflected and/or scattered energy, which provide the information on the objects of interest. Sonar systems have both civilian and military applications. For purposes of this discussion, sonar systems can be categorized as low frequency (< 1000 Hz), mid-frequency (1 – 20 kHz), and high frequency (> 20 kHz; Hildebrand 2004). Sonar systems are widely used throughout the world's oceans and are likely to impact ESA-listed species in the *Action Area*.

Military Sonars

Military sonars generally operate at higher source levels, with broader frequency ranges than civilian sonars, and are used for target detection, localization, and classification. Low Frequency Active (LFA) sonars are used for broad-scale surveillance; they are designed to allow submarine tracking over scales of many hundreds to thousands of kilometers. Specialized support ships are used to deploy LFA sonars, which consist of a set of transmitters suspended vertically below the ship. However, we cannot estimate the actual number of exposures, which may occur by other military entities at this time. The geographical areas where the United States conducts these types of operations does not overlap with the *Action Area* of this proposed action (see NMFS 2018d). However, we discuss United States operations as a surrogate for those operations which could be conducted by various Nations within the *Action Area* in order to clarify the potential impacts these species may face.

The United States Navy's Surveillance Towed Array Sensor System (SURTASS) LFA sonar is a long-range system that transmits in the low-frequency band and is composed of both active and passive components, the active component is the LFA sonar array (United States Navy 2018). The LFA sonar uses an array of projectors operating in the frequency range of 100 to 500 Hz, with 215 dB re 1µPa @ 1 m source level for each projector (United States Navy 2018). These systems are designed to project beams of energy in a horizontal direction. The effective source level of an LFA array, when viewed in the horizontal direction, can be 235 dB re 1µPa @ 1 m or higher. The signal includes both continuous-wave (CW) and frequency-modulated (FM) components with a bandwidth of approximately 30 Hz. A ping sequence can last 6 to 100 seconds, with a time between pings of 6 to 15 minutes and a typical duty cycle of 10 to 15 percent. Signal transmissions are emitted in patterned sequences that may last for days or weeks (United States Navy 2018). The frequency range of LFA sonar is within the hearing range of the ESA-listed whales considered in this opinion, with greater overlap for sei whales which are considered low-frequency hearing group cetaceans (7 Hz to 35 kHz) and less overlap for sperm whales which are considered mid-frequency hearing group cetaceans (150 Hz to 160 kHz; NMFS 2018a). The frequency range also overlaps with what is known about hearing ability in sea turtles (~ 50 Hz to 5 kHz) and elasmobranchs (~30 to 150 Hz; Popper et al. 2014).

Mid-frequency tactical Anti-Submarine Warfare (ASW) sonars are designed to detect submarines over several tens of kilometers. They are incorporated into the hulls of submarine-hunting surface vessels such as destroyers, cruisers, and frigates. The AN/SQS-56 is one of the most advanced surface ship ASW sonar used by the United States Navy. The AN/SQS-56 sonar generates frequency-modulated pulses of 1-2 second duration in the 5-10 kHz band, at source levels of 223 dB re 1µPa @ 1 m (Evans and England 2001; D'Amico and Pittenger 2009). These sonars emit beams of sound in the horizontal direction. These systems are used to track both surface and submerged vessels, often picking up surface ships at greater range than most radar systems. Other mid-frequency military sonars in use by navies include depth sounders and communication sonars for inter-platform information exchange or device activation (Hildebrand 2004). The frequency range of these mid-frequency sonars is within the hearing range of the ESA-listed whales considered in this opinion (NMFS 2018d). The frequency range should be above what is understood to be the hearing ranges of sea turtles and elasmobranchs (Popper et al. 2014).

Military applications of high-frequency sonars are for weapons (torpedoes and mines) and weapon countermeasures (mine countermeasures or anti-torpedo devices). These sonars operate over ranges of a few hundred meters to a few kilometers. Minehunting sonars operate at tens of kHz for mine detection and above 100 kHz for mine localization. These sonars are highly directional and use pulsed signals. Other high-frequency military sonars include side-scan sonar for seafloor mapping, generally operated at frequencies near 100 kHz (Hildebrand 2004b). Given the frequency range of high-frequency sonar, these sources are only within the hearing range of the sperm whale for ESA-listed species considered in this opinion.

Commercial Sonar

Commercial sonars are designed for fish finding, depth sounding, and sub-bottom profiling. They typically generate sound in the mid- to high-frequency ranges of 3 to 200 kHz, with only a narrow frequency band generated by an individual sonar system. Source levels range from 150-235 dB re 1μPa @ 1 m (Hildebrand 2004). Multibeam echosounders, single beam echosounders, side-scan sonar, sub-bottom profilers are used by research institutions, governments and the oil and gas industry for mapping the seafloor. Commercial depth sounders and fishfinders are typically designed to focus sound into a downward beam. Depth sounders and sub-bottom profilers are operated primarily in nearshore and shallow environments, however, fish finders are operated in both deep and shallow areas (Hildebrand 2004).

A significant fraction of the world's marine vessels and boats, both commercial and private are equipped with some form of commercial sonar, especially for depth sounding and fish finding applications (Hildebrand 2004). These types of sonar typically operate above 50 kHz and would therefore only impact sperm whales of the ESA-listed species considered in this opinion.

4.5.4 Military Operations

As the *Action Area* contains very large portions of international waters and the EEZs of many countries, military activities may occur in the *Action Area* that are unknown to us. As a result, we do not have the ability to describe the risk posed to the species under consideration though we address United States resources in the Region.

The United States conducts military operations within the Mariana Islands Training and Testing (MITT) area. However, the MITT and the *Action Area* due not overlap in geographical space (NMFS 2017b). United States federal agencies working in the *Action Area* would be expected to complete ESA Section 7 consultation before any training or readiness operations could occur within the *Action Area*. However, we discuss United States operations as a surrogate for those operations which could be conducted by various Nations within the *Action Area* in order to clarify the potential impacts these species may face by other Nations. Nevertheless, routine operations such as patrolling, search and rescue, maritime inspections, flight operations, refueling, humanitarian aid, disaster relief, etc. may occur on a regular basis by either the United States or other military entities within the *Action Area*.

Other militaries may conduct training and testing activities and other military readiness activities in the *Action Area*. During training, existing and established weapon systems and tactics may be used in realistic situations to simulate and prepare for combat. Activities may include routine gunnery, missile, surface fire support, amphibious assault and landing, bombing, sinking, torpedo, tracking, and mine exercises. Testing activities may be conducted for different purposes and include at-sea research, development, evaluation, and experimentation.

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).

Additionally, the Republic of the Marshall Islands (RMI) has agreed to allow the United States Government to use certain areas within the RMI, including eleven islets at Kwajalein Atoll that are administered by the United States Army Kwajalein Atoll (USAKA). The *Action Area* under consideration in this biological opinion includes the RMI. Military training and munition testing activities in the RMI at USAKA are ongoing and are expected to continue into the future.

Reviewing applicable consultations, no species under consideration for the purse seine fishery were exempted take as defined under the ESA or by the Compact (48 United States C. 1921) for munition testing actions at USAKA. Most species considered herein are pelagic in nature and not expected to be present at the various missile impact sites on land. Exemptions were provided for other species applicable to the USAKA Environmental Standards, but not the ESA. Potential effects to sea turtles on land from these actions would be considered in consultation with the USFWS to which NMFS has no jurisdiction. However, these actions may affect most species as missile components which fall into the broad ocean area between the continental United States and the RMI, could impact an animal at the water's surface or expose animals to elevated sounds through sonic booms. NMFS has concluded the effects from falling missile components into the broad ocean area and effects from sonic booms to ESA-listed species are highly unlikely and therefore discountable (ex: see NMFS 2019e, 2019f).

The United States Air Force and other military entities may conduct flight operations within the *Action Area*, however affects to ESA-listed species are not expected as aircraft would be at altitude where auditory affects to underwater animals would be minimal. As discussed in the paragraph above, 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.

4.6 Synthesis of Baseline Impacts

The listed resources considered in this biological opinion have been exposed to a wide variety of the past and present state, federal, and private actions in the *Action Area*, which includes of all proposed federal projects in the *Action Area* that have already undergone formal or early consultation, and state or private actions that are contemporaneous with this consultation. While the impact of those activities on the status, trend or the demographic processes of threatened and endangered species is largely unknown, some are likely to have had and will continue to have lasting effects on the ESA-listed species considered in this consultation.

The preceding section of this biological opinion addresses global climate change, harvest, fisheries and fisheries bycatch, vessel strikes, pollution from chemicals and marine debris, and ocean noise from variety of sources and effects these stressors have on listed resources. Some of these stressors have resulted in mortality or serious injury to individual animals (e.g., fishing, vessel strike), whereas other stressors (e.g., noise) may induce sub-lethal responses like changes in behavior that could impact important biological functions such as feeding or breeding.

Of the stressors considered herein, the cumulative effect of fisheries in the *Action Area* likely has had some of the most serious and lasting effects on the ESA-listed species considered in this biological opinion, and the populations that comprise those species. This is because of the scale and the magnitude of the impact of the stressor– fisheries capture and injure or kill more individuals of these listed species, than we would expect would be injured or die from vessel strikes.

The stress regime created by activities discussed in this *Environmental Baseline* continue to be a threat for the fin, sei, or sperm whales. Threats like vessel strikes and marine pollution pose a risk to these species. Of the activities considered 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 these whale species. The risk of extinction due to environmental stochasticity and the threats from climate change and military activities likely occurs but is difficult to document or observe unless strandings occur. Lastly, the number of individuals that continue to be captured and killed in fisheries throughout the *Action Area* contributes 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. These species could also be impacted by additional activities in the *Action Area* but we do not know to what degree. They are most likely impacted to a greater extent further away from the *Action Area*.

The stress regime created by activities discussed in this *Environmental Baseline* continue to be a threat for the hawksbill sea turtle. Threats associated with climate change related to loss of nesting habitat, vessel strikes, and marine pollution pose a risk to these species however overexploitation has been the largest risk for hawksbills. However, the number of individuals that continue to be captured and killed in fisheries in the *Action Area* contributes to the extinction risk of the species.

The stress regime created by the activities discussed in this *Environmental Baseline* continues to have a serious and adverse impact on leatherback sea turtles. Leatherbacks face 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 high a 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 various 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 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.

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 on 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 the activities discussed in this *Environmental Baseline* continue to be a threat for loggerhead sea turtles, even though there is limited indication of population increases due to increased nesting at some of the key nesting beaches over the last decade. Still, with increasing nesting in some locations loggerhead sea turtles have a risk of extinction due to both environmental and demographic stochasticity, especially in nesting areas which do not overlap with the *Action Area*. The number of individuals that continue to be captured and killed in fisheries in the *Action Area* contributes to the 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 loggerhead sea turtles. However, loggerhead 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.

Of the different types of pollution that can adversely affect loggerhead sea turtles, the ingestion and entanglement risk of marine debris may pose the most serious threat specifically to loggerhead sea turtles (Polovina et al. 2004; Howell et al. 2010). 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). 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 loggerhead 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 olive ridley 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, a risk of extinction due to environmental stochasticity and the threats from fishing gear and vessel strikes is still possible. These species could also be impacted by additional activities in the *Action Area* but we do not know to what degree. They are most likely impacted to a greater extent further away from the *Action Area.* Particularly nesting locations and adjacent nearshore waters for the endangered breeding population.

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 bycatch in the industrial purse seine fishery and targeted harvest 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 track (Germanov et al. 2018), they can harbor high levels of toxins and persistent organic pollutants and transfer these toxins to the animal once ingested (Worm et al. 2017). If entangled in marine debris, the giant manta ray is at risk of severing of the cephalic and pectoral fin, severe injuries that can lead to a reduction in feeding efficiency and even death. The number of individuals that continue to ingest and become entangled in marine debris in the *Action Area* contributes to the increased extinction risk of the species.

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

The stressors discussed in this *Environmental Baseline* are also a threat for the oceanic whitetip and Indo-West Pacific scalloped hammerhead sharks. Both species 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 hammerhead sharks 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 bycatch 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.

5 EFFECTS OF THE ACTION

In *Effects of the Action* sections of biological opinions, NMFS presents the results of its assessment of the probable effects of federal actions on threatened and endangered species and designated critical habitat that are the subject of a consultation. According to 50 CFR 402.02, *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. Furthermore, 50 CFR 402.17 defines reasonably certain to occur as "A conclusion of reasonably certain to occur must be based on clear and substantial information, using the best scientific and commercial data available. Factors to consider when evaluating whether activities caused by the proposed action (but not part of the proposed action) or activities reviewed under cumulative effects are reasonably certain to occur include, but are not limited to: (1) past experiences with activities that have resulted from actions that are similar in scope, nature, and magnitude to the proposed action;(2) existing plans for the activity; and (3) any remaining economic, administrative, and legal requirements necessary for the activity to go forward.

As we described in the *Approach to the Assessment* section of this biological opinion, we organize our effects' analyses using a stressor identification - exposure – response – risk assessment framework. The *Integration and Synthesis* section of this opinion follows the *Effects of the Action*, and integrates information we presented in the *Status of Listed Resources* and *Environmental Baseline* sections of this biological opinion with the results of our exposure and response analyses to estimate the probable risks the proposed action poses to endangered and threatened species. Because NMFS has previously concluded that the proposed action is not likely to adversely affect several listed species and areas designated as critical habitat for listed species, these listed resources are not considered in the analyses that follow. Species and critical habitat not likely to be adversely affected by the proposed action are discussed in Section 3.2 of this biological opinion, *Status of Listed Resources Not Considered Further* and in Appendix A.

Since 2010 there has been a requirement for 100% observer coverage in the WCPO purse seine fishery, prior to that it was 20% for United States vessels. Due to the COVID pandemic, observer coverage has been less than 100% in 2020 and 2021. Consequently, the data availability and reliability has changed over the years, and overtime. Information on sea turtle and marine mammal interactions in this fishery has been collected since the mid-1990s; however, the level of importance and detail of those data have evolved since that time. During that initial period, the collection of this information was an ancillary duty of the observers and they received limited training, if any, on these species and no debriefing on the data that were collected. During that period (mid 90s-mid 2000s), most of the interactions were recorded as unidentified turtle or marine mammal; in addition, the program only had disposition codes for green or hawksbill sea turtles, and not leatherback, loggerhead or olive ridley sea turtles. Additional details on the limitation of the data are described in greater detail in the 2006 biological opinion for this fishery (NMFS 2006).

In December of 2005, the WCPFC passed a resolution to reduce the frequency and severity of interactions with sea turtles in the WCPO fisheries. This resolution required countries to collect and provide information on sea turtle interactions and to take measures to reduce entanglement and injury and to increase survival of turtles captured incidental to purse seine activities. Increased and higher quality training and emphasis on the collection of data for sea turtles and marine mammals started to improve due to these requirements. A more experienced cadre of observers also contributed to the collection of higher quality data. There was also a greater effort put into the debriefing process to increase the quality of the data. This has been an ongoing process and the frequency of the debriefing has gradually increased and improved since 2012. In 2012, 12.95% of all trips were debriefed, and in 2018, 98.81% of the trips were debriefed.

Data collection for sharks and the giant manta ray began in 2010 and 2012 respectively. For our analysis we focused on data available from 2008-2018 since the data collection priorities have been the most consistent during this period. Correct identification of giant manta rays is difficult in the field because there are several species that look similar. As described in the *Approach to the Assessment*, the years used to calculate the 5-year running averages (2010-2018) differ slightly than the information presented herein regarding effects from the action which include 2008 and 2009.

5.1 Potential Stressors

Potential stressors associated with the proposed action include:

1. Interaction with fishing gear including capture (entrapment) or entanglement of non-target species, such as listed species or their prey;
2. derelict fishing gear (FADs, that have been lost, abandoned or discarded into marine waters);
3. collisions with vessels;
4. vessel noise; and
5. introduction of vessel wastes, which including, oils, cardboard, air emissions, etc.

We determined that vessel noise, vessel collisions, introduction of discharges and other wastes, gear loss and vessel emissions are stressors that are not likely to adversely affect any species (See Section 3.1; *Listed Resources Not Considered Further* and Appendix A for more details). As a result, in this section we focus primarily on the stressors created by active fishing, which results in capture (entrapment), entanglement, injury, and mortality of listed species.

5.2 Exposure Analyses

With the data currently available, we cannot estimate the total number of endangered or threatened marine animals that are likely to be exposed to the United States WCPO purse seine fishery entirely. We cannot estimate the total number of sea turtles, sharks, giant mantas, and whales that are encircled by the purse seine net because some may swim under before the net is pursed. Additionally, we do not know how many animals may become entangled in FADs since there are currently no data on the number of FADs deployed by each vessel, soak time, or information on the rate that protected species become entangled in FADs. Therefore, our exposure analysis focuses on entrapments that lead to release from the net, entanglement, or landed animals that have been observed and reported. We have limited information on FAD entanglements because we do not know the number of FADs deployed by the United States vessels, the frequency with which they are inspected, and the number that are lost. We have some reported instances of interactions with FADs where an observer noted an animal near a

FAD or entangled at the surface. Since the FADs may contain netting that are hundreds of feet below the surface it is not possible to assess how many animals may be entangled. We have broken the following sections into interactions with purse seine sets including vessel operations, and interactions with the FADs which do not involve the setting of a purse seine net.

5.2.1 Interactions with Purse Seine Fishing Operations

Effects to animals exposed to direct interaction with the purse seine fishing operation range from behavioral modifications for animals that are encircled by the net but released prior to pursing, to injury from entanglement or crushing, to death.

Data collected by observers almost certainly underestimate the actual number of interactions because we cannot account for all individuals that were encircled and escaped before the net was pursed. In some cases, these escapes have been observed for marine mammals and turtles but we cannot be sure that this is observed in every instance. Despite several efforts to assess the significance of unobserved catch and the number of unobserved interactions (for example, Moyes et al. 2006; Murray 2011; Warden and Murray 2011; Gilman et al. 2013), the difference between the number of observed interactions and the actual number of interactions remains unknown. Slipping is the term used to describe animals that escape from the net, or are purposefully released to adjust the amount of catch — particularly when vessels capture more than the capacity of the vessels' hold. The literature primarily focuses on target species, however, slipping is applicable to bycatch species as well, whether intentional or not.

For some sea turtles and marine mammals, their direct exposure ends after they are encircled and are either released by fishermen, or swim free on their own prior to pursing the net and hauling. Generally, it is unclear when an animal is most likely to escape capture. An individual's reponse to these interactions are likely to be behavioral, and result in only short term interruptions to foraging, movement, or socializing. However, some individuals may exhibit signs of stress through alterations in blood chemistry values or metabolic disturbance (Wilson et al. 2014). Marine mammals that break through the net to swim free may experience abrasions and other minor injuries (Wilson et al. 2014). Sea turtles, sharks, and rays become entangled in the net, and these animals are either released by fishermen prior to pursing and hauling, or they are landed prior to release. No marine mammals considered in this biological opinion have been observed entangled in fishing gear.

If animals are not released prior to the pursing and hauling stage they may be retained in the net and landed on the vessel along with the targeted catch. Direct effects to sea turtles, sharks and rays resulting from these interactions often include some element of internal or external wounding, crushing, and hydrostatic effects (i.e. bringing animals up from depth too quickly) (Wilson et al. 2014). Capture in the fishing gear may result in abrasion against nets or against other catch. Poor handling by crew may exacerbate already elevated stress levels or increase the likelihood of delayed mortality post-release (Davis 2002). Studies indicate relatively low mortality rates for sea turtles from these interactions, but these only consider mortalities at time of release and do not account for delayed post-release mortality (Molony 2005; Bourjea et al. 2014). For sharks and rays, the need for sustained swimming can lead to exhaustion. Studies tend to show high mortality rates for sharks and rays due to the varying degrees of physical trauma associated with purse seine gear, and mobility restrictions during the brailing process, among other impacts observed for bycatch (Poisson et al. 2012; Poisson et al. 2014b; Hutchinson et al. 2015; Croll et al. 2016).

As discussed in the *Approach to the Assessment* section, we categorized ESA-listed marine mammal interactions with the purse seine fishery into three general interaction types: (1) direct interaction/capture during an active free set (Free-Set); (2) direct interaction/capture during an active FAD set (FAD-Set); and (3) direct interaction/capture during an active set, set type unknown (Unknown-Set).

Fishery observers document how marine mammals, sea turtles, and elasmobranchs were captured and what the condition of the animal was when released. The observers use the following codes for release conditions: (A0) alive and unknown, (A1) alive and healthy, (A2) alive but injured, (A3) alive but dying, (A4) alive entangled (but released disentangled), (D) dead, and (U) unknown. For animals with condition code alive and unknown (A0), we used the proportions calculated for the known release conditions of live animals (i.e. release conditions A1, A2, and A3) within each interaction type to assign condition codes to those animals. Similarly, for animals with condition code U, that is, is was unknown if the animal was released alive or dead, and if alive what its condition was, we used the proportions calculated for the known release conditions (i.e. release conditions A1, A2, A3 and D) within each interaction category to assign condition codes to each interaction.

The release condition codes (i.e. A0, A1, A2, A3, A4, D and U) were only used sporadically prior to about 2015 for elasmobranchs. Therefore, if the observer records did not record release conditions, the codes were translated to the release condition codes used for marine mammals and sea turtles as follows: DPA= alive and healthy (A0); DPD, RCC, ROR, RWW, and DFR = dead (D); and DPU, DUS, and UUU = unknown (U). Like marine mammals and sea turtles, we used the proportions calculated for the known release conditions (i.e. A1, A2, A3 and D) within each interaction category to assign condition codes to those sharks and rays.

In the following sections we describe the number of individuals exposed to the stressors from a vessel searching and conducting a set using the available observer data.

5.2.1.1 Marine Mammals

Since 2008 there have been 86 observed records of ESA-listed and unidentified marine mammals captured by the United States WCPO purse seine fishery. Cetaceans are typically encircled in purse seines due to the tendency of tunas to school around them. Some fisherman will seek out whales and set their net around them, using the whale as a FAD. This intentional setting on whales is illegal under the MMPA and ESA and is required to be reported to the Office of NOAA Law Enforcement for investigation when observed. In the last six years 13 different vessels have been charged with intentionally setting on marine mammals and two of the vessels were charged two or more times (NOAA OLE pers. comm. 2019). In the observer database, the type of set is noted and these types include unassociated, feeding on baitfish, drifting log, debris or dead animal, drifting raft, FAD or payao, anchored raft, FAD or payao, live whale, live whale shark, other, and no tuna associated. Of these interaction types, live whale implies that the set was made on a fish school identified by the presence of a live whale. In our analysis of ESA-listed whale interactions with the United States WCPO purse seine fishery, some of the interactions were noted to be live whale sets by the observer, which suggests unlawful setting on whales but in many cases the unlawfulness of the set cannot be determined precisely as many of the observer records do not contain enough detail to determine if the set was deliberate, or a whale was incidentally captured during the set, causing the observer to record it as a whale set. Although some of these sets were likely unlawful, currently we do not have the data to differentiate between intentional and incidental whale sets. Hence, in order to not underestimate the number of anticipated interactions, we used interactions that occurred on sets identified to have occurred on live whales in our analysis for both the anticipated future number of interactions and anticipated impacts to the species. One exception to this is for blue whales, as discussed in Appendix A we do not anticipate any incidental captures of blue whales and consider the single record in the observer database of a blue whale capture to have likely occurred on an illegal whale set.

The WCPFC also adopted a measure to prohibit all nations from intentionally setting on whales in 2011 (CMM 2011-03). Even when vessels are not intentionally setting on whales, they can still be caught inadvertently. The CMM requires that "in the event that a cetacean is unintentionally encircled in the purse seine the master of the vessel shall:

a. Ensure that all reasonable steps are taken to ensure its safe release. This shall include stopping the net roll and not recommencing fishing operation until the animal has been released and is no longer at risk of recapture; and
b. reporting the incident to the relevant authority of the flag State, including details of the species (if known) and number of individuals, location and date of such encirclement, steps taken to ensure safe release, and an assessment of the life status of the animal on release (including, if possible, whether the animal was released alive but subsequently died)" (CMM 2011-03).

For purposes of this exposure analysis, we reviewed all observer reports of marine mammal interactions, and all marine mammal reports sent in by vessel operators as required under NMFS' Marine Mammal Authorization Program. In many cases observers are unable to identify marine mammals to species, due to distance to the animal, restricted visibility, and similar body shapes that lack strong identifying characteristic of many baleen whales. In all cases where observers could not identify a whale to species they use higher taxonomic categories. We reviewed all of these 'unidentified categories' in the observer data and identified all that could potentially contain ESA-listed marine mammals. The unidentified species were proportioned across the observed and identified species as described in the *Approach to the Assessment* section.

Most observed marine mammal interactions occurred on FAD-sets (70%), followed by free sets and interactions that occurred on sets when the set type was unknown (15% each) (Table 18). There was only one sei whale recorded as condition alive entangled (A4) and examination of observer records suggest this whale was released alive and healthy after breaking through the net. As no damage or injury was observed to the whale, IFD and PRD therefore reassigned this sei whale to condition code alive and healthy (A1) for their respective analysis. No marine mammals were described as dead (A4) or released alive but unlikely to survive (A3) after any type of interactions for ESA-listed marine mammals (Table 19).

In the species-specific sections that follow, we break out interactions by interaction type and release condition for each species (Table 19). To summarize, for all marine mammal species, 1% of free-set interactions resulted in injury (A2) and 10% of FAD set interactions resulted in injury (Table 20). There were no "alive but unlikely to survive" (A3) or "dead" (A4) conditions described for any marine mammal interactions. Fifty-nine percent of all interactions occurred during FAD sets with a release condition of "alive and healthy" and 14% of interactions occurred during free sets with a release condition of "alive and healthy." Most cetaceans (84%) were coded as released “alive and healthy” (A1) after their interaction with the United States WCPO purse seine fishery (Table 20).

Fin whales represent approximately 5% of the 86 observed interactions that had potential to be an ESA-listed species. Sperm whales represent approximately 64% of the interactions, and sei whales represented approximately 52% of the interactions. Unidentified marine mammals represented 37% of the total and this includes the codes marine mammal unidentified, unidentified baleen whale, and unidentified toothed whales (Table 18).

Table 18. Number of ESA-listed whales and unidentified marine mammals that have been observed captured and the total estimated for the United States WCPO purse seine fishery since 2008 (numbers in parentheses are the 95% confidence intervals of the estimated totals). The type of set is described also. All the observed unidentified whales were proportioned out by identified species and included in the total.

	Sei Whale	**Sperm Whale**	**Fin Whale**	**Unidentified[1]**	**Total**
Observed	45	5	4	32	86
Total estimated	96 (87, 140)	6 (4, 13)	14 (9, 19)	NA	116 (100, 172)
% of Total observed	52	6	5	37	-
Interaction type					
Free-Set	4	1	0	8	13
FAD Set	30	4	4	22	60
Unknown-Set	11	0	0	2	13

[1] Unidentified includes the following categories: marine mammal unidentified, unidentified baleen whale, and unidentified toothed whale codes.

Table 19. Number of marine mammal interactions in the United States WCPO purse seine fishery from 2008-2018 by the interaction type and release condition. See the sections that follow for species-specific information.

	Release Condition		
Free-Set	12	1	13
FAD Set	51	9	60
Unknown-Set	9	4	13
Grand Total	72	14	86

Table 20. Proportions of marine mammal interactions by release condition and set type from 2008-2018.

	Release Condition		
Free-Set	0.14	0.01	0.15
FAD Set	0.59	0.10	0.70
FAD Interaction	0.00	0.00	0.00
Unknown-Set	0.11	0.05	0.15
Grand Total	0.84	0.16	1

Sei Whale

Since 2008, there have been 45 observed interactions with sei whales. In all but one year, there were 10 or fewer interactions; in 2014 there were 25 interactions accounting for 34% of all interactions (Table 21). Most sei whales were uninjured as a result of their interaction (85%; Table 22) while 15% were considered injured (Table 22). After proportioning the unidentified marine mammals and accounting for unobserved sets the estimated total number of interactions with sei whales from 2008-2018 is 181 (95th percentile: 214).

Table 21. Number of sei whales caught by year with the release condition code recorded by the observer. This includes the number observed and the total estimated number caught considering unobserved sets and unidentified marine mammals.

	Condition		
Year	**A1 Alive and healthy**	**A2 Alive but injured**	**A4 Alive and entangled**
2008	1	0	0
2009	1	0	0
2010	4	4	0
2011	3	0	0
2012	0	0	0
2013	3	2	0
2014	12	0	1
2015	5	2	0
2016	1	0	0
2017	2	2	0
2018	1	1	0
Total	33	11	1

Table 22. Total number of observed sei whales from 2008-2018 partitioned by the set type and condition code upon release.

	Condition	
	A1 **Alive and healthy**	**A2** **Alive but injured**
Free-Set	3	1
FAD Set	23	7
Unknown Set Type	8	3
Total	34	11

Demographic Patterns of Exposure

Two interactions documented by the observers described the whales as juveniles but there were not any other details to describe why they considered them juveniles. There was no mention in the observer's data describing interactions with a mother and calf pair or anything to give us an indication of the sex of the animals.

Spatial and Temporal Patterns of Exposure

We mapped all of the observed interactions from 2008-2018 to see if there were any visible patterns as they occurred throughout the fishery's range. Additionally, due to the spatial remoteness of the points, a heatmap was not viable.

We looked at the number of interactions by year (Figure 35) and by month (Figure 36). In most years, (73%) there were four or fewer interactions. In 2014, the highest number of interactions occurred representing 29% of the interactions, these interactions occurred during 7 sets involving one to four whales on each set. The interactions that occur in November and December occur between 154°E to 179°E and do not extend further east of 179°E (longitude) during that time of the year. Overall, the interactions span across about 2,500 nm East to West and 800 nm North to South. Th northern-most interaction occurred at about 5°N, the southern-most at about 9°S, the western-most at about 149°E and the eastern-most at about 170°W. The interactions are distributed relatively continuously within that area with no discernable pattern.

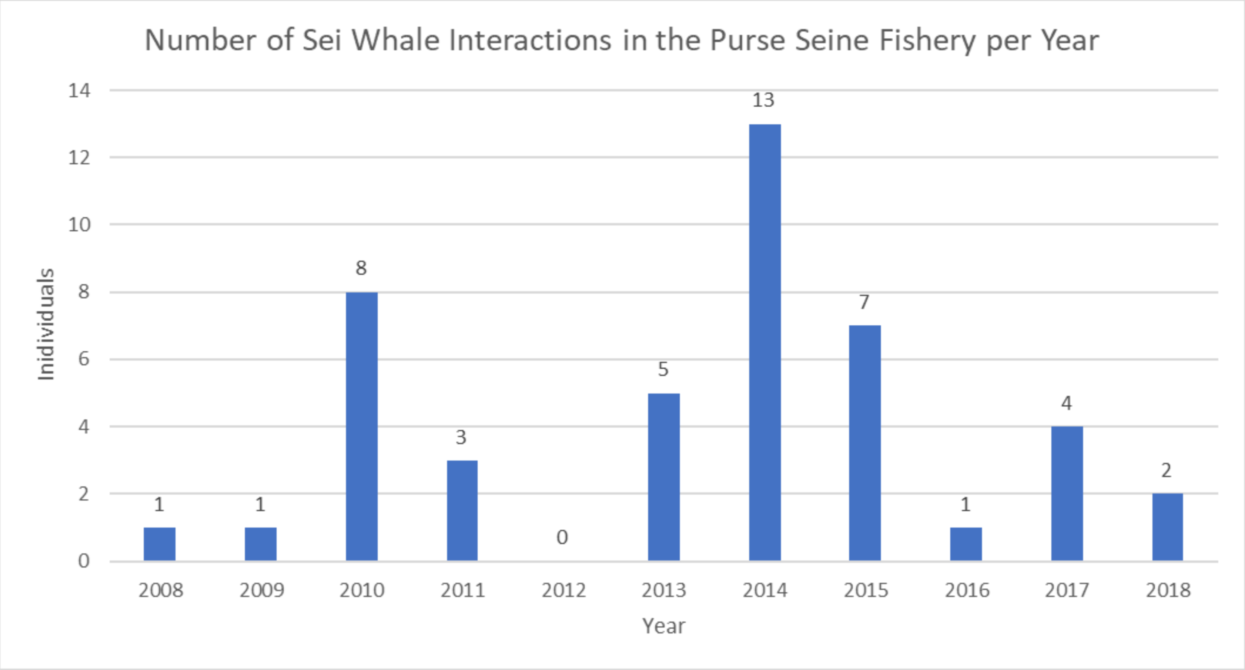


Figure 35. Number of observed sei whale interactions observed by year in the United States WCPO purse seine fishery.

We also looked at the interactions by month and found that the most interactions occurred in the months of February, September, and December (56%) with few interactions from March through August (Figure 36). The sets were made between 05:16 and 17:36, indicating that there would have still been daylight at the time of the set for all except the set made at 05:16, which would have been just prior to sunrise.

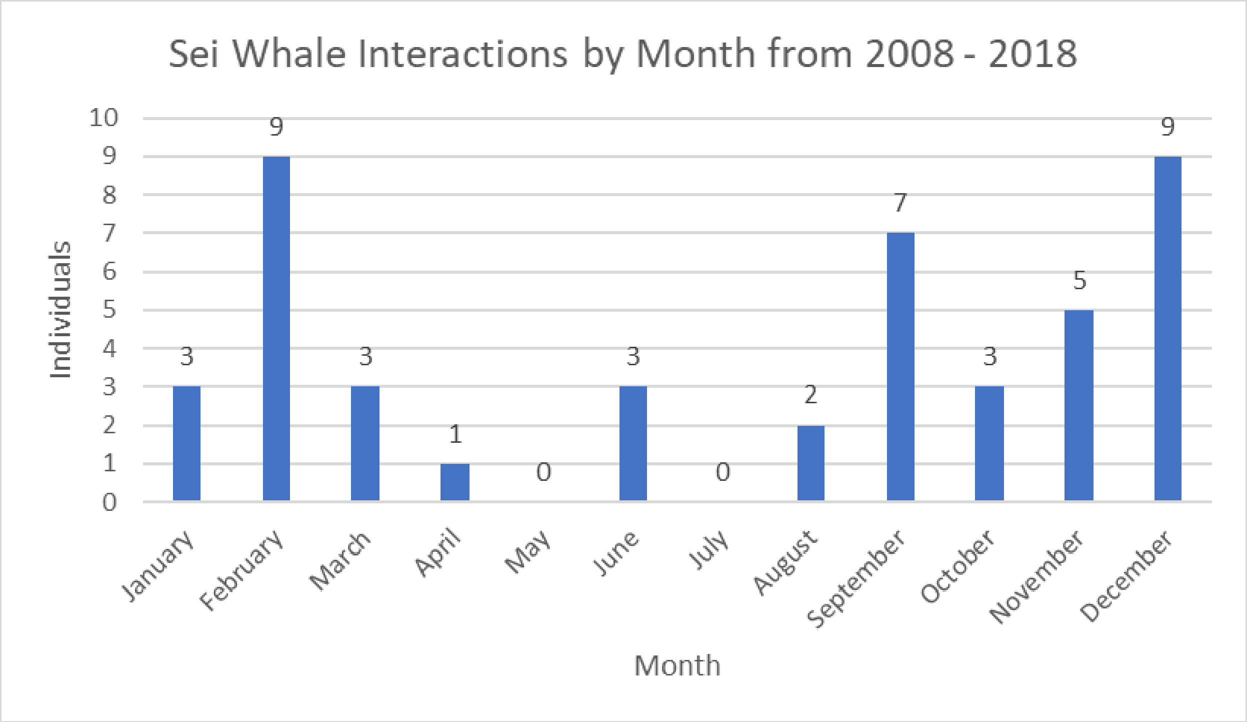


Figure 36. Graph showing the number of observed sei whale interactions from 2008-2018, by month.

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interactions using Bayesian statistical inference techniques as described in the *Approach to the Assessment* section of this biological opinion (NMFS 2019c). The predictions were based on the historical interactions recorded by observers, which includes unidentified marine mammals and are based on a future effort estimate of 3,100 sets, with 1,581 of those being FAD sets. We also calculated a total estimate to account for unobserved sets. As noted in Section 2.2.2.6, we found the 95th percentile from the Bayesian inference model generally underestimated the highest numbers of anticipated captures and we therefore used a nonparametric bootstrap analysis to estimate the 95th percentile of anticipated captures. Table 23 contains the resulting predictions of future total annual interactions (mean and 95th percentile) and the maximum 5-year running average and sum for sei whales interacting with the United States WCPO purse seine fishery.

Table 23. Mean and 95th percentile of sei whale posterior estimated exposures in terms of anticipated annual interactions with fishing gear and the maximum 5-year running average and running sum.

Bayesian Inference Model		
	Annual Mean	**Annual 95th percentile**
Predicted Exposures	3	7
Maximum Running Average and Sum		
	5-yr Running Average	**5-yr Running Sum**
Predicted Exposures	6.0	30

Sperm Whale

Since 2008, four sets that involved interactions with sperm whales have been observed, interacting with five sperm whales. After proportioning the unidentified marine mammals and accounting for unobserved sets the estimated total number of interactions with sperm whales from 2008-2018 is 6 (95th percentile: 13) (Table 18). There are no observer notes describing the interactions. Three are coded with an A1 release code, which is considered alive and healthy. Three of the four interactions occurred on a FAD set and one occurred on a free school set.

Demographic Patterns of Exposure

There is no information from the observer to determine the size or sex of the animal.

Spatial and Temporal Patterns of Exposure

Due to the limited number of observed interactions with sperm whales there is no discernable pattern of exposure to the United States WCPO purse seine fishery when displayed using ArcGIS All four observed interactions were caught in different months with no seasonal patterns. The sets were made between 12:50 and 17:36, indicating that there would have still been daylight at the time of each set. Spatial remoteness between interaction locations did not allow for heatmaps to be produced.

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interactions using Bayesian statistical inference techniques as described in the *Approach to the Assessment* section of this biological opinion (NMFS 2019c). The predictions were based on the historical interactions recorded by observers, which includes unidentified marine mammals and are based on a future effort estimate of 3,100 sets, with 1,581 of those being FAD sets. We also calculated a total estimate to account for unobserved sets. As noted in Section 2.2.2.6, we found the 95th percentile from the Bayesian inference model generally underestimated the highest numbers of anticipated captures and we therefore used a nonparametric bootstrap analysis to estimate the 95th percentile of anticipated captures. Table 24 contains the resulting predictions of future total annual interactions (mean and 95th percentile) and the maximum 5-year running average and sum for sperm whales interacting with the United States WCPO purse seine fishery.

Table 24. Mean and 95th percentile of sperm whale posterior estimated exposures in terms of anticipated annual interactions with fishing gear and the maximum 5-year running average and running sum.

Bayesian Inference Model		
	Annual Mean	**Annual 95th percentile**
Predicted Exposures	0	1
Maximum Running Average and Sum		
	5-yr Running Average	**5-yr Running Sum**
Predicted Exposures	0.67	4

Fin Whale

Since 2008 there have been four observed interactions with fin whales which occurred in 2017 and 2018, no fin whales had been recorded prior to 2017. The four whales were captured on 4 sets, all of which were FAD sets. Three of the four whales were released alive and healthy. The condition of the sixth was unknown, this whale was trapped in the net during the haul so it may be inappropriate to use the proportioning methods to assign a condition code to the unknown whale as we have for other whales and we consider this whale to have been injured or in distress (code A2) upon release. After proportioning the unidentified marine mammals and accounting for unobserved sets the estimated total number of interactions with fin whales from 2008-2018 is 14 (95th percentile: 18).

Demographic Patterns of Exposure

There is no information from the observer description to determine the size or sex of the whales.

Spatial and Temporal Patterns of Exposure

Due to the limited number of observed interactions with fin whales there is no discernable pattern of exposure to the United States WCPO purse seine fishery when mapped geographically with ArcGIS. However, all four interactions occurred north of the Solomon Islands. Two of the three sets occurred in December and one was in March. The sets were made between 12:50 and 17:36, indicating that there would have still been daylight at the time of each set.

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interactions using Bayesian statistical inference techniques as described in the *Approach to the Assessment* section of this biological opinion

(NMFS 2019c). The predictions were based on the historical interactions recorded by observers, which includes unidentified marine mammals and are based on a future effort estimate of 3,100 sets, with 1,581 of those being FAD sets. We also calculated a total estimate to account for unobserved sets. As noted in Section 2.2.2.6, we found the 95th percentile from the Bayesian inference model generally underestimated the highest numbers of anticipated captures and we therefore used a nonparametric bootstrap analysis to estimate the 95th percentile of anticipated captures. Table 25 contains the resulting predictions of future total annual interactions (mean and 95th percentile) and the maximum 5-year running average and sum for fin whales interacting with the United States WCPO purse seine fishery.

Table 25. Mean and 95th percentile of fin whale posterior estimated exposures in terms of anticipated annual interactions with fishing gear and the maximum 5-year running average and running sum.

Bayesian Inference Model		
	Annual Mean	**Annual 95th percentile**
Predicted Exposures	0	1
Maximum Running Average and Sum		
	5-yr Running Average	**5-yr Running Sum**
Predicted Exposures	0.75	4

5.2.1.2 Sea Turtles

Since 2008, there have been 297 records of sea turtles being captured by the United States WCPO purse seine fishery (Table 26). Green sea turtles represented 33% of these interactions, loggerhead sea turtles 23%, hawksbills sea turtles 20%, olive ridley sea turtles represented 19%, leatherback sea turtles represented 3% of the turtle interactions, and unidentified (hard shell) sea turtles 3% of the total observed interactions. NMFS developed predictions of annual bycatch rates and estimates of future interactions using Bayesian statistical inference techniques as described in the *Approach to the Assessment* section of this biological opinion. 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 at 508 (95% CI: [444, 606]) sea turtles (Table 26).

Some studies have indicated that the probability of sea turtles interacting with a purse seine set may be related to whether the set is made on a FAD (Molony 2005). Therefore, we further divide fishery interaction types by whether the set was on a FAD (FAD set), not on a FAD (free set), or if this information was not known (unknown set type). Since 2008, the United States WCPO purse seine fishery has captured 273 sea turtles during an active set of the fishery (Table 26). 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 26).

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There was only one sea turtle recorded as condition A4 and examination of observer records suggest this turtle was released alive and healthy after being disentangled from the net. We therefore reassigned this turtle to condition code A1. Table 27 summarizes the numbers of observed sea turtle interactions for each release condition. In the species-specific sections that follow, we break out interactions by interaction type and release condition for each species (see Table 29, Table 33, Table 36, Table 40, and Table 44). To summarize for all sea turtle species, 14% of free set interactions resulted in injury (A2) while 26% of FAD set interactions resulted in injury (Table 28). For dying and dead sea turtles (A3 or D), 5% of free set interactions and 10% of FAD set interactions resulted in those condition codes. Looking at all injuries (i.e. all sea turtles assigned a release condition code of A2), 46% of them occurred on free set interactions while 52% occurred on FAD set interactions. Similarly, for all dead or dying turtles (A3 or D), 44% occurred on free set Interactions and 56% occurred on FAD set interactions. For all fishery set interaction types combined, condition code A1 accounted for 76% of the interactions, A2 accounted for 18% of the interactions, and A3 and D combined accounted for 7% of the interactions (Table 28).

There are regulations in place for the United States WCPO purse seine fishery for releasing captured sea turtles from the net and for appropriate handling if landed to improve the survival of sea turtles captured by this fishery. According to the WCPFC Conservation and Management Measure 2018-04 and 50 CFR 300.223(f), fishermen are required to:

- Implement, as appropriate, the FAO Guidelines to Reduce Sea Turtle Mortality in Fishing Operations and ensure the safe handling of all captured sea turtles;
- Bring aboard, if practicable, any captured hard-shell sea turtle that is comatose or inactive as soon as possible to foster its recovery before returning it to the water;
- Avoid encirclement of sea turtles; if a sea turtle is encircled, take practicable measures to safely release it;
- To the extent practicable, release all sea turtles observed entangled in FADs;
- If a sea turtle is entangled in the net, stop net roll as soon as the turtle comes out of the water; disentangle the turtle without injury before resuming the net roll; and to the extent practicable, assist the recovery of the turtle before returning it to the water;
- Carry and employ dip nets when appropriate to handle turtles;
- Record all incidents involving sea turtles; and
- Provide the results of the reporting to the WCPFC.

Table 26. 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 95% confidence intervals. Reported values for interaction types are the numbers of observed sea turtles.

	Green*	**Leatherback**	**Loggerhead**	**Olive Ridley**	**Hawksbill**	**Unidentified**	**Total**
Observed	97	8	68	57	59	8	297
Total estimated	173 (152, 200)	10 (8, 20)	121 (107, 144)	105 (90, 118)	99 (87, 124)	-	508 (444, 606)
Interaction Type							
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 27. Number of sea turtles captured by the United States WCPO Purse Seine fishery during active fishery sets between 2008 and 2018 by the interaction type and release condition. See the sections that follow for species-specific information.

	Release Condition				
Interaction Category	**A1** **Alive and healthy**	**A2** **Alive but injured**	**A3** **Alive but unlikely to live**	**D** **Dead**	**Grand Total**
Free-Set	126	22	5	4	157
FAD Set	61	25	8	2	96
Unknown Set Type	19	1	0	0	20
Grand Total	206	48	13	6	273

Table 28. Proportions of sea turtles captured by the United States WCPO Purse Seine fishery during active fishery sets between 2008 and 2018 by the interaction type and release condition. See the sections that follow for species-specific information.

	Release Condition				
Interaction Category	**A1** **Alive and healthy**	**A2** **Alive but injured**	**A3** **Alive but unlikely to live**	**D** **Dead**	**Grand Total**
Free-Set	0.47	0.08	0.02	0.01	0.58
FAD Set	0.22	0.09	0.03	0.01	0.35
Unknown Set Type	0.07	0.00	0.00	0.00	0.07
Grand Total	0.76	0.18	0.05	0.02	1.00

Direct interactions with purse seine gear primarily affects sea turtles by being encircled and possibly retained in the net through pursing and hauling, which can lead to entanglement or being landed on the vessel along with the targeted catch. For some sea turtles, their direct interaction with the fishery ends after they are encircled by the net but prior to pursing and hauling when they are either released by fishermen, or swim free on their own. Direct impacts to sea turtles that are not released from the net prior to pursing and hauling can include some element of internal or external wounding and crushing. Capture in the fishing gear can result in abrasion against nets or against other catch. Furthermore, crew mishandling of landed, live animals once onboard may exacerbate already elevated stress levels or increase the likelihood of delayed mortality post-release (Davis 2002).

Hawksbill Sea Turtle

From 2008 to 2018, 59 hawksbill sea turtles were observed interacting with the United States WCPO purse seine fishery (Table 29). After proportioning the unidentified sea turtles and accounting for unobserved sets the estimated total number of interactions with hawksbill sea turtles from 2008-2018 is 99 (95% CI: [87, 124]). They may also be attracted to FADs as sources of prey from the colonizing epifauna and shelter, although higher numbers of hawksbill sea turtles were observed captured by the fishery during free-sets compared to FAD-sets (59% of interactions on fishery sets with known set type occurred during free sets (27 out of 46 interactions; Table 29)

Table 29 describes the number of hawksbill sea turtles captured in the United States WCPO purse seine fishery between 2008 and 2018 during active sets by their interaction category and release condition. Fifty-one hawksbill sea turtles have been observed captured by this fishery during sets. Of those, 76% were considered by the observers to be uninjured at the time of release, 8% were released injured and 16% were released either dying or dead (Table 30).

Table 29. Number of hawksbill sea turtles captured by the United States WCPO Purse Seine fishery during active fishery sets between 2008 and 2018 by the interaction type and release condition.

	Release Condition				
Interaction Category	**A1 Alive and healthy**	**A2 Alive but injured**	**A3 Alive but unlikely to live**	**D Dead**	**Grand Total**
Free-Set	22	2	3	0	27
FAD Set	12	2	4	1	19
Unknown Set Type	5	0	0	0	5
Grand Total	39	4	7	1	51

Table 30. Proportions of hawksbill sea turtles captured by the United States WCPO Purse Seine fishery during active fishery sets between 2008 and 2018 by the interaction type and release condition.

	Release Condition				
Interaction Category	**A1** **Alive and healthy**	**A2** **Alive but injured**	**A3** **Alive but unlikely to live**	**D** **Dead**	**Grand Total**
Free-Set	0.431	0.039	0.059	-	0.529
FAD Set	0.235	0.039	0.078	0.020	0.373
Unknown Set Type	0.098	-	-	-	0.098
Grand Total	0.765	0.078	0.137	0.020	1.00

Demographic Patterns of Exposure

Hawksbill sea turtles are globally listed under the ESA as endangered. There are low levels of nesting and nearshore juvenile foraging reported from nations throughout the area of the fishery (see Section 3.2.4 for specific information) and it is anticipated that the hawksbill sea turtles captured by the United States WCPO purse seine fishery are from a mixture of these sources. 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; 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 genetic mixed stock analyses and limits robust assessments of nesting population contributions (Gaos et al. 2016). A recent study using nDNA indicates that there are at least 3 hawksbill sea turtle populations globally, East Pacific, West Pacific and Atlantic (Banerjee et al. 2019). As noted in the *Status of Listed Resources* section, given that population structure is yet well defined for hawksbill sea turtles, we assume that hawksbill sea turtles interacting with the purse seine fishery are all from the West Pacific in this assessment.

Hawksbill sea turtle life history is characterized by early development in the pelagic zone followed by later development in nearshore habitats, especially coral reefs. Adults will make trans-oceanic migrations from foraging grounds to nesting beaches. As such, small, pelagic-stage juveniles and breeding adults are the most likely life stages to be encountered by the United States WCPO purse seine fishery. Table 31 lists the size and estimated ages of the 24 hawksbill sea turtles captured by the fishery between 2008 and 2018.

Chaloupka and Limpus (1997) indicate that nesting hawksbill sea turtle size in northern Australia ranges from 76 to 93 cm CCL. Witzell (1985) reports a range of 68 to 93 cm SCL for the Solomon Islands and 71.6 to 82.7 cm SCL for Australia for nesting hawksbill sea turtles. Seitz et al. (2012) reports a range of 73 to 89 cm SCL for Hawaii. We therefore consider 68 cm SCL as the minimum size of adult hawksbill sea turtles. Hence, the largest captured hawksbill sea turtle (60 cm CL) may have been a sub-adult, and the remaining individuals were juveniles.

Table 31 also shows the estimated age based on skeletochronology aging data presented by Snover et al. (2013). We estimate these turtles ranged between 1-year old for the individual at 15 cm, upwards of 9 years old for the turtle at 60 cm. There is not enough information available to identify hawksbill sea turtles captured in this fishery to sex, as it is not possible to determine the sex of juvenile sea turtles from external examination, and there are no records of sex in adults based on tail length.

Table 31. Straight carapace length (SCL) measurements and estimated age for the hawksbill sea turtles that were measured by observers during interactions with the United States Purse Seine Fishery between 2008 and 2018. Age estimates are based on the growth curve of Snover et al. (2013).

SCL (cm)	Estimated Age (yrs)
15	1
29	2.6
29	2.6
29	2.6
29	2.6
30	2.7
30	2.7
30	2.7
35	3.4
35	3.5
35	3.5
35	3.5
36	3.4
36	3.4
37	3.8
38	4
38	4
38	4
39	4.3
41	4.6
42	4.7
46	5.7
49	6.3
60	9.2

Table 31 also shows the estimated age based on skeletochronology aging data presented by Snover et al. (2013). We estimate these turtles ranged between 1-year old for the individual at 15 cm, upwards of 9 years old for the turtle at 60 cm. There is not enough information available to identify hawksbill sea turtles captured in this fishery to sex, as it is not possible to determine the sex of juvenile sea turtles from external examination, and there are no records of sex in adults based on tail length.

Spatial and Temporal Patterns of Exposure

Hawksbill sea turtle interactions with the United States WCPO purse seine fishery are distributed throughout the area of activity for this fishery. There is a cluster of 12 interactions occurring north of Papua New Guinea, approximately 200 km north of Manus Island. Most of those 12 interactions occurred between 2009 and 2012, with one additional interaction occurring in 2018.

The highest numbers of hawksbill sea turtle interactions occurred in 2011, with 10 interactions (Figure 37). Annual interactions ranged from zero to 10. Interactions have occurred in all months except July, with the highest numbers of interactions in March and May (Figure 38).

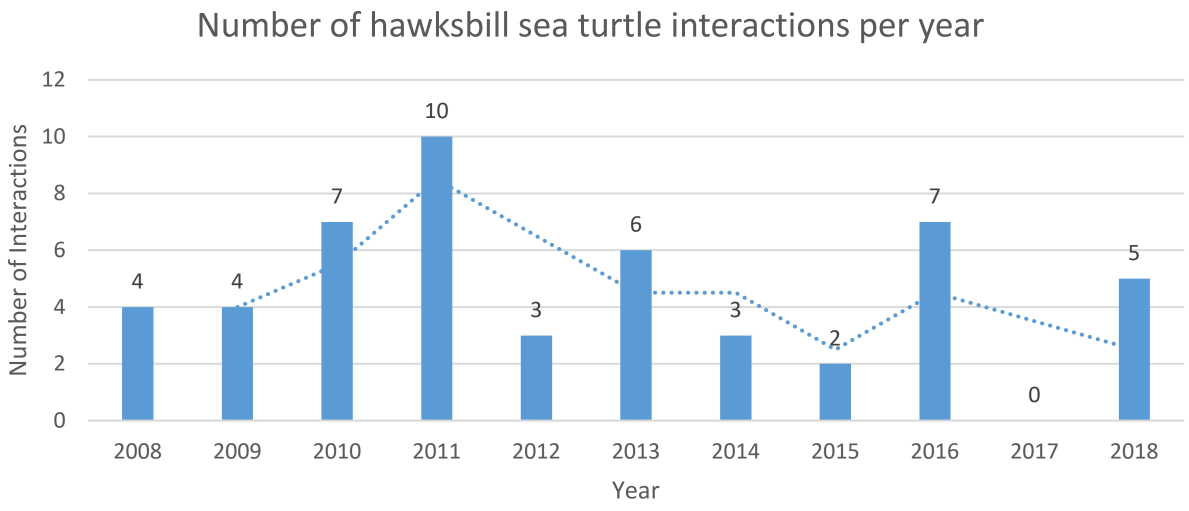


Figure 37. Hawksbill sea turtle interactions by year in the United States WCPO purse seine fishery between 2008 through 2018 with 2-year moving average (blue dashed line).

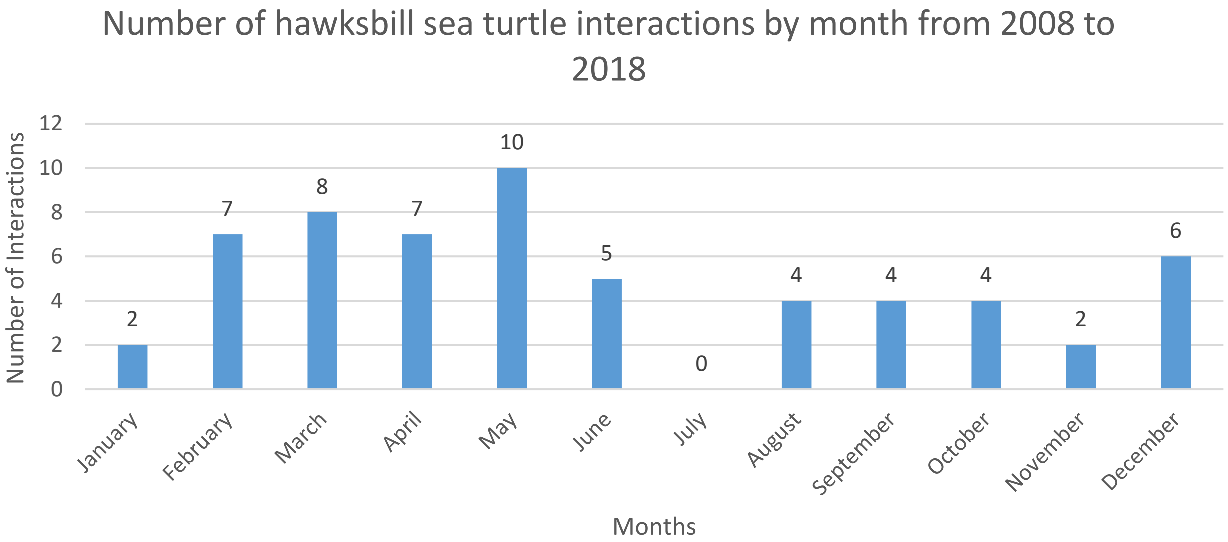


Figure 38. Hawksbill sea turtle interactions by month in the United States WCPO purse seine fishery between 2008 through 2018.

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interactions using Bayesian statistical inference techniques as described in the *Approach to the Assessment* section of this biological opinion (Martin et al. 2015). The predictions were based on the recorded bycatch of hawksbill sea turtles in the United States WCPO purse seine fishery and the assumption of 3,100 annual sets with 1,581 of those being FAD sets. As noted in Section 2.2.2.6, we found the 95th percentile from the Bayesian inference model generally underestimated the highest numbers of anticipated captures and we therefore used a nonparametric bootstrap analysis to estimate the 95th percentile of anticipated captures. Table 32 contains the resulting predictions of future total annual interactions (mean and 95th percentile) and the maximum 5-year running average and sum for hawksbill sea turtles interacting with the United States WCPO purse seine fishery. These estimates are inclusive of FAD interactions (see the *Entanglement in FADS* section).

Table 32. Mean and 95th percentile of hawksbill sea turtle posterior estimated exposures in terms of anticipated annual interactions with fishing gear (inclusive of FAD interactions, see Section 5.2.2.2, *Entanglement in FADs – Sea Turtles*) and the maximum 5-year running average and running sum.

Bayesian Inference Model		
	Annual Mean	**Annual 95th percentile**
Predicted Exposures	4	8
Maximum Running Average and Sum		
	5-yr Running Average	**5-yr Running Sum**
Predicted Exposures	4.5	23

Leatherback Sea Turtle

Leatherback sea turtles are the least common species of sea turtle to interact with the United States WCPO purse seine fishery with eight observed interactions between 2008 and 2018. After proportioning the unidentified sea turtles and accounting for unobserved sets the estimated total number of interactions with leatherback sea turtles from 2008-2018 is 10 (95% CI: [8, 20]). As their preferred prey are mid-column gelatinous species such as sea jellies, siphonophores, and tunicates, leatherback sea turtles are not known to associate with tuna schools or be attracted to FADs, although three out of the eight observed captures were on FAD associated sets. All eight of the observed leatherback sea turtle captures were landed.

Table 33 describes the number of leatherback sea turtles captured in the United States WCPO purse seine fishery between 2008 and 2018 by their interaction category and release condition. Eight leatherback sea turtles have been observed captured in this fishery, with five occurring on free-sets and 3 occurring on FAD sets. Seven of the eight leatherback sea turtles (88%) were released uninjured while one was released injured (Table 33, Table 34).

Table 33. Number of leatherback sea turtles captured in the United States WCPO Purse Seine fishery between 2008 and 2018 by the interaction type and release condition.

	Release Condition				
Interaction Category	**A1 Alive and healthy**	**A2 Alive but injured**	**A3 Alive but unlikely to live**	**D Dead**	**Grand Total**
Free-Set	5	0	0	0	5
FAD Set	2	1	0	0	3
Unknown Set Type	0	0	0	0	0
Grand Total	7	1	0	0	8

Table 34. Proportions of leatherback sea turtles captured by or interacting with the United States WCPO Purse Seine fishery between 2008 and 2018 by the interaction type and release condition.

	Release Condition				
Interaction Category	A1 Alive and healthy	A2 Alive but injured	A3 Alive but unlikely to live	D	Grand Total
Free-Set	0.63	-	-	-	0.63
FAD Set	0.25	0.13	-	-	0.38
Unknown Set Type	-	-	-	-	-
Grand Total	0.88	0.13	-	-	1.00

Demographic Patterns of Exposure

As noted above, all eight of the observed leatherback sea turtles captured by the United States WCPO purse seine fishery were landed. However, the observer data only contained size information estimates for three individuals. The reported sizes were 100 and 84 cm SCL and 105 cm straight carapace width. Using data on nesting female leatherback sea turtle carapace lengths and widths from Stewart et al. (2007), we estimate the 105 cm straight carapace width turtle to be at least 143 cm SCL.

For the purpose of this assessment, leatherback sea turtles greater than 124 cm SCL are considered adults. Average minimum nesting length worldwide is 124 cm SCL, and 147 cm SCL is the best estimate of the average size of nesting adult leatherback sea turtles in the Pacific (Jones et al. 2011). The best estimates provided by observers suggest that the leatherback sea turtles that interact with this fishery with are a combination of juveniles and adults.

As discussed in the *Status of Listed Resource* section, we anticipate that only leatherback sea turtles from the Western Pacific population will interact with the United States WCPO purse seine fishery because: (1) there are very few captures of leatherback sea turtles in the United States WCPO purse seine fishery and all of the observed interactions occurred west of 169º W (see below in the *Spatial and Temporal Patterns of Exposure* section); 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* (Willis-Norton et al. 2015; NMFS 2017a) and (2) all of the turtles that have been captured by the ASLL fishery (overlapping *Action Areas*) have been from the western Pacific population.

The sex of leatherback sea turtles that this fishery interacts with is unknown because sex cannot be determined from external evaluation in juveniles, and sex (based on tail length) was not recorded for the single adult-sized animal. However, between 1998 and 2005 the Hawaii-based deep-set longline and ASLL fisheries were responsible for the death of ten smaller leatherback sea turtles (Siders et al. 2018). These turtles were returned to NMFS where they were necropsied. Seven of the leatherback sea turtles were from the Hawaii-based deep-set longline fishery and three from the ASLL fishery. Two of these turtles (20%) were identified as males during necropsy and seven were identified as females (80%), and one turtle could not be identified to sex (Siders et al. 2018). Similarly, Benson et al. (2011) tagged turtles in the California foraging grounds at a ratio of 3:1 females to males. The IUCN recently used a sex ratio of 3:1 female to male for their assessment to account for adult males, which is based on the work of Tapilatu and Tiwari (2007 as cited in Tiwari et al. 2013).

Spatial and Temporal Patterns of Exposure

Interactions of leatherback sea turtles with the United States WCPO purse seine fishery occurred primarily south of the equator, as far as 7.7° south. One leatherback sea turtle was captured north of the equator at 1.15°N. From east to west, captures of leatherback sea turtles span the area of highest effort for the United States WCPO purse seine fishery with captures ranging from 153.7°E to 169.2°W. Captures of leatherback sea turtles in the United States WCPO purse seine fishery occurred year-round (Figure 39) with 38% (3 out of 8) occurring in August. Captures occurred at a rate of zero (*n* = 4 years), one (*n* = 6 years) or two (*n* = 1 year) leatherback sea turtles per year with no discernable trend in numbers.

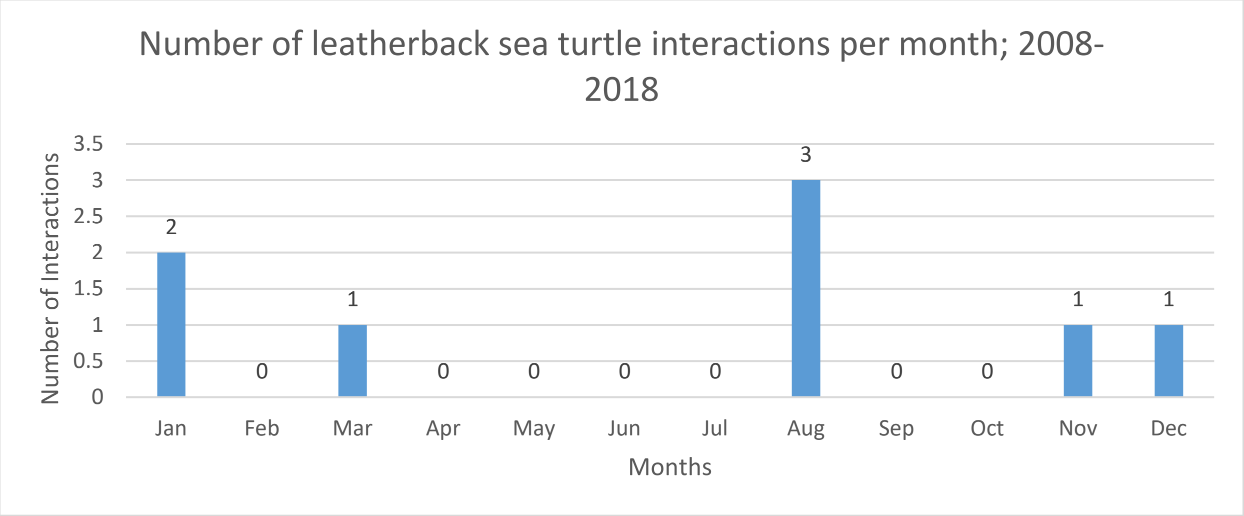


Figure 39. Numbers of observed leatherback sea turtle captures by month in the United States WCPO purse seine fishery from 2008 through 2018.

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interactions using Bayesian statistical inference techniques as described in the *Approach to the Assessment* section of this biological opinion (NMFS 2019c). The predictions were based on the recorded bycatch of leatherback sea turtles in the United States WCPO purse seine fishery and on the assumption of 3,100 annual sets with 1,581 of those being FAD sets. As noted in Section 2.2.2.6, we found the 95th percentile from the Bayesian inference model generally underestimated the highest numbers of anticipated captures and we therefore used a nonparametric bootstrap analysis to estimate the 95th percentile of anticipated captures. Table 35 contains the resulting predictions of future total annual interactions (mean and 95th percentile) and the maximum 5-year running average and sum for leatherback sea turtles in the United States WCPO purse seine fishery.

Table 35. Mean and 95th percentile of leatherback sea turtle posterior estimated exposures in terms of anticipated annual interactions with fishing gear (inclusive of FAD interactions, see Section 5.2.2.2, *Entanglement in FADs – Sea Turtles*) and the maximum 5-year running average and running sum.

	Annual Mean	**Annual 95th percentile**
Predicted Exposures	1	2
Maximum Running Average and Sum		
	5-yr Running Average	**5-yr Running Sum**
Predicted Exposures	0.5	3

South Pacific Loggerhead Sea Turtle

South Pacific loggerhead sea turtles were the second-most common species of sea turtle observed interacting with the United States WCPO purse seine fishery with 68 observed interactions between 2008 and 2018 (including FAD interactions; Table 26). After proportioning the unidentified sea turtles and accounting for unobserved sets the estimated total number of interactions with loggerhead sea turtles from 2008-2018 is 121 (95% CI: [107, 144]). They may also be attracted to FADs as sources of prey from the colonizing epifauna and shelter, although comparing FAD sets to free sets, 72% of the South Pacific loggerhead sea turtle interactions occurred during free sets.

Table 36 describes the number of South Pacific loggerhead sea turtles captured in the United States WCPO purse seine fishery between 2008 and 2018 by their interaction category and release condition. Sixty-five South Pacific loggerhead sea turtles have been observed captured by this fishery during sets. Of those, 75% were considered by the observers to be uninjured at the time of release, 17% were released injured and 8% were released either dying or dead (Table 37).

Table 36. Number of South Pacific loggerhead sea turtles captured by the United States WCPO Purse Seine fishery during active fishery sets between 2008 and 2018 by the interaction type and release condition.

	Release Condition				
Interaction Category	**A1 Alive and healthy**	**A2 Alive but injured**	**A3 Alive but unlikely to live**	**D Dead**	**Grand Total**
Free Set	33	5	2	3	43
FAD Set	11	6	0	0	17
Unknown Set Type	5	0	0	0	5
Grand Total	49	11	2	3	65

Table 37. Proportions of South Pacific loggerhead sea turtles captured by the United States WCPO Purse Seine fishery during active fishery sets between 2008 and 2018 by the interaction type and release condition.

	Release Condition				
Interaction Category	**A1 Alive and healthy**	**A2 Alive but injured**	**A3 Alive but unlikely to live**	**D Dead**	**Grand Total**
Free-Set	0.508	0.077	0.031	0.046	0.662
FAD Set	0.169	0.92	0	0	0.262
Unknown Set Type	0.077	0	0	0	0.077
Grand Total	0.754	0.169	0.031	0.046	1.00

Demographic Patterns of Exposure

As described previously in the *Status of Listed Resources* section of this biological opinion all of the loggerhead sea turtles sampled in this fishery are likely South Pacific loggerhead sea turtles. The primary nesting for this species is in Eastern Australia and to a lesser extent New Caledonia, and genetics studies have indicated a lack of genetic distinction between these nesting areas (Boyle et al 2009), there for we do not consider any subpopulations within this species. The mean length of South Pacific loggerhead sea turtles interacting with the WCPO purse seine fishery from 2008-2015 was 48.3 cm SCL for landed and measured turtles (range: 35-80 cm SCL; Table 38Table 38). These lengths are consistent with pelagic stage juveniles for the Australian breeding population as the new recruits move to nearshore foraging habitats which range from 66.7-93.9 cm curved carapace length (CCL) with a mean of 78.6 cm CCL (Limpus and Limpus 2003a) which is approximately 73.3 cm SCL (Snover et al. 2010). Limpus and

Limpus (2003a) found a mean duration of 13 years from coastal recruitment to first breeding for the South Pacific loggerhead sea turtle.

There are no estimates of pelagic stage growth rates or age to maturity for South Pacific loggerhead sea turtles. As a proxy, we use pelagic stage growth rates for Atlantic loggerhead sea turtles (Avens et al. 2013). Avens et al. (2013) report size-specific mean pelagic growth rates for Atlantic loggerhead sea turtles: 2.7 cm/yr. for 30-39 cm SCL, 2.8 cm/yr. for 40-49 cm SCL, 3.1 cm/yr. for 50-59 cm SCL, 2.1 cm/yr. for 60-69 cm SCL and 2.1 for 70-79 cm SCL. We applied these growth rates to each loggerhead sea turtle measured by United States WCPO purse seine fishery observers to estimate the mean time to reach 73 cm SCL, or the mean size at recruitment to nearshore forage habitats in Australia (Limpus and Limpus 2003a). We then added 13 yr. as the mean time from near shore recruitment to reproductive maturity for female loggerhead sea turtles in Australia (Limpus and Limpus 2003a) to estimate total time remaining until maturity. Limpus and Limpus (2003a) report that 29% of the Australian loggerhead sea turtle population is female.

Table 38. Straight carapace length (SCL) measurements and estimated age for the loggerhead sea turtles that were measured by observers during interactions with the United States WCPO purse seine fishery between 2008 and 2015. Estimated time to maturity is based on juvenile growth rates estimates of Avens et al. (2013) and time from recruitment to maturity estimates of Limpus and Limpus (2003a).

SCL (cm)	**Estimated Time Until Reproductive Maturity (yrs.)**
30	29.7
35	27.8
36	27.5
40	26.0
40	26.0
40	26.0
43	24.9
44	24.6
44	24.6
45	24.2
46	23.8
46	23.8
46	23.8
50	22.4
51	22.1
58	19.8
59	19.5
62	18.2
70	14.4
80	13.0

Spatial and Temporal Patterns of Exposure

Loggerhead sea turtle interactions with the United States WCPO purse seine fishery are distributed throughout the *Action Area*. Most of the interactions were west of 180° longitude (69%). To the east of 180° longitude, 94% of the interactions were south of the equator, while west of 180° 58% of the interactions were south of the equator.

The highest recorded number of loggerhead sea turtle interactions occurred in 2009 with 13 interactions (Figure 40). Interactions occur in all months, with the highest numbers of interactions in May followed by August and September (Figure 41).

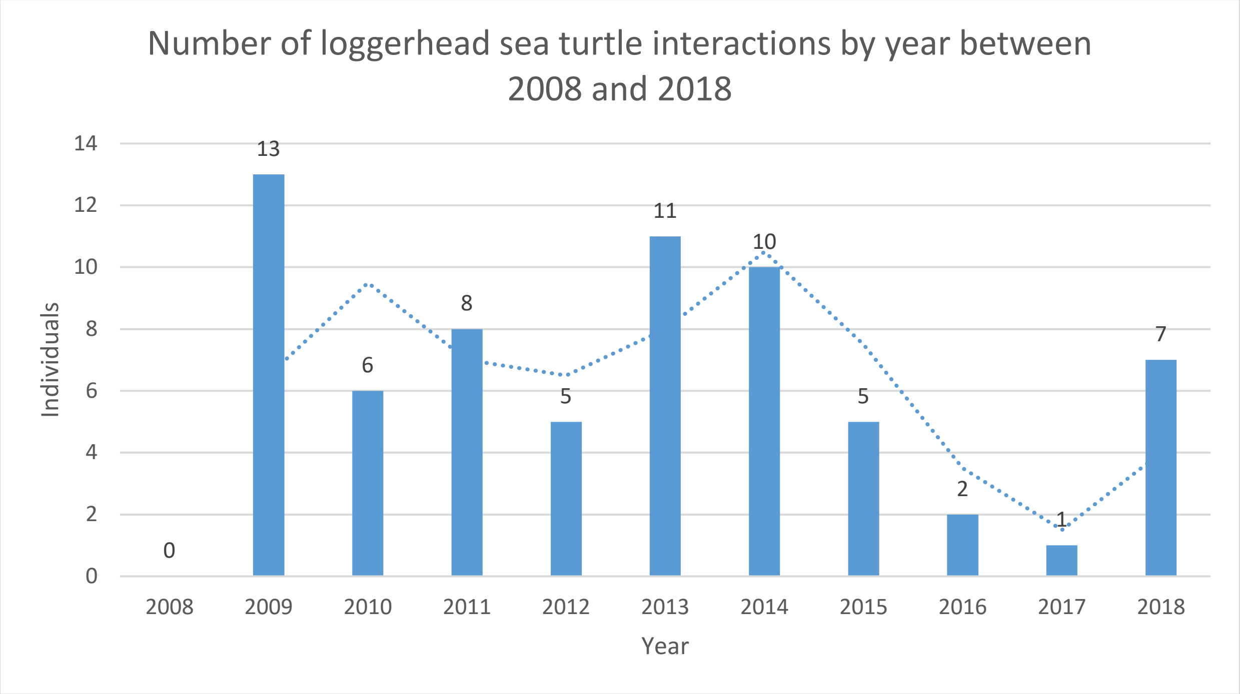


Figure 40. Loggerhead sea turtle interactions by year in the United States WCPO purse seine fishery between 2008 through 2018 with 2-year moving average (blue dashed line).

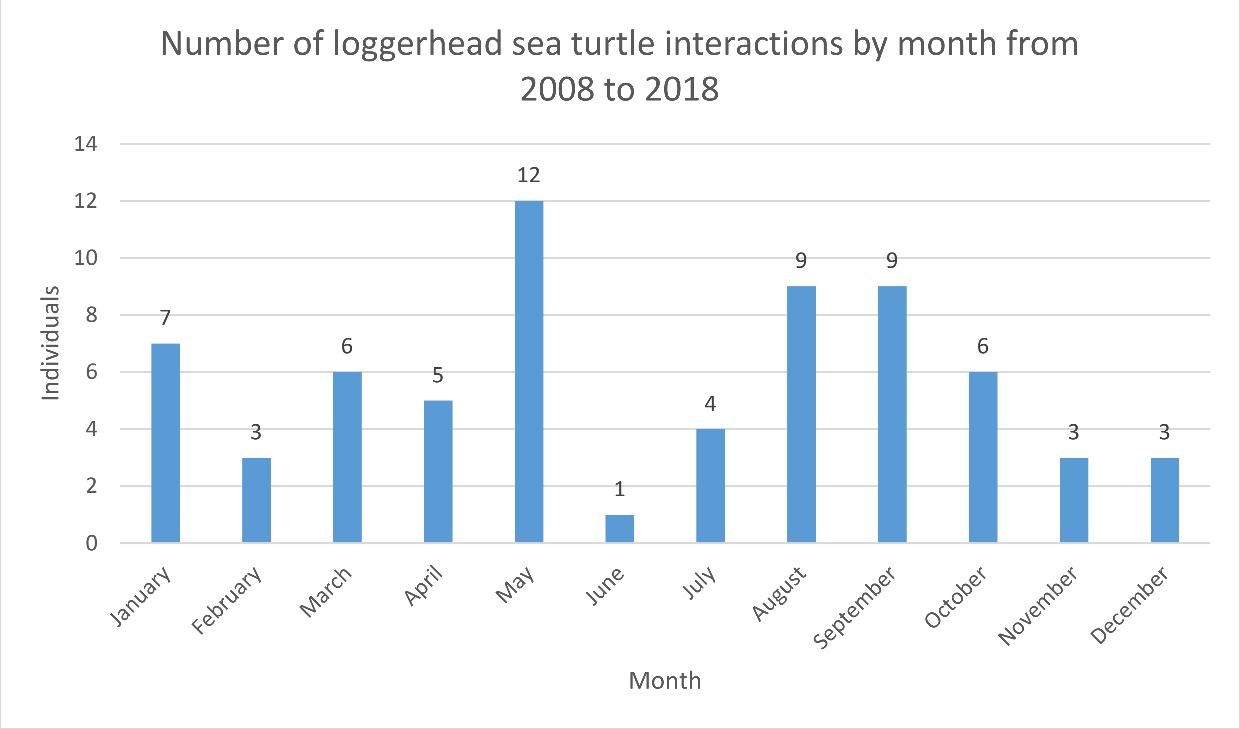


Figure 41. Loggerhead sea turtle interactions by month in the United States WCPO purse seine fishery between 2008 through 2018.

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interactions using Bayesian statistical inference techniques as described in the *Approach to the Assessment* section of this biological opinion (NMFS 2019c). The predictions were based on the recorded bycatch of loggerhead sea turtles in the United States WCPO purse seine fishery and the assumption of 3,100 annual sets with 1,581 of those being FAD sets. As noted in Section 2.2.2.6, we found the 95th percentile from the Bayesian inference model generally underestimated the highest numbers of anticipated captures and we therefore used a nonparametric bootstrap analysis to estimate the 95th percentile of anticipated captures. Table 39 contains the resulting predictions of future total annual interactions (mean and 95th percentile) and the maximum 5-year running average in the United States WCPO purse seine fishery.

Table 39. Mean and 95th percentile of loggerhead sea turtle posterior estimated exposures in terms of anticipated annual interactions with fishing gear (inclusive of FAD interactions, see Section 5.2.2.2, *Entanglement in FADs – Sea Turtles*) and the maximum 5-year running average and running sum.

	Annual Mean	**Annual 95th percentile**
Predicted Exposures	5	8
Maximum Running Average and Sum		
	5-yr Running Average	**5-yr Running Sum**
Predicted Exposures	5.0	25

Olive Ridley Sea Turtle

Numbers of olive ridley sea turtle interactions in the United States WCPO purse seine fishery are similar to those of hawksbill sea turtles with 57 observed interactions from 2008 to 2018 (including FAD interactions; Table 40). After proportioning the unidentified sea turtles and accounting for unobserved sets the estimated total number of interactions with olive ridley sea turtles from 2008-2018 is 105 (95% CI: [90, 118]). As the olive ridley sea turtle life history is primarily pelagic, it was not anticipated that they would be attracted to FADs as sources of food or shelter, and olive ridley sea turtle interactions occurred at a slightly higher frequency on free sets compared to FAD sets, with 54% of fishery set interactions occurring on free sets.

Table 40 describes the number of olive ridley sea turtles captured in the United States WCPO purse seine fishery during active fishery sets between 2008 and 2018 by their interaction type and release condition. Fifty-four olive ridley sea turtles have been observed captured by this fishery during sets. Of those, 72% were considered by the observers to be uninjured at the time of release, 28% were released injured and none were released dying or dead (Table 41).

Table 40. Number of olive ridley sea turtles captured by the United States WCPO purse seine fishery during active fishery sets between 2008 and 2018 by the interaction type and release condition.

	Release Condition				
Interaction Category	**A1 Alive and healthy**	**A2 Alive but injured**	**A3 Alive but unlikely to live**	**D Dead**	**Grand Total**
Free-Set	22	7	0	0	29
FAD Set	16	8	0	0	24
Unknown Set Type	1	0	0	0	1
Grand Total	39	15	0	0	54

Table 41. Proportions of olive ridley sea turtles captured by the United States WCPO purse seine fishery during active fishery sets between 2008 and 2018 by the interaction type and release condition.

	Release Condition				
Interaction Category	**A1 Alive and healthy**	**A2 Alive but injured**	**A3 Alive but unlikely to live**	**D Dead**	**Grand Total**
Free-Set	0.407	0.130	-	-	0.537
FAD Set	0.296	0.148	-	-	0.444
Unknown Set Type	0.019	-	-	-	0.019
Grand Total	0.722	0.278	-	-	1.00

Demographic Patterns of Exposure

As described in the *Status of Listed Resources* section of this biological opinion, there are two listed populations of olive ridley sea turtles that are exposed to the action; the endangered breeding population on Mexico's Pacific coast and the threatened global population. There are no genetic data available to determine whether olive ridley sea turtles interacting with the United States WCPO purse seine fishery originate from the East or West Pacific nesting populations. Genetic data on olive ridley sea turtles interacting with the ASLL fishery suggest that 60% of them originate from the Western Pacific while the remaining 40% are from the eastern Pacific nesting beaches and are a combination of the threatened and endangered populations. (NMFS unpublished data). The American Samoan longline fishery overlaps with the *Action Area*. Given that (1) the distribution of the interactions with the United States WCPO purse seine fishery (see the *Spatial and Temporal Patterns of Exposure* section below) indicate that 89% of olive ridley sea turtles were captured west of 180°W and Eastern Pacific olive ridleys are not thought to use habitats that far west in the Pacific Ocean (Morreale et al. 2007); and (2) the majority of olive ridley sea turtles interacting with the ASLL fishery are from the West Pacific populations, we anticipate a higher proportion of the olive ridley turtles captured in the purse seine fishery are from the threatened global population generally, and the West Pacific populations specifically.

The olive ridley sea turtles that have interacted with the United States WCPO purse seine fishery ranged in size from 15 to 88 cm SCL (SPC unpublished data; Table 42). The mean length of measured olive ridley sea turtles interacting with the United States WCPO purse seine fishery was 49 cm SCL. As described in the *Status of Listed Resources* section of this biological opinion, the minimum size of adults reported in the literature varies and may be as low as 53 cm SCL (Zug et al. 2006). Hence, the sizes of olive ridley sea turtles interacting with the purse seine fishery are representative of juveniles and adults, with approximately 61% juveniles and 39% adult or sub-adults. Using the age and size relationship of Zug et al. (2006) we estimated the ages of the juvenile olive ridley sea turtles interacting with the fishery to range from 5 to greater than 25 years (Table 42).

Table 42. Straight carapace length (SCL) measurements and estimated age for olive ridley sea turtles which have interacted with the United States WCPO purse seine fishery between 2008-2018 and for which the observer records contained measurements (*n* = 31). Age estimates are based on Zug et al. (2006).

SCL (cm)	**Estimated Age (yrs.)**
15	4.7
18	5.5
30	8.3
30	8.3
32	8.7
32	8.7
34	9.1
35	9.3
38	9.9
40	10.3
40	10.3
40	10.3
40	10.3
40	10.3
42	10.8
45	11.3
47	11.6
49	12.1
50	12.3
54	13.5
59	21.1
60	22.2
61	>25
64	>25
64	>25
65	>25
72	>25

SCL (cm)	Estimated Age (yrs.)
72	>25
75	>25
87	>25
88	>25

Spatial and Temporal Patterns of Exposure

We analyzed the location and dates of the interactions that occurred from 2008 to 2018 to see if any spatial or temporal patterns emerged. Olive ridley sea turtle interactions with the United States WCPO purse seine fishery are distributed throughout the *Action Area* of the purse seine fishery. Most of the interactions were west of 180° longitude (89%). To the west of 180° longitude, 67% of the interactions occurred south of the equator. As noted above, given that Eastern Pacific olive ridleys are not thought to use habitats in the Western Pacific Ocean (Morreale et al. 2007), we anticipate that olive ridley sea turtles capture west of 180° longitude are from the threatened West Pacific population. Olive ridley sea turtles capture east of 180° longitude are likely a combination of East and West Pacific olive ridley sea turtles and may contain individuals from the endangered population.

The highest recorded number of olive ridley sea turtle interactions from 2008 to 2018 occurred in 2011 with 19 interactions (Figure 42). Interactions occur in all months, with the highest numbers of interactions in March and October (Figure 43).

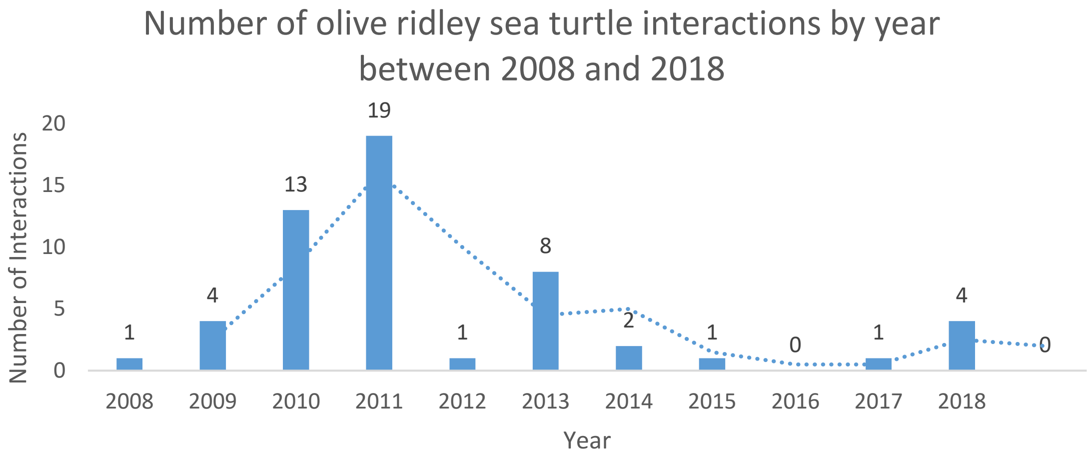


Figure 42. Olive ridley sea turtle interactions by year in the United States WCPO purse seine fishery between 2008 through 2018 with 3-year moving average (blue dashed line).

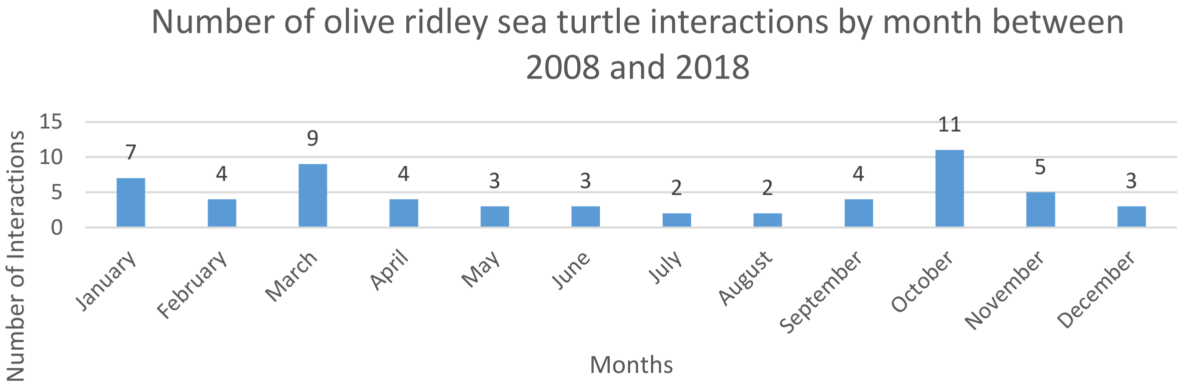


Figure 43. Olive ridley sea turtle interactions by month in the United States WCPO purse seine fishery between 2008 through 2018.

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interactions using Bayesian statistical inference techniques as described in the *Approach to the Assessment* section of this biological opinion (NMFS 2019c). The predictions were based on the recorded bycatch of olive ridley sea turtles in the United States WCPO purse seine fishery and the assumption of 3,100 annual sets with 1,581 of those being FAD sets. As noted in Section 2.2.2.6, we found the 95th percentile from the Bayesian inference model generally underestimated the highest numbers of anticipated captures and we therefore used a nonparametric bootstrap analysis to estimate the 95th percentile of anticipated captures.

Table 43 contains the resulting predictions of future total annual interactions (mean and 95th percentile) and the maximum 5-year running average and sum for olive ridley sea turtles in the United States WCPO purse seine fishery.

Table 43. Mean and 95th percentile of olive ridley sea turtle posterior estimated exposures in terms of anticipated annual interactions with fishing gear (inclusive of FAD interactions, see Section 5.2.2.2, *Entanglement in FADs – Sea Turtles*) and the maximum 5-year running average and running sum.

	Annual Mean	**Annual 95th percentile**
Predicted Exposures	4	7
Maximum Running Average and Sum		
	5-yr Running Average	**5-yr Running Sum**
Predicted Exposures	6.3	32

Green Sea Turtle

Green sea turtles were the most common species of sea turtle observed interacting with the United States WCPO purse seine fishery with 97 observed interactions from 2008 to 2018 (including FAD interactions; Table 26). After proportioning the unidentified sea turtles and accounting for unobserved sets the estimated total number of interactions with green sea turtles from 2008-2018 is 173 (95% CI: [152, 200]. They may be attracted to FADs as sources of prey from the colonizing epifauna and shelter, however higher numbers of green sea turtles were observed captured by the fishery during free sets compared to FAD sets (60% of fishery set interactions were during free sets; Table 44).

Table 44 describes the number of green sea turtles captured in the United States WCPO purse seine fishery between 2008 and 2018 by their interaction category and release condition. Eighty-seven green sea turtles have been observed captured by or interacting with this fishery. Of those, 79% were considered by the observers to be uninjured at the time of release, 14% were released injured and 7% were released dying or dead (Table 45).

Table 44. Number of green sea turtles captured by the United States WCPO purse seine fishery during active fishery sets between 2008 and 2018 by the interaction type and release condition.

	Release Condition				
Interaction Category	**A1** **Alive and healthy**	**A2** **Alive but injured**	**A3** **Alive but unlikely to live**	**D** **Dead**	**Grand Total**
Free-Set	41	5	0	1	47
FAD Set	20	6	4	1	31
Unknown Set Type	8	1	0	0	9
Grand Total	69	12	4	2	87

Table 45. Proportions of green sea turtles captured by the United States WCPO Purse Seine fishery during active fishery sets between 2008 and 2018 by the interaction type and release condition.

	Release Condition				
Interaction Category	**A1 Alive and healthy**	**A2 Alive but injured**	**A3 Alive but unlikely to live**	**D Dead**	**Grand Total**
Free-Set	0.471	0.057	0	0.011	0.540
FAD Set	0.230	0.069	0.046	0.011	0.356
Unknown Set Type	0.092	0.011	0	0	0.103
Grand Total	0.793	0.138	0.046	0.023	1.000

Demographic Patterns of Exposure

As previously discussed in the *Status of Listed Resources*; no genetic data has been collected from green sea turtles interacting with the United States WCPO purse seine fishery. While there are genetic data available from the ASLL fishery, this fishing area overlaps with only a small a portion of the *Action Area* of the purse seine fishery. We therefore will consider the impacts of all interactions and potential mortalities to each potentially affected green sea turtle species within the Pacific. The potentially affected species are the Central South Pacific, the Southwest Pacific, the East Pacific, the Central West Pacific, and the East Indian-West Pacific.

Looking at the interactions between 2008 and 2018, all 32 measured green sea turtles would be considered juveniles as green sea turtles do not mature in this region until reaching 80 cm (NMFS and FWS 1998b; Zug et al. 2002; Table 46). The two larger green sea turtles (65 and 67 cm SCL) may have been sub-adults. There is not enough information available to identify green sea turtles captured in this fishery to sex, and the size of those captured does not provide information assist to provide us any indication of gender. Table 46 also shows age estimates for each measured turtle based on skeletochronology aging data presented by Zug and Balazs (2000) and Zug et al. (2002). We estimate that these turtles ranged between 3 years old for the individual at 18 cm, upwards of 17 years old for the turtle at 67 cm SCL.

Table 46. Straight carapace length (SCL) measurements and estimated age for green sea turtles which have interacted with the United States WCPO purse seine fishery between 2008 and 2015 and for which the observer records contained measurements (*n* = 32). Age estimates are based on Zug and Balazs (2000), and Zug et al. (2002).

SCL (cm)	Estimated Age (yrs.)
18	2.6
21	3.6
28	4.8
29	5.0
30	5.2
31	5.4
32	5.7
34	6.1
34	6.1
35	6.4
35	6.4
36	6.6
36	6.6
37	6.8
38	7.1
39	7.3
39	7.3
40	7.5
42	8.2
43	8.5
45	9.1
45	9.1
45	9.1
46	9.4
48	10.1
48	10.1
50	10.7

SCL (cm)	Estimated Age (yrs.)
54	12.0
55	12.3
56	12.6
65	16.0
67	16.8

Spatial and Temporal Patterns of Exposure

We analyzed the location and dates of the interactions that occurred from 2008 to 2018 to see if any spatial or temporal patterns emerged. Green sea turtle interactions with the United States WCPO purse seine fishery are distributed throughout the *Action Area* of the purse seine fishery. Most of the interactions were west of 180° longitude (69%). To the west of 180° longitude, 63% of the interactions occurred south of the equator; similarly, to the east of 180° longitude, 64% of the interactions occurred south of the equator.

The highest recorded number of green sea turtle interactions from 2008 to 2018 occurred in 2010 with 18 interactions (Figure 44). Interactions occur in all months, with the highest numbers of interactions in January, March, April, and May (Figure 45).

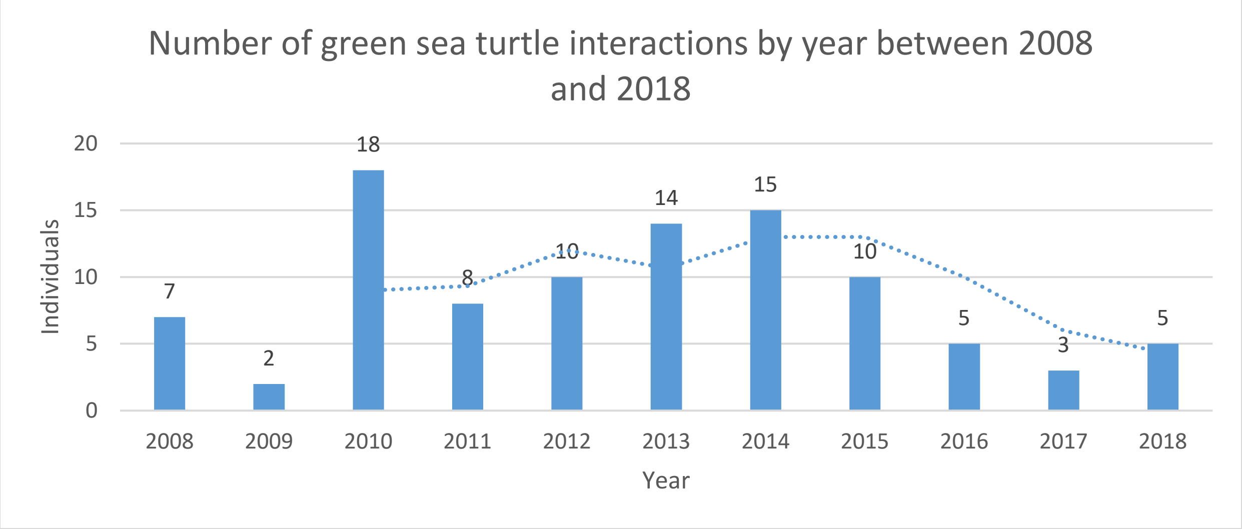


Figure 44. Green sea turtle interactions by year in the United States WCPO purse seine fishery between 2008 through 2018 with 3-year moving average (blue dashed line).

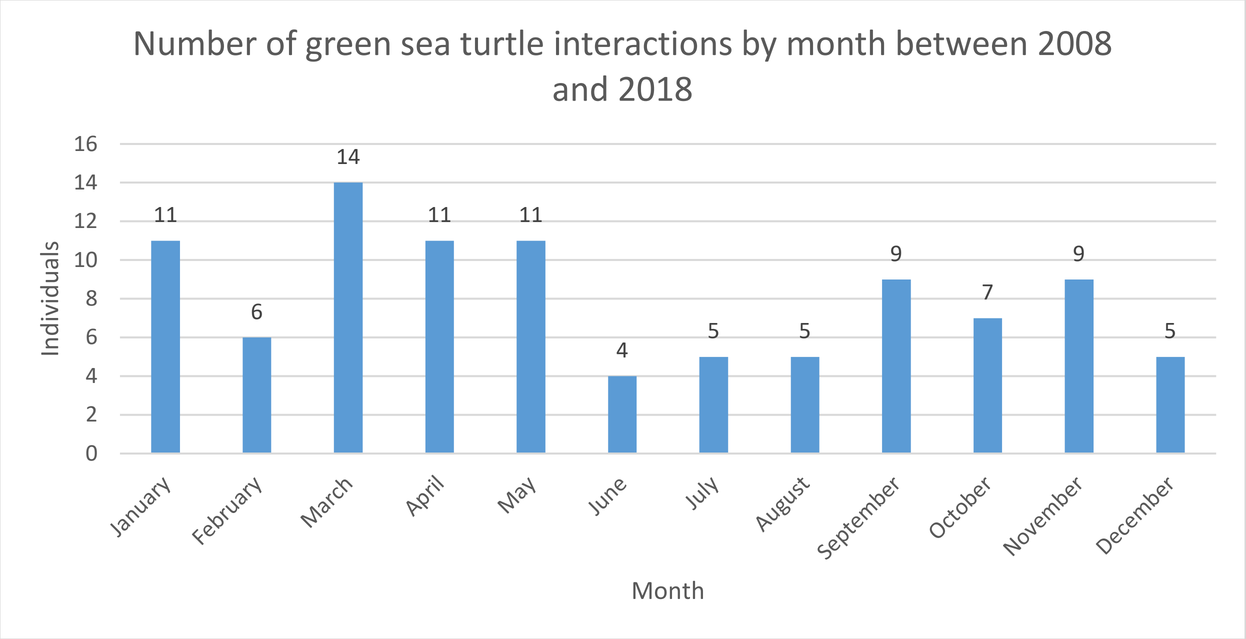


Figure 45. Green sea turtle interactions by month in the United States WCPO purse seine fishery between 2008 through 2018.

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interactions using Bayesian statistical inference techniques as described in the *Approach to the Assessment* section of this biological opinion (NMFS 2019c). The predictions were based on the recorded bycatch of green sea turtles in the United States WCPO purse seine fishery and the assumption of 3,100 annual sets with 1,581 of those being FAD sets. As noted in Section 2.2.2.6, we found the 95th percentile from the Bayesian inference model generally underestimated the highest numbers of anticipated captures and we therefore used a nonparametric bootstrap analysis to estimate the 95th percentile of anticipated captures. Table 47 contains the resulting predictions of future total annual interactions (mean and 95th percentile) and the maximum 5-year running average and sum for green sea turtles in the United States WCPO purse seine fishery.

Table 47. Mean and 95th percentile of green sea turtle posterior estimated exposures in terms of anticipated annual interactions with fishing gear (inclusive of FAD interactions) and the maximum 5-year running average and running sum.

	Annual Mean	**Annual 95th percentile**
Predicted Exposures	7	11
Maximum Running Average and Sum		
	5-yr Running Average	**5-yr Running Sum**
Predicted Exposures	7.6	38

5.2.1.3 Elasmobranchs

Oceanic Whitetip Shark

Demographic Patterns of Exposure

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 of the animal, and the crew. However, sharks may be boarded if the crew is unable to identify a shark's presence within the catch. Currently, demographic data for oceanic whitetip sharks are not collected in the purse seine fishery. At this time, we do not have information on the size, age, or gender classes of oceanic whitetip sharks are being caught in this fishery. However due to the geographic range of the fishery, only oceanic whitetips in the Pacific Ocean would be affected by this action (see the *Status of the Listed Resources* for discussion on genetic differentiation between ocean basins). Although abundance estimates are not available for the eastern Pacific or global population, and the current available genetic information is not sufficient to genetically differentiate oceanic whitetip sharks between the western and eastern Pacific. That is to say, the stock assessments (Rice and Harley 2012; Tremblay-Boyer et al. 2019) only provide a picture of a portion of the Pacific population by assessing only a western "stock". We expect all individuals to be from this portion of the Pacific population based on the location of the observed interactions. Table 48 summarizes oceanic whitetip shark interactions in the purse seine fleet from 2008 to 2018, and their release conditions. These data are inclusive of unknown species that were determined likely to be oceanic whitetip sharks.

There was only one shark recorded as alive and entangled (A4) and examination of the observer record did not provide any relevant commentary on the outcome of the interaction. As previously discussed, unidentified shark records were prorated for other identified shark species, including oceanic whitetip sharks. This resulted in up to five interactions that are assigned this condition code (A4). Three of these five are shown as whole sharks in the data. The remaining two were an aggregation of decimals in the data, which is unlikely when considering individual sharks. However, over the time span considered, two sharks are expected to possess this condition code when returned to the ocean. We therefore reassigned these sharks to condition code A2, as injuries would be expected from entanglements.

Table 48. Oceanic whitetip shark condition codes as described by the observer data from 2008 to 2018.

Release condition	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	Total
A0	56	46	67	48	61	41	41	62	56	40	40	105
A1	32	25	38	27	35	23	31	27	22	21	35	558
A2	40	32	48	34	44	30	31	27	24	28	59	316
A3	9	7	10	7	9	6	6	8	6	5	12	397
A4	0	0	1	0	1	0	0	1	0	0	0	85
D	95	78	115	98	103	70	98	62	72	58	76	3
Total	232	188	279	214	253	170	207	187	180	152	222	2284

Adjusted categorical values are listed in Table 49. Accordingly, 23% of oceanic whitetip were recorded as released alive and heathy (A1), 30% of oceanic whitetip were recorded as alive but injured (A2), and 47% of the interactions were considered alive but unlikely to survive (A3) and or dead (D). Therefore, the observed *at vessel* mortality for oceanic whitetip sharks in the United States WCPO purse seine fishery with observed and unidentified sharks prorated to oceanic whitetip sharks between 2008 and 2018 was 47%.

The values listed in Table 49 are the numbers of at-vessel mortality and does not account for post release mortality which would be expected to occur as a result from capture and handling. We conducted a systematic review of the literature to assess what metrics would be relevant to the species from operational processes (e.g. gear, methodology, handling, etc.) which occur in this fishery. Hutchinson et al. (2015) determined that there is high overall mortality of silky sharks (84.2%) in this fishery, particularly once sharks were brailed. Hutchinson et al. (2015) concluded that sharks landed after brailing will not reduce fishing mortality and predicted a 6.67% survival rate for sharks that are brailed.

We expect oceanic whitetip sharks would suffer the same physiological responses from hypoxia and crushing injuries during brailing as they would be handled in the same manner as silky sharks, and considering they have the same respiratory mode. As a result, we expect the 679 sharks identified as A2 in the observer data would likely die from current handling practices as they would have been brailed. The combination of release conditions of A2, A3, and D, result in an estimated mortality of 77%. This estimate assumes that no individuals in the A1 release condition of alive and healthy would experience latent mortality as a result of their interaction. However, we do not know what proportion of individuals in A1 would die. Therefore, due to the lack of information regarding post release mortality for oceanic whitetip sharks in the available literature for purse seine fisheries, we used Hutchinson et al. (2015) for our effects analysis using a surrogate species.

Table 49. Adjusted categorical number of oceanic whitetip sharks listed as A0 and prorated to each A1, A2, A3, and D for the years 2008-2018; and A4 codes reclassified as A2.

Release condition	Individuals per Category	Percent of Total
Alive and healthy (A1)	536	23
Alive but injured (A2)	679	30
Alive but unlikely to survive (A3)	144	6
Dead (D)	925	41
Totals	2,284 (95% CI: [1983,2596])	100

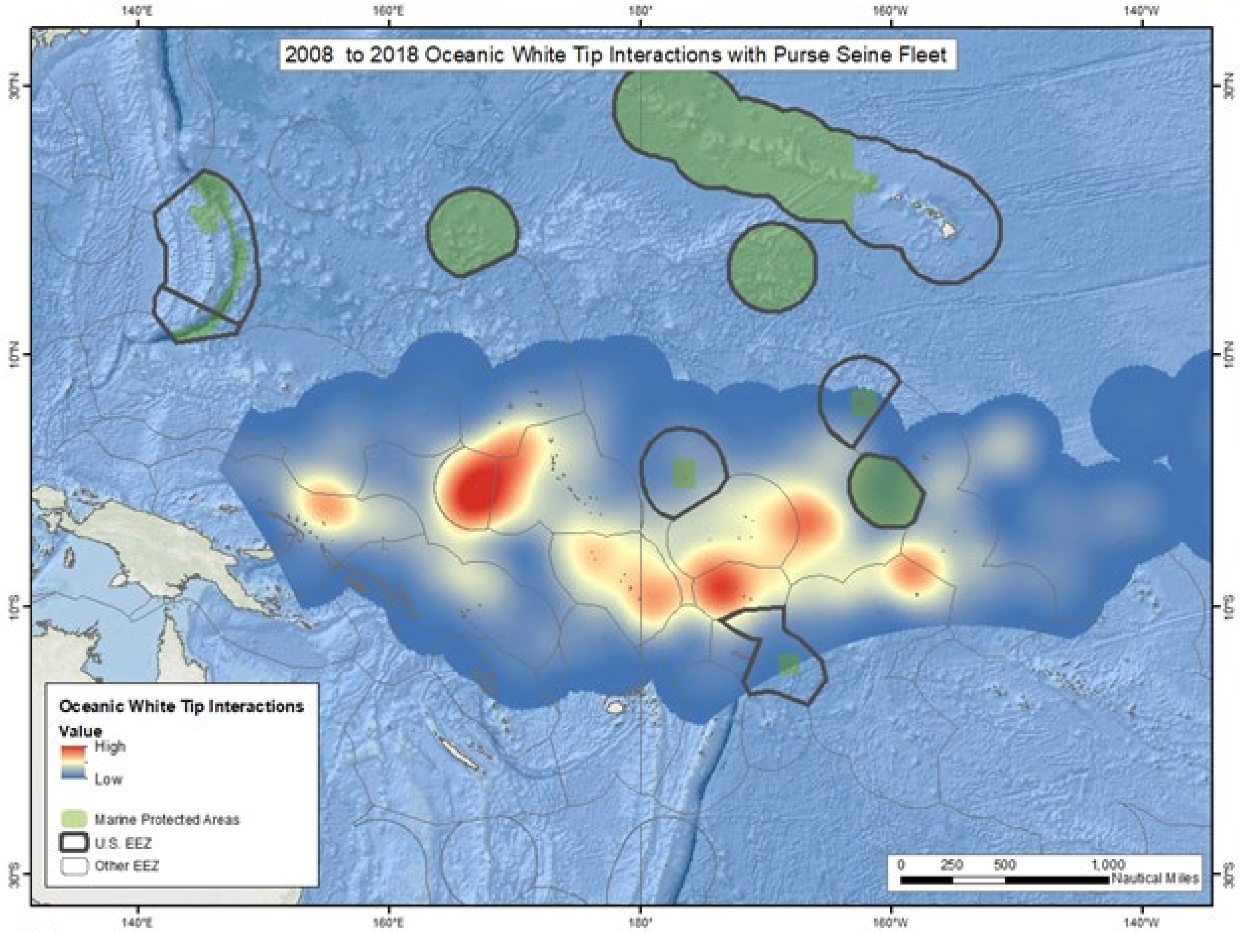
Spatial and Temporal Patterns of Exposure

Between 2008 and 2018, 680 sets have interacted with oceanic whitetip sharks resulting in the capture of 1,330 individuals (NMFS unpublished data).

A hotspot analysis was conducted using ArcGIS to map the 680 sets that have interacted with oceanic whitetip sharks (Figure 46). The ArcMap kernel density tool was used to calculate a density, which is based on point features using interaction and set data where confirmed interactions with the species occurred. This function allows for the creation of a hot spot or heat map, which illustrates the 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 edges that are more fluid. 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.

The data show that interactions appear to occur throughout the operational range of the fishery, but with higher densities of interactions within specific areas, such as Nauru or Tokelau's EEZ. Considering the total number of sets throughout this time frame within the *Action Area* was 45,905 sets, these hotspots are considered significant and represent 1.48% of the sets (680/45905*100). Meaning the vast majority of sets (98.5%) do not interact with the species, but those that do, occur in specific areas.

When comparing giant manta ray hotspot analysis (Figure 58) with oceanic whitetip sharks, we can see shark interactions range further to the east and have a higher density of sets that interacted with the species in those areas. Figure 47 shows the aggregate number of sets with interactions by month. The data show a reduction in interactions between July and September with minor peaks in April and November.



2008 to 2018 Oceanic White Tip Interactions with Purse Seine Fleet

Map by: NOAA Fisheries, Pacific Island Regional Office, G:\Projects\Protected_Resources, 10-24-2019.

Figure 46. Hotspot analysis of aggregate oceanic whitetip shark interactions (sets; *n* = 680) representing the capture of 1,330 individual sharks.

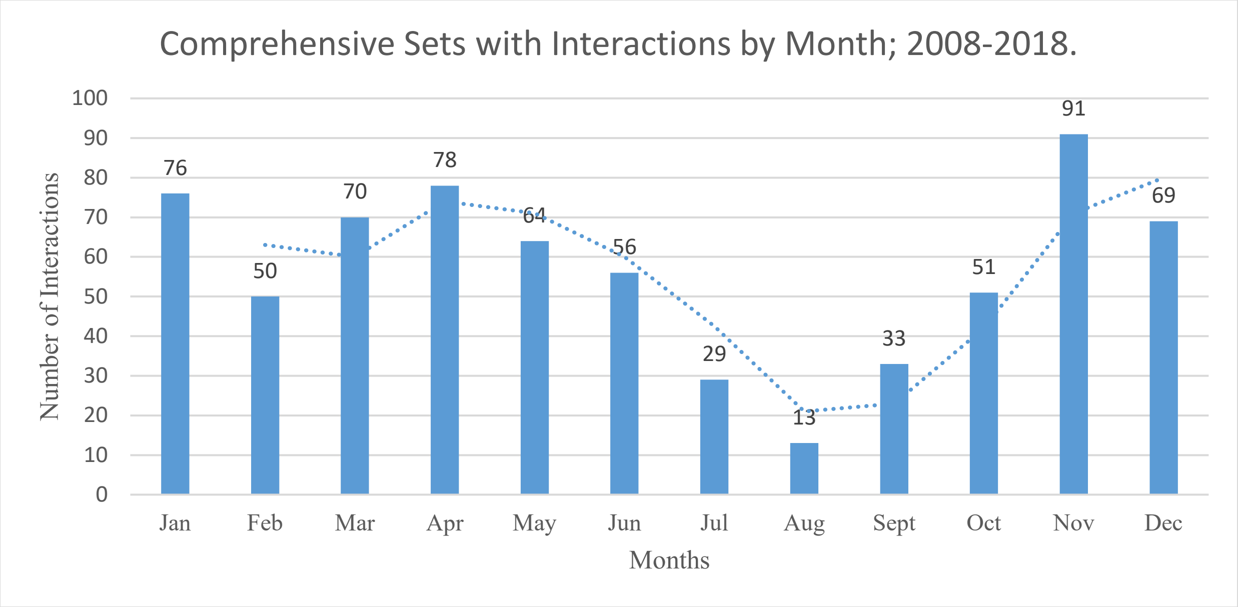


Figure 47. Comprehensive number of sets with interactions of one or more oceanic whitetip sharks by month from 2008 to 2018 in the United States WCPO purse seine fishery.

When looking at the total number of interactions by month, the same trend is evident with more individuals caught in April and November (Figure 48). These interactions are consistent with the temporal peaks identified in the recent Hawaii SSLL biological opinion (NMFS 2019b). The SSLL had a strong peak in April. At this time, we are unclear what these seasonal peaks may signify.

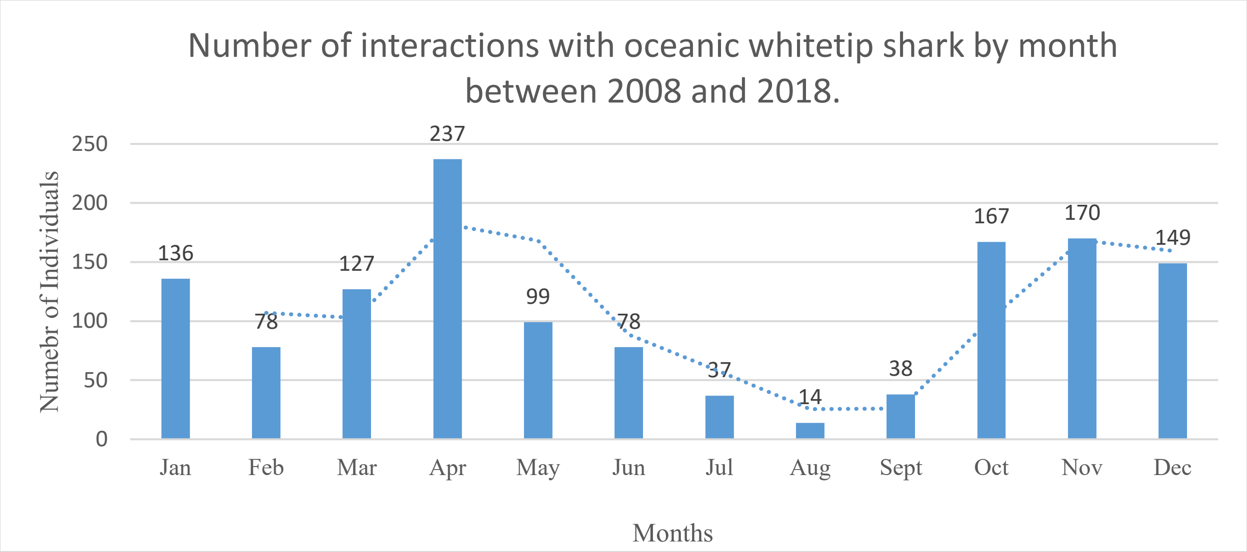


Figure 48. Comprehensive number of individual oceanic whitetip sharks interacting with the United States WCPO purse seine fishery by month from 2008 to 2018.

The average number of oceanic whitetip sharks caught per set per month from 2008 to 2018 also shows peaks, with more sharks caught per set in April and October (Figure 49). In November there are more sets that interact with the species (Figure 47), but average the same number of sharks per set as April (Figure 49).

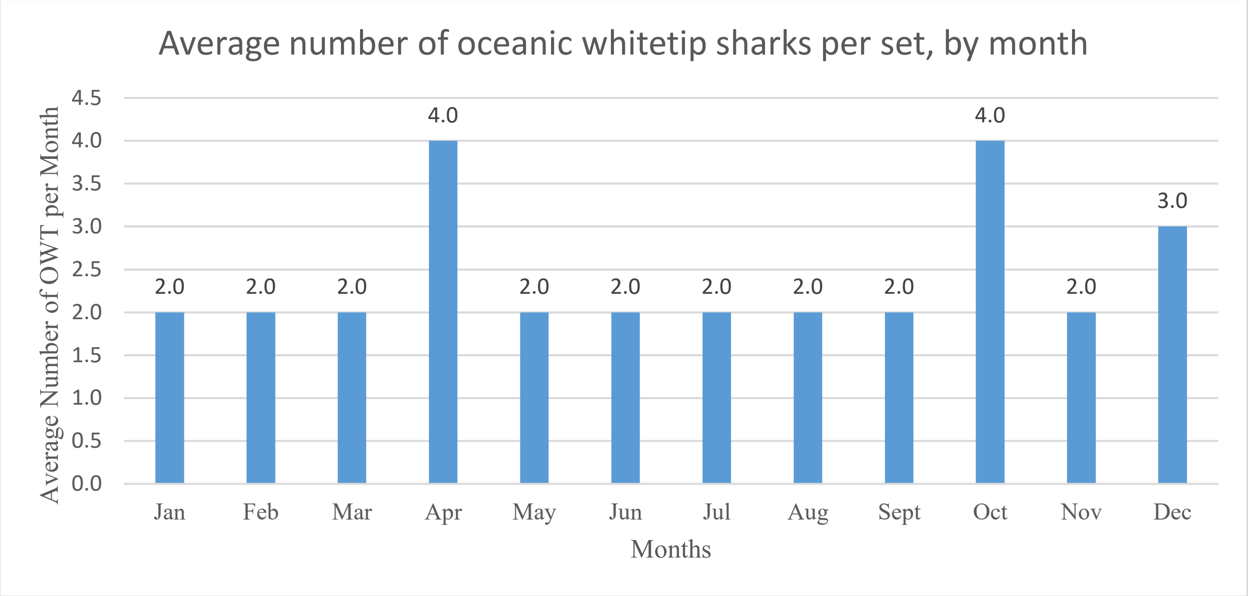


Figure 49. Average number of individual oceanic whitetip sharks caught per set per month from 2008 to 2018.

We then assessed the total number of individual sharks caught and the number of sets that interacted with the species for each year from 2008 to 2018 (Figure 50). These data show an increase in the number of sets (blue) that interact with the species and a minor decline in the number of sharks caught (orange) after peaking in 2010. A polynomial trendline was added to the figure to show the increase in interactions. Due to the variation (highs and lows) in the interaction data over this period, a moving average trendline was added for the number of individuals per year.

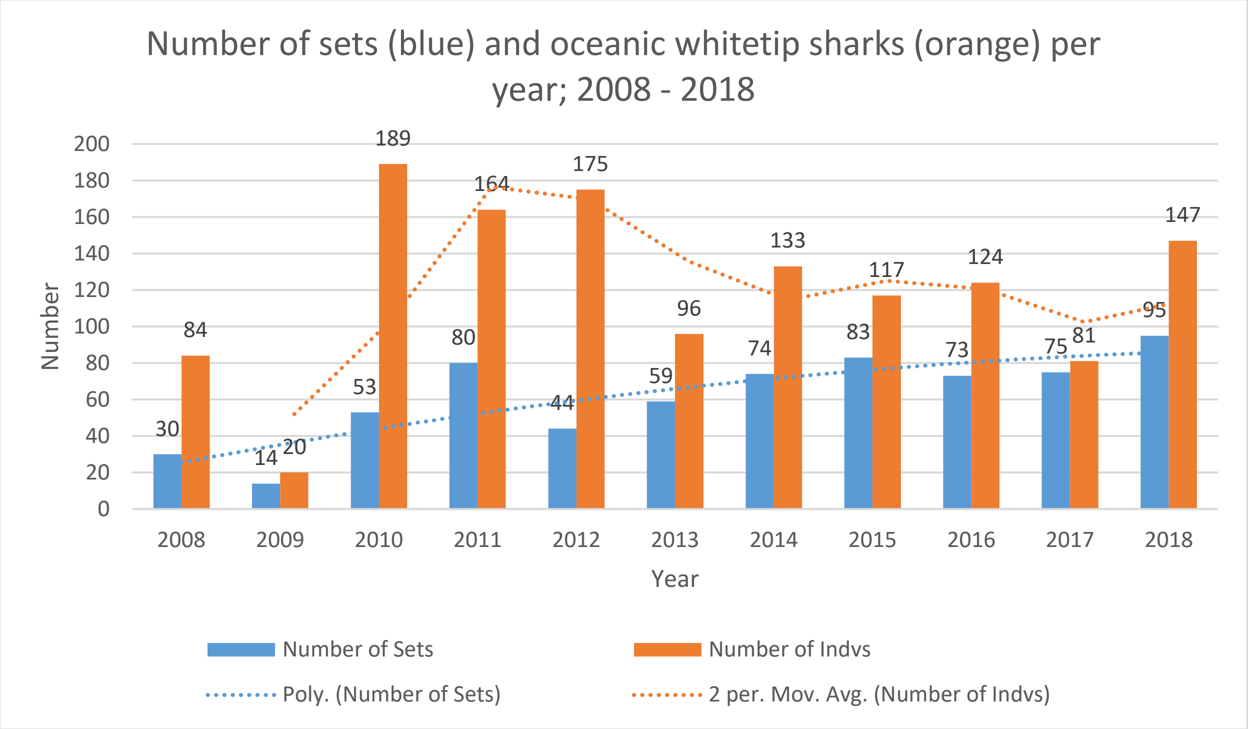


Figure 50. Number of sets (blue) with oceanic whitetip shark interactions and total individual oceanic whitetip shark interactions (orange) per year from 2008 to 2018.

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interaction levels 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 oceanic whitetip sharks and the unidentified shark classification and the assumption of 3,100 annual sets with 1,581 of those being FAD sets. As noted in Section 2.2.2.6, we found the 95th percentile from the Bayesian inference model generally underestimated the highest numbers of anticipated captures and we therefore used a nonparametric bootstrap analysis to estimate the 95th percentile of anticipated captures. Table 50 contains the resulting predictions of future annual interactions (mean and 95th percentile) and the 5-year running average and sum for oceanic whitetip sharks in the purse seine fishery (NMFS 2019c).

Table 50. Annual mean and 95th percentile of oceanic whitetip shark posterior estimated exposure in terms of anticipated annual interactions with fishing gear (i.e., anticipated annual interactions from NMFS 2019c).

	Annual Mean	**Annual 95th percentile**
Predicted Exposure	88	160
Maximum Running Average and Sum		
	5-yr Running Average	**5-yr Running Sum**
Predicted Exposures	102.8	514

Indo-West Pacific Scalloped Hammerhead Shark

Between 2008 and 2018, a total of 14 Indo-West Pacific scalloped hammerhead sharks were caught and positively identified in the United States WCPO purse seine fishery (Figure 51). 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). However, NMFS IFD estimates a total of 41 (95% CI: [31, 51]). Indo-West Pacific scalloped hammerhead sharks were captured between 2008 and 2018 using the Bayesian model approach (NMFS 2019c).

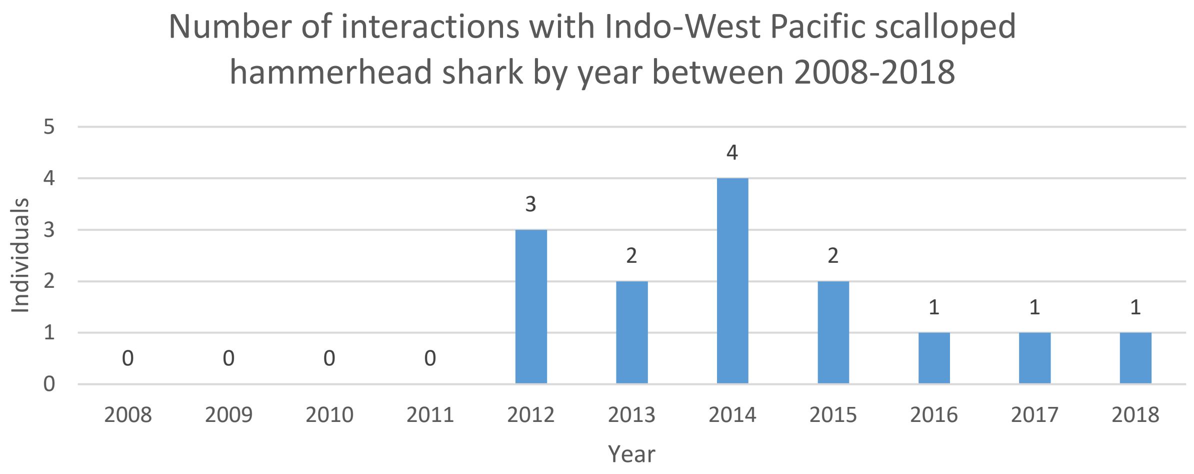


Figure 51. Number of observed Indo-West Pacific scalloped hammerhead shark interactions in the purse seine fishery per year from 2008 to 2018.

As described above for oceanic whitetip sharks, the release condition codes used for marine mammals and sea turtles (i.e. A0, A1, A2, etc.) were not regularly assigned to scalloped hammerhead sharks. Only two of the release condition codes described above were assigned to scalloped hammerhead sharks observed interacting with the United States WCPO purse seine fishery, discarded, unwanted species (DUS; 93%) and discarded, fins retained (DFR; 7% or one individual). There is no information to use regarding the release conditions of the individuals listed as DUS, we assumed that all were mortalities based on the physiology of this species, our understanding of current handling practices, the lack of information in the available literature for potential surrogate species, and through collaborative discussion with NMFS IFD. We do note that one study (Eddy et al. 2016) provides some support for our assumption of 100% mortality in the Indo-West Pacific scalloped hammerhead shark using a surrogate scalloped hammerhead species. However, due to the insufficient sample size and the expected handling practices in this specific fishery, we felt it prudent not to rely on this study alone. Therefore, we assume all 14 observed Indo-West Pacific scalloped hammerhead sharks died from their interactions with this fishery (i.e. 100% mortality). This is a reasonable assumption because, as previously discussed in the *Status of Listed Resources*, scalloped hammerhead sharks are obligate ram ventilators, meaning they must continue to move forward in order to pass water over their gills and breathe. Since each shark was landed they likely suffocated as a result of their capture.

Demographic Patterns of Exposure

The only measurements collected that on captured Indo-West Pacific scalloped hammerhead sharks are weights. Length, sex, and vertebrae are not collected in this fishery. Individuals weighed between 39.68 lbs. to 404.96 lbs., with a mean of 131 lbs. per shark. Only one shark weighed more than 176 lbs., suggesting that most sharks ranged from juvenile to adult age classes when comparing to the length-weight scatter plot provided by Romanov and Romanova (2009).

Spatial and Temporal Patterns of Exposure

We reviewed all observer data on Indo-West Pacific scalloped hammerhead sharks captured in the purse seine fishery between 2008 and 2018. We plotted the 14 interactions that have occurred in the fishery 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. Interactions span a latitudinal distance of approximately 2000 nm across the *Action Area*. The mean distance between interaction locations (*n* = 14) is approximately 351 nm (range 64 nm to 1,018 nm). Additionally, eight of the interactions occurred in Kiribati's EEZ when looking at all three distinct EEZ zones (e.g. separated by areas of high seas). Of these eight interactions, six occurred in the EEZ surrounding Kiribati's capital of Tarawa.

Temporally, some consistency in interactions occurs when looking across all years (2008-2018). Six interactions occurred in the month of November. However, because we only have a small number of data points and expansive spatial distance between interactions we cannot discern any meaningful patterns in interactions at this time.

According to the data, it also appears that this species is mostly caught due to their association with FAD sets. Ten of the 14 sharks were caught on FAD sets (71.43 %; 95th percentile: 88.28 %), the remaining four were associated with bait (*n* = 2) or were unassociated (*n* = 2) (14.29%; 95th percentile: 39.95%).

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interaction levels 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 scalloped hammerhead sharks in the purse seine fishery and are based on a future effort estimate of 3,100 sets, with 1,581 of those being FAD sets. As noted in Section 2.2.2.6, we found the 95th percentile from the Bayesian inference model generally underestimated the highest numbers of anticipated captures and we therefore used a nonparametric bootstrap analysis to estimate the 95th percentile of anticipated captures. Table 51 contains the resulting predictions of future annual interactions (mean and 95th percentile) and the 5-year running average and sum.

Table 51. Mean and 95th percentile of scalloped hammerhead shark posterior estimated exposure in terms of anticipated annual interactions with fishing gear (i.e., anticipated annual interactions from NMFS 2019c) aand the maximum 5-year running average and running sum.

	Annual Mean	**Annual 95th percentile**
Predicted Exposure	2	2
Maximum Running Average and Sum		
	5-yr Running Average	**5-yr Running Sum**
Predicted Exposures	2.0	10

Giant Manta Ray

Due to their large size and removal practices, giant manta rays are typically injured or killed when separated from the catch or during the process to return them to the ocean. Observers did not consistently record giant mantas across the whole observer data set used. Interactions between ray species, including giant manta rays, in the purse seine fishery are listed in Table 52. Similar to oceanic whitetip sharks, the release condition codes used for marine mammals and sea turtles (i.e. A0, A1, A2, etc.) have not been assigned to giant manta rays.

Common methods used to lift giant manta rays off the deck include inserting a hook attached to a pulley into the gills, punching a hole in the pectoral fin and passing a cable through it, or tying a cable to one of the cephalic lobes (Hall and Roman 2013). The internal organs of giant manta rays are not protected by a rigid skeleton but rather held in place by connective tissues, making them vulnerable to internal organ damage when removed from the water due to the pressure of their own weight (Poisson et al. 2012, 2014b). If giant manta rays are lifted by their head or tail, injuries to the spinal cord, ligaments, tendons and internal organs are likely to occur (Poisson et al. 2012; Poisson et al. 2014b). Currently nets are not used to lift mantas off of the deck, they may sit on deck until the end of the haul, and larger individuals cannot be moved off the deck by hand. Furthermore, larger individuals provide greater handling risk to the crew in order to return them to the ocean. Therefore, it is likely that any animal discarded following these handling procedures was either dead or unlikely to survive. This is consistent with other studies that indicate post-release survival is unlikely given the release methods used (Hall and Roman 2013; Croll et al. 2016). Therefore, we assume that all individuals released with unknown condition were dead, resulting in an assumed 96% mortality rate for giant manta rays interacting with the United States WCPO purse seine fishery (94% unknown release condition plus 2% known dead). As noted in the *Environmental Baseline* section, CMM 2019-05 will prohibit the use of the destructive and injurious measures described above for releasing mobulids, encourages the use of slings and the immediate (as practicable) release of landed mobulids. We acknowledge that these measures may improve post-release survival of giant manta ray, however until there are data available to ascertain the effectiveness of these measures at reducing mortality, we will continue to assume high mortality rates for this species.

While reviewing the data, it was not apparent if intentional setting on giant manta rays occurs in this fishery. The fleet identify target catch using helicopters and should be able to identify giant manta rays near the surface. While observers record FAD sets, the data does not relay whether the Captain or Fish Master are using giant manta rays on the surface, as FADs. This has been an issue with whales even with prohibitions in place. Both the IATTC and Indian Ocean Tuna Commission (IOTC) have adopted and implemented conservation measures to restrict intentional setting on mobulids within their respective convention areas. As previously described in the *Status of Listed Resources* and the *Environmental Baseline*, intentional targeting of this species occurs worldwide. Many vessels participating in this fishery also switch to the IATTC fishery in the Eastern Pacific where intentional setting on giant manta rays is already prohibited. The WCPFC has recently put forth a measure (i.e. CMM 2019-05) to dissuade intentional setting, suggesting it may be an issue within the Western Pacific but we have no data to confirm whether this practice occurs or not.

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 records use 'not elsewhere included' [nei] to indicate individuals not identified to species). 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. Likewise, if the observer is unable to identify the ray species as a manta, they are instructed to use other categories most closely resembling the potential species which would exclude the giant manta ray. 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. Manta rays were not reliably identified in 2008 or 2009, therefore we focus our analysis from 2010 to 2018. In total, from 2010 to 2018, 661 sets were observed interacting with a total of 1,523 rays, which observers identified as giant manta rays. Finally, 42,633 sets occurred within the *Action Area* from 2010 to 2018, meaning 1.55% of all sets interacted with the species over this interval.

Table 52. Number of observed interactions recorded by observers with release condition of rays captured in the purse seine fishery between 2010 and 2018. The caught and release dispositions are assumed dead per as previously described above.

Year	Giant Manta Ray	Alive	Dead	Mobulidae spp.	Alive	Dead	Unknown status
2010	242	2	240	180	2	177	1
2011	316	23	293	197	6	191	-
2012	174	4	170	654	17	637	-
2013	98	1	97	169	5	164	-
2014	285	4	281	133	8	121	4
2015	89	3	86	112	7	99	6
2016	152	0	152	104	-	104	-
2017	41	2	39	50	4	45	1
2018	125	22	103	65	14	48	3
Total	1,523	61	1,461	1,664	63	1,586	15

NMFS IFD combined the two categories ("*Mobula* nei" and "Sharks, rays, skates, etc. nei") for the purpose of this analysis. Through the Bayesian analysis approach of Martin et al. (2015) IFD estimated the number of interactions and mortalities for all years. In addition, IFD used the ratio of observed giant manta rays to *Mobula* (devil ray) to estimate the potential giant manta rays included in the *Mobula* species category. The model assumes the observations of interactions occur following a Poisson distribution with an uninformative gamma prior for rarely encountered species and a normal distribution with uninformative mean (normal) and precision (gamma) priors for more commonly encountered species. The estimates of the interaction rates, across different interaction classes were used in two ways in this analysis. First, they 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, and second to estimate the potential future impact of the fishery on ESA-listed species given anticipated numbers of FAD, free school, and total sets in future years.

Table 53. Number of estimated interactions* with individuals identified by observers as giant manta rays from 2008 to 2018 with estimated number of mortality events based on the observed data. We consider these to be overestimates given the likelihood of misidentification for this species as discussed in the text below.

Year	Alive	Dead	Total
2008	12	287	299
2009	13	322	335
2010	11	492	503
2011	26	440	466
2012	16	542	558
2013	9	291	300
2014	14	459	473
2015	12	265	277
2016	4	289	293
2017	8	164	172
2018	32	240	272
Totals	125	3,551	3,676

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

In total, based on observer data, 3,676 (95% CI: [3,119, 4,467]) giant manta rays were captured in the United States WCPO purse seine fishery between 2008 and 2018 (Table 53). This accounts for observed individuals identified as giant manta ray assuming species identifications are correct, those categorized as unidentified mobulids, which were prorated to estimate the number of them that may be giant manta rays, and estimates of the interactions that occurred in the unobserved portion of the fishery for both categories. In subsequent sections of this biological opinion, we explain why this is likely a significant overestimate of the number of individual giant manta rays that this fishery captures.

Demographic Patterns of Exposure

The sex of giant manta rays that this fishery interacts with is unknown as these data are not collected. Additionally, while weights are estimated, observers are not instructed to measure or estimate the size of giant manta rays.

According to the status review on giant manta rays (*Manta birostris*) and reef manta rays (*Manta alfredi*; Miller and Klimovich 2017), only one study, Bigelow and Schroeder (1953) present weight at birth data (12.7 kg) for *Manta birostris*, and the data are from the Northern Atlantic before the *Manta* genus was split (Miller and Klimovich 2017). However, Matsumoto and Uchida (2008), reported *M. birostris* (determined to be *M. alfredi* after the split, based on photographic evidence provided in the reference) weight at birth to be 68.5 kg (0.0685 mt;

151.07 lbs.; *n* = 1) (Miller and Klimovich 2017; M. Miller pers. comm. Oct. 8, 2019). Three individuals classified as juveniles were described by Marshall et al. (2008) and ranged from 71 to 75 kg. While classification of juvenile versus adult weights has not been determined to date in the literature for either species, we assume all individuals at 0.0685 mt or below identified in this analysis are neonates or juveniles, although these may also represent small mobulid species that have been misidentified as giant manta ray. We used this at-birth-weight maximum defined in the literature, as metrics are not well known for either species, the species-specific (*M. birostris*) data from 1953 was in another ocean basin and the species were lumped at that time, *M. birostris* are larger than *M. alfredi,* and Matusmoto and Uchida's (2008) data were more recent and conducted with animals from the Pacific – even though determined to be *M. alfredi* post publication. Additionally, Cabanillas-Torpoco et al. (2019) reported morphometric measures for a fetus *M. birostris* in Peru with a weight of 23.6 kg. However, we did not consider this weight in our analysis as the animal was still present in the womb when it was collected and it may not have been fully developed.

Based on the size of manta rays caught in this fishery, most are likely juveniles. Using estimated weights of individuals the majority of the individuals (76%) caught in this fishery are either neonates/juveniles or are another, smaller manta/mobilid species that was misidentified as giant manta ray (Figure 52).

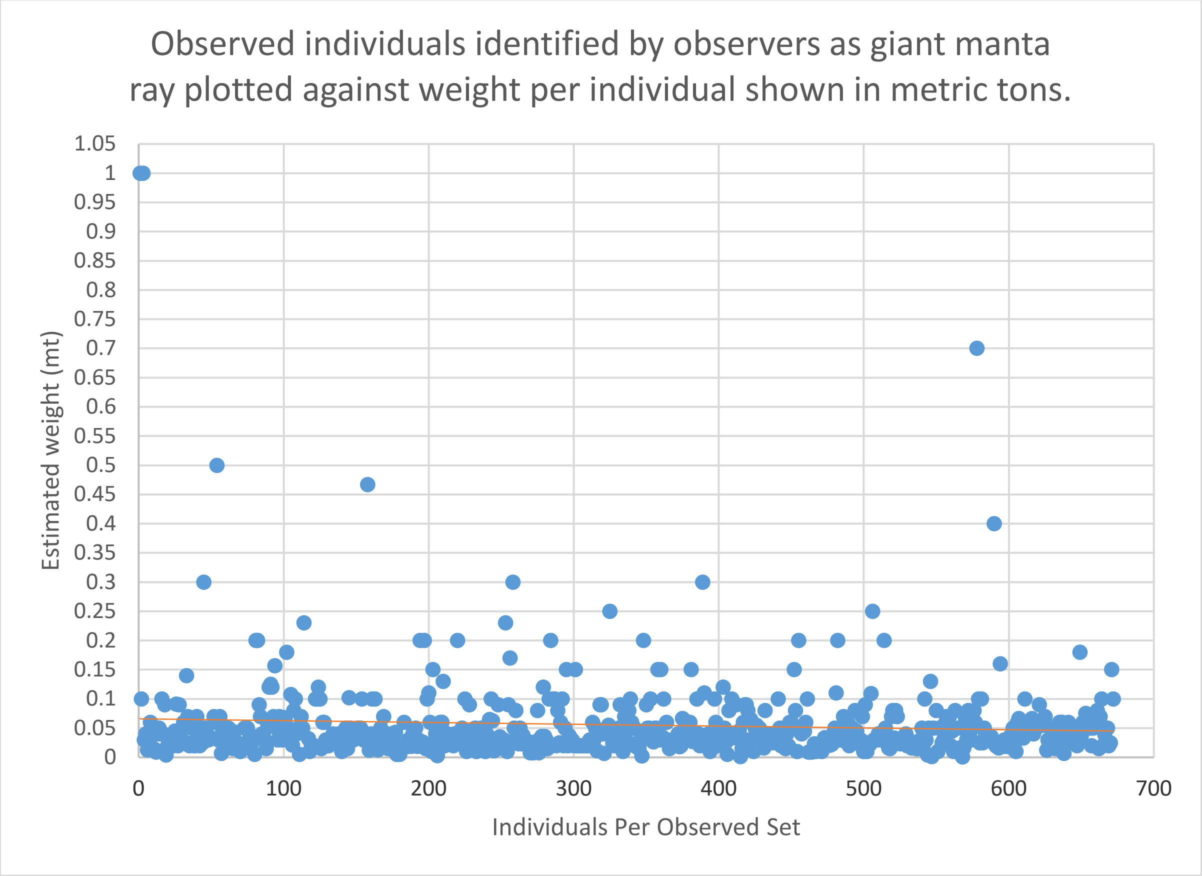


Figure 52. Observed individuals identified by observers as giant manta ray plotted against weight per individual shown in metric tons. The orange line shows the benchmark live-at-birth weight value of 0.0685 mt (68.5 kg; 151.07 lbs.) as observed from Matsumoto and Uchida (2008).

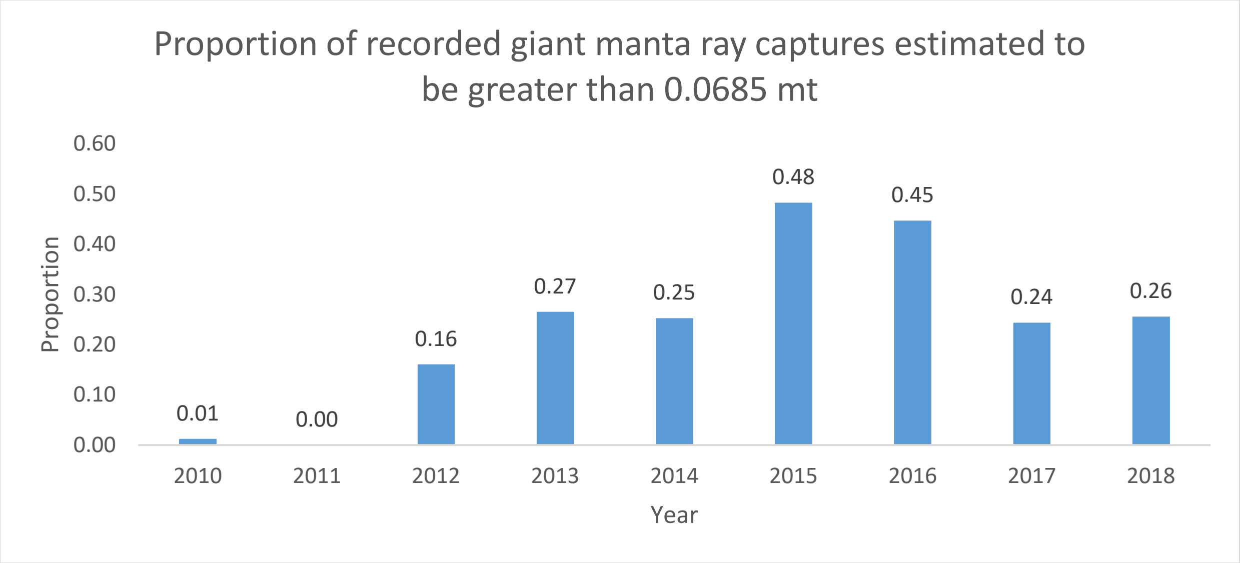


Figure 53. Proportion of individuals identified by observers as giant manta rays that were estimated as larger than 0.0685 mt (estimated birth weight for giant manta rays) in the United States WCPO purse seine fishery from 2010 to 2018. Numbers above the bars specify the exact proportion.

While the majority of individual rays captured and identified as giant manta ray were estimated by observers to be less than 0.0685 mt (Figure 52), the exact proportion of individuals estimated as larger than 0.0685 mt has changed over the years, with more larger animals being identified after 2013, which is potentially indicative of improved identification (Figure 53). We conducted a nonparametric bootstrap of the annual proportions of individuals larger than 0.0685 mt from 2010 to 2018 and estimated a mean value of 0.24 with a 95% confidence interval of 0.13 to 0.34.

There are caveats to this analysis of observed giant manta ray weights. First, there is a lack of morphological and demographic data for the species as previously discussed. Second and most importantly, weights are estimated not measured in this fishery. Third, the weight at various life stages, including maturity is not known for either gender. Maturity is typically assessed using length (disc width) measurements, clasper evaluation, ultrasounds, etc.; none of which is available in this data set. Additionally, reported weights with multiple individuals caught in the same set are assumed to be uniform in size and weight (weight was divided by number of individuals caught per set when greater than 1). Lastly, if the benchmark weight value that was determined by Bigelow and Schroder (1953) of 12.7 kg (0.0127 mt) or the fetus weight of 23.6 kg (0.023) were used instead of the higher documented weight (0.0685 mt), the number of juveniles/neonates may be overestimated.

Based on our evaluation, we expect that many of the *Mobula* captured in the United States WCPO purse seine fishery are likely misidentified to species. Most may be reef mantas or another species *Mobula* that are smaller than giant manta rays. In a separate analysis of the United States WCPO purse seine fishery observer data, Nowlis (2021) suggests that, based on reported weights and the weight distribution of all *Mobula* that may interact with the fishery, no more than 25% are likely to be giant manta ray. Until better morphological data and photos are taken of captured individual rays, we cannot be confident that the number of rays recorded as giant manta rays in this fishery is accurate. Like this fishery, large numbers of giant manta rays were reportedly captured in the IATTC fishery, but after improving identification techniques and protocols within the fishery, reef mantas were found to be caught with higher frequency than giant mantas (Stewart pers. comm. to J. Rudolph, 11 May 2020). Presently, we have no information (e.g., photos or tissue samples) other than observer identification that confirms this fishery interacts with giant manta rays. Absent at least a subsample of data that allows us to evaluate the accuracy of observer identification, we cannot use the available observer data to estimate the extent to which inaccurate identification of *Mobula* species is occurring within the United States WCPO fishery. Nevertheless, based on Nowlis (2021) and our own analysis, we expect that the number of individual rays identified by observers as giant manta rays is likely an overestimate of the number of giant manta rays interacting with this fishery. Even so, it seemed prudent to examine the spatial patterns of exposure using the data that were reported by this fishery on individuals identified by observers as giant manta rays (also referred to as observed giant manta rays), as well as the aggregated observed plus prorated estimates. We will return to the issue of species misidentification later in this biological opinion.

Spatial and Temporal Patterns of Exposure

Our analysis of observed giant manta ray data revealed both spatial and temporal trends. Temporally, when looking at all data from 2010 to 2018, sets that interacted with the species peaked in 2014 (133 sets), with higher number of individuals noted in 2010, 2011, and 2014 (Figure 54). Moving average trendlines were added to the figure to relay the trend considering multiple peaks and valleys within the data. Overall, the trend suggests that the number of sets remains fairly constant across all years, with a higher number of interactions noted early on in the data and a spike when the number of sets that interacted with the species essentially doubled (i.e. 2014). We also looked at the trends in catch per unit effort (CPUE) over time (Figure 55). Given the short length of the time series, the trends were not statistically significant and the 95% confidence intervals do not preclude the possibility of increasing trends. However, the mean trends suggest an overall mean decline of 7.9% per year (95% CI: -24% to 8.2%; p=0.29) with the free sets CPUE declining at a mean of 12.6% per year (95% CI: -30% to 5.2%; p=0.14). Since these are long-lived species, declines in the observed interaction rate suggest declines in the subpopulations with which this fishery interacts. As noted in both the *Status of Listed Resources* and *Environmental Baseline* sections, Tremblay-Boyer and Brouwer (2016) showed increasing trends in giant manta ray CPUE for all WCPO purse seine fisheries (United States and international) from 2005 to 2016; however, because the species code for giant manta ray was not routinely used by observers in these fisheries until about 2011 (NMFS 2019c) they caution that the increases are attributable to increases in compliance with reporting using the giant manta ray species code cannot be considered an index of population change. Therefore their results do not conflict with those shown in Figure 55.

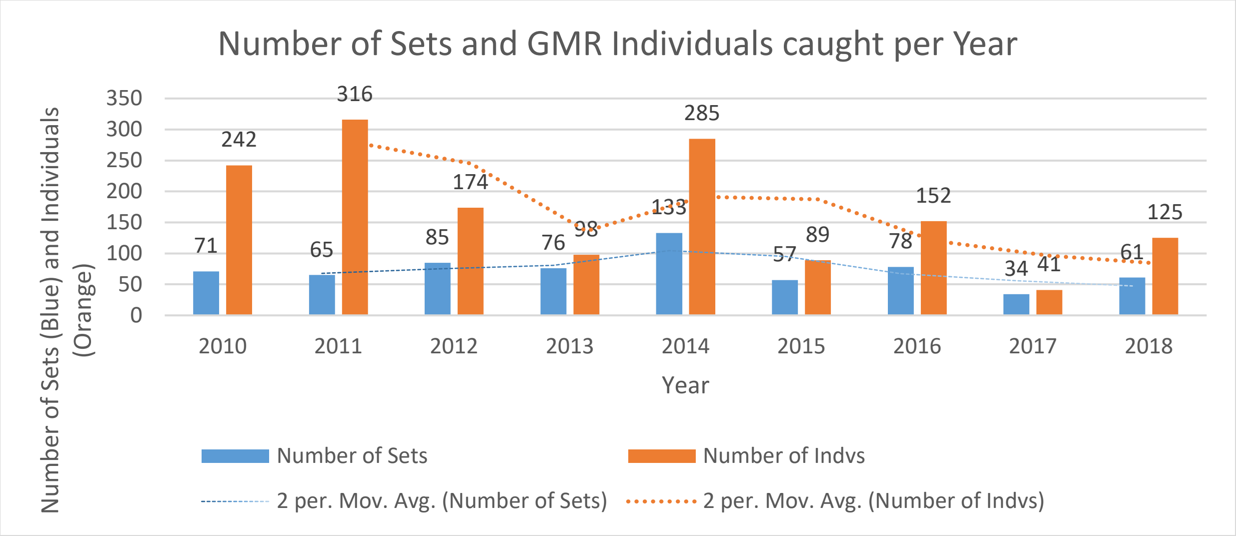


Figure 54. Number of sets (blue) that interacted with individual rays identified by observers as giant manta rays and the number of individuals (orange) observed caught in those sets in the purse seine fishery from 2010 to 2018.

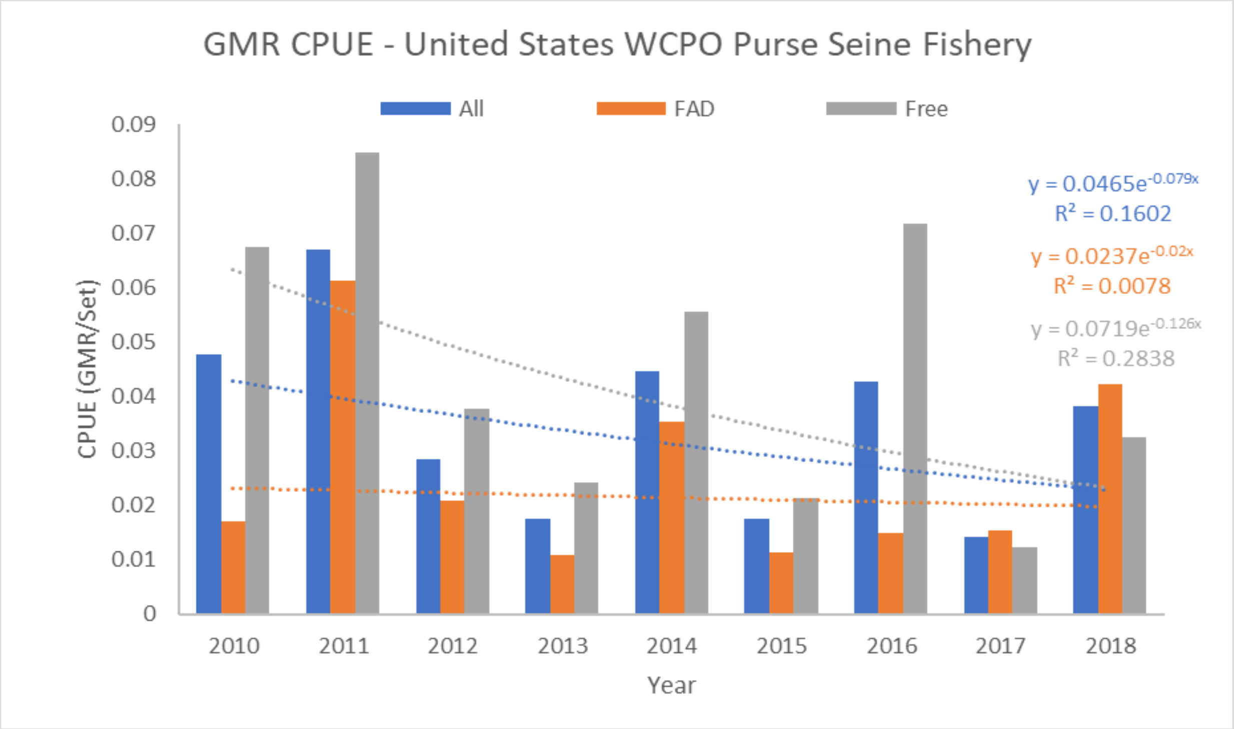


Figure 55. Catch per unit effort (CPUE) for observed giant manta ray captures in the United States WCPO purse seine fishery for all sets (blue circles), FAD sets (orange circles) and free sets (gray circles). Dashed lines are best-fit exponential regressions where the exponent is the percent per year change in the trend.

When comparing the observed number of individuals to the estimated number of total interactions, inclusive of the unidentified category, the data show an increasing trend in interactions until a peak in 2012, with a marked decline afterwards (Figure 56). Observed interaction values were shown in orange in both figures to show these are the same data to help relay trends. The Bayesian model predicted a larger number of interactions in 2012 based on the higher number of unidentified records which occurred in that year.

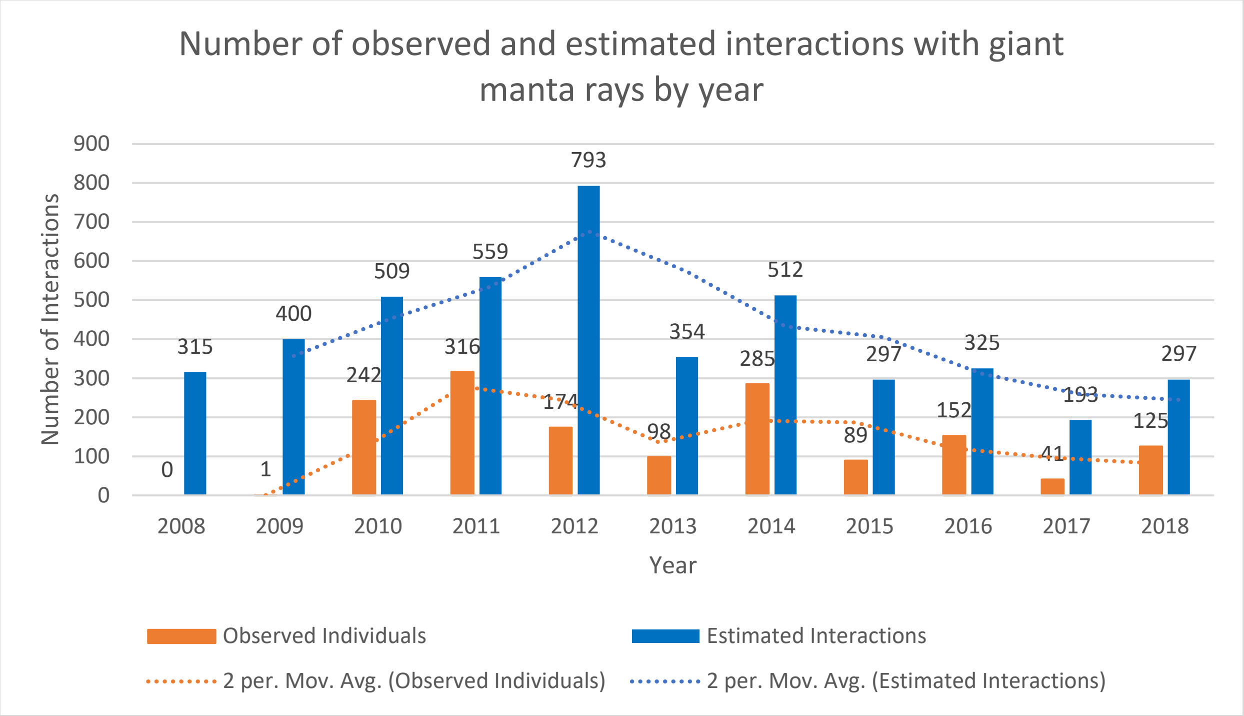


Figure 56. Number of observed giant manta ray interactions (orange) and estimated interactions with unidentified prorated for the species (blue) from 2008 to 2018.

When looking at all sets from 2010 to 2018 by month, the number of observed sets and the number of observed interactions increases throughout the year peaking in November and December (Figure 57). October had the highest proportion of individual mantas caught per set with an average of 4 per set across all years. Two slight depressions are noted in the number of observed interactions in May, and again in July and August. However, at this time it is not apparent what affects these reductions in catch. A moving average trendline was added to show the general trend of the data with multiple peaks and valleys across the overall data. Nowlis (2021) conducted a separate analysis of the United States WCPO purse seine fishery observer data, in this case pooling observed captures across all Mobulidae species codes and separating them into large (greater than 60 kg) and small (less than 60 kg) individuals based on weights recorded by observers. He found that all sizes of mobulids were more likely to be captured closer to the equator, farther east, and on sets that did not use FADs. He also found that larger mobulids had seasonal patterns similar to those reported in Figure 58, with higher rates of capture from November to February, lower rates of captures from April to July, and intermediate rates of captures in the transition months between those timeframes.

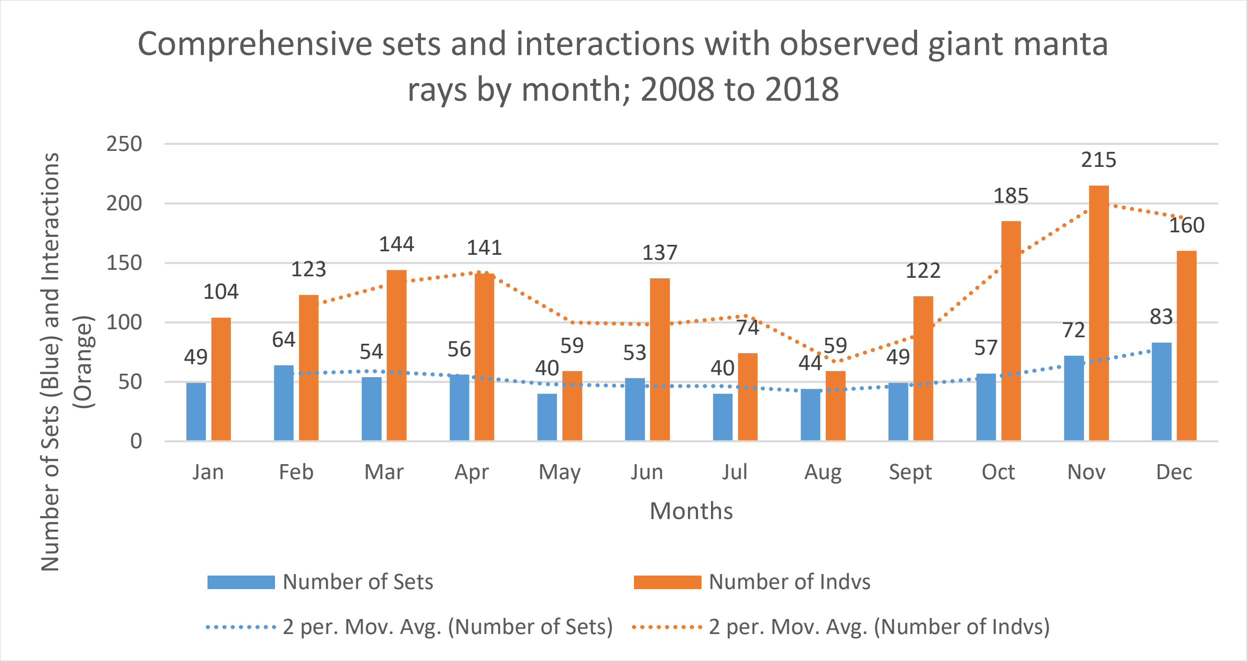
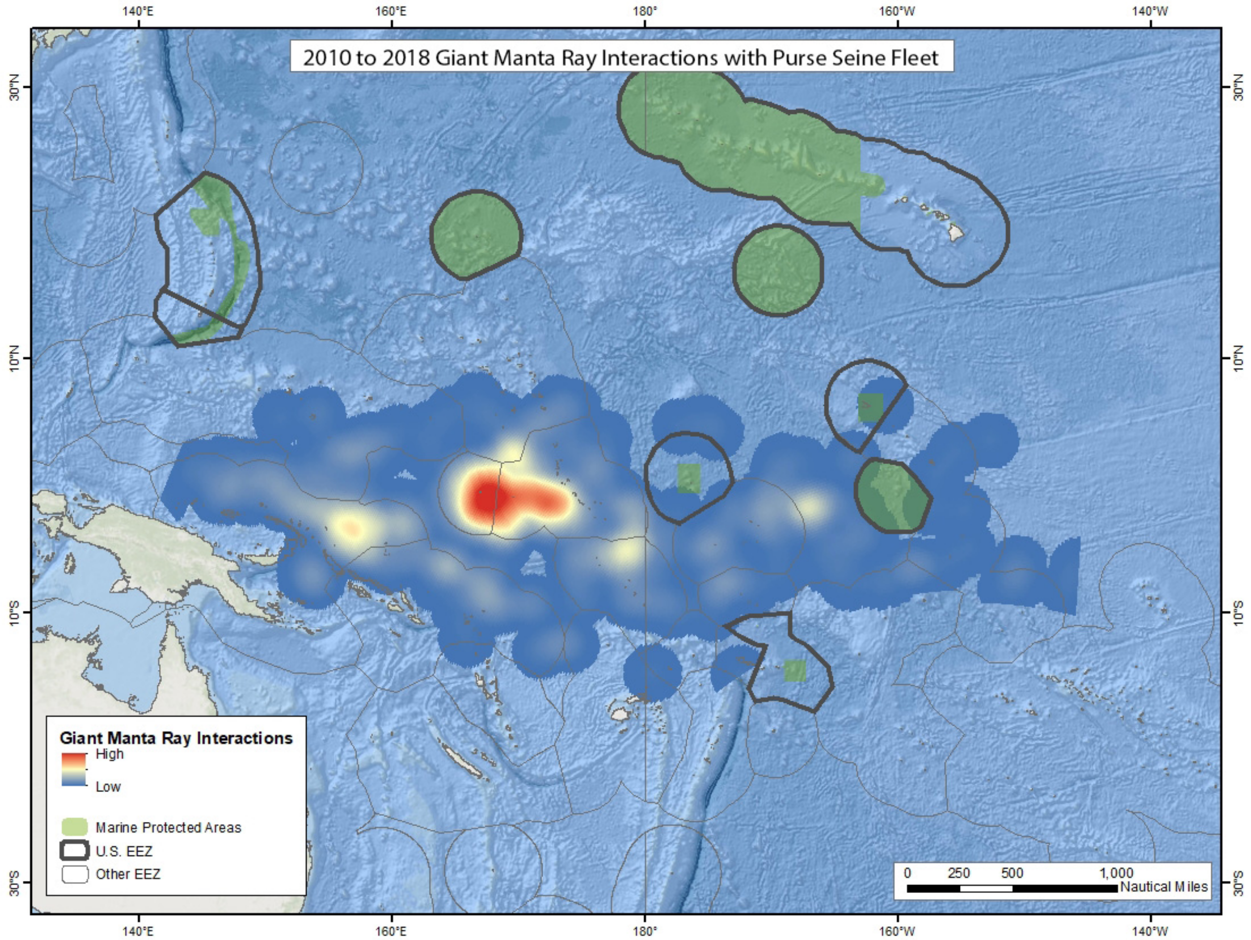


Figure 57. Number of observed sets that interacted with individual rays identified by observers as giant manta rays and the number of observed individuals in those sets by month from 2010 to 2018.

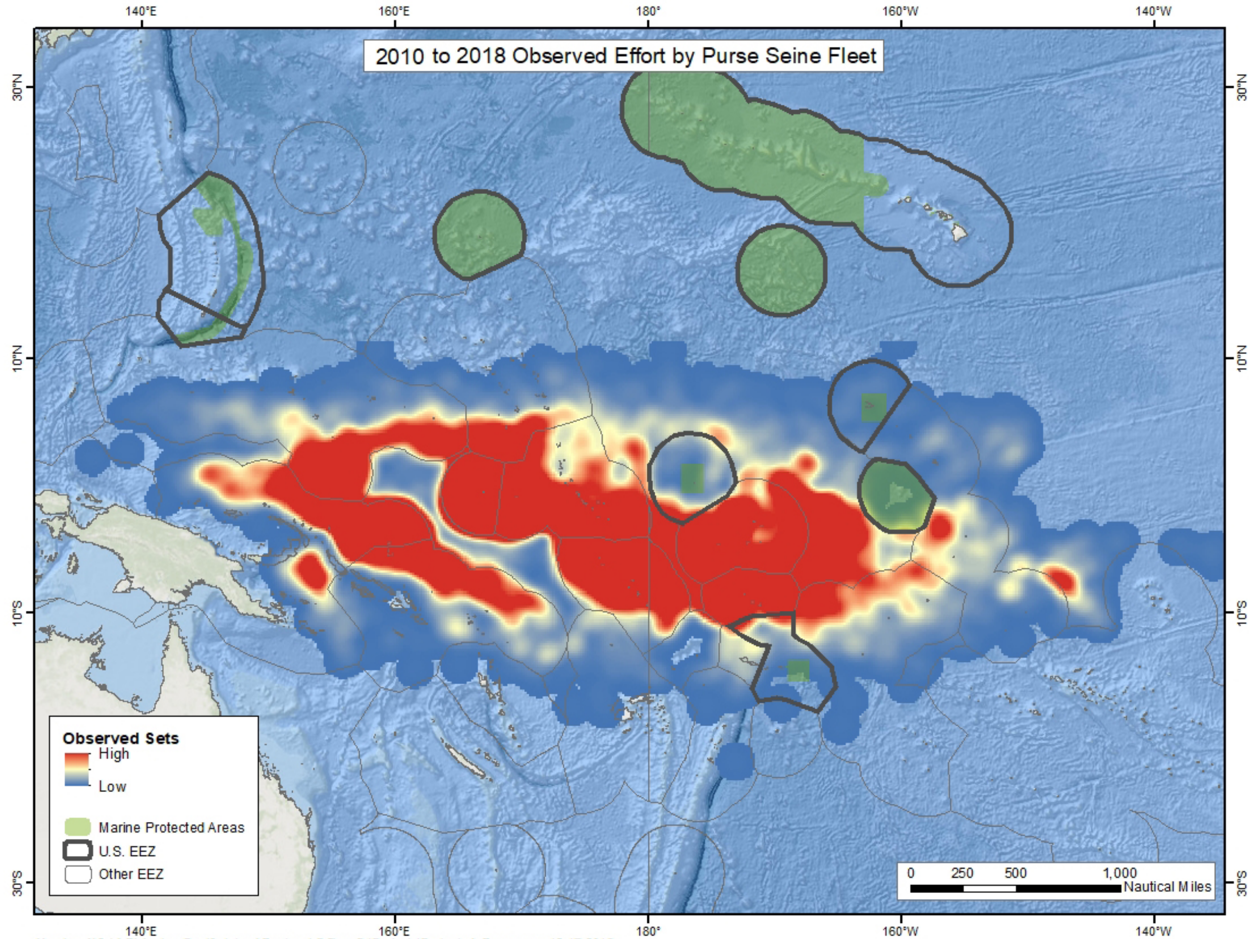
We used the same methods as described for oceanic whitetip sharks to conduct a hotspot analysis for giant manta rays. As with the previous figures, the blue areas have lower interaction or effort rates than those areas with orange or reds. The key was adjusted to illustrate low versus high density values.



Map by: R. O'Conner, NOAA Fisheries, Pacific Island Regional Office, G:\Projects\Protected_Resources, 10-24-2019.

Figure 58. Geographical representation of aggregate observed giant manta ray interactions (sets; *n* = 661) from 2010 to 2018 in the United States purse seine fishery representing the capture of 1,523 individual rays identified by observers as giant manta ray.

Spatially, 43% of all sets that interacted with the individual rays identified by observers as giant manta rays (*n* = 281) occurred around the Republic of Nauru; Tapiwa, Kiribati; and to the west abutting the western coast of South Tawara, Kiribati (Figure 58). This resulted in 790 individuals, or 52% of all interactions (790/1,523*100), being caught in this area when looking at data from 2010 to 2018. Considering only 1.55% of all fishery sets interact with the species, this geographical area where interactions are occurring is considered significant; especially when comparing the interactions versus the total fishing fleet effort (*n* = 42,633 sets; Figure 60 and Figure 60).



Map by: NOAA Fisheries, Pacific Island Regional Office, G:\Projects\Protected_Resources, 12-17-2019.

Figure 59. Total fishing effort of the United States WCPO purse seine fishery between 2010 and 2018 (*n* = 42,633 sets).

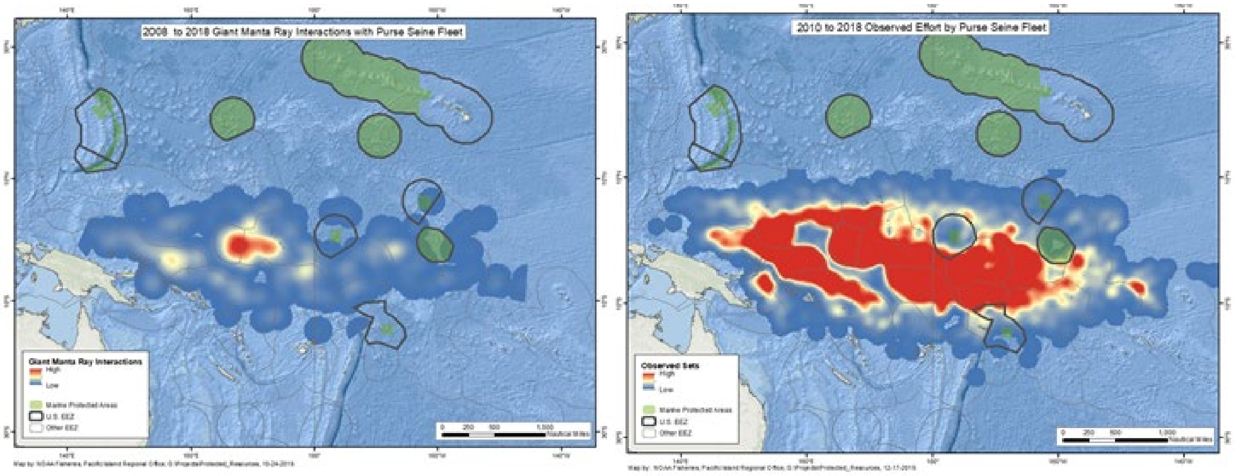


Figure 60. Side-by-side comparison of aggregate observed giant manta ray interactions (*n* = 661 sets; left) with fishing effort (*n* = 42,633 sets; right) from 2010 to 2018.

Across all years, the number of sets that interacted with individual rays identified by observers as giant manta rays peaks in February and December. However, when looking at the number of individuals, peaks occur in November and December (Figure 61). Additionally, a significant peak occurred in 2014 accounting for 86 of the 281 sets that interacted with the species within this hotspot. Assessing 2014 interactions, marked declines in interactions occurred in May and from July to October. The month of December, like in the aggregated data, produced the highest number of sets where interactions occurred for this specific year within the hotspot and accounted for 40% of all interactions for the month when compared to all data. The number of sets that interacted with the mantas in 2014 per month ranged from 2 to 17.

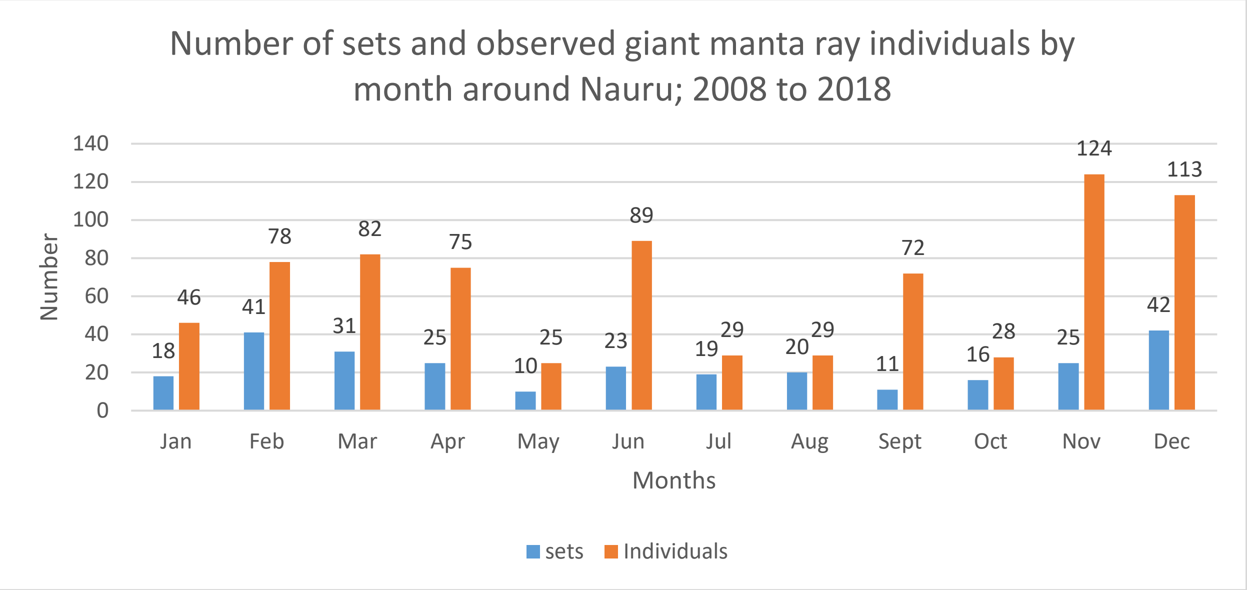


Figure 61. Number of sets and individual rays identified by observers as giant manta rays by month in the hotspot around Nauru and Kiribati from 2008 to 2018.

A second hotspot was identified north of Bougainville, Solomon Islands in the eastern side of PNG's EEZ, which is represented by a substantial portion of sets interacting with giant manta rays. We will refer to this hotspot as PNG East. This location also corresponds to a hotspot seen in oceanic whitetip sharks (see Figure 58), although smaller in scope when comparing the two species. Between 2010 and 2018, 53 sets interacted with observed giant manta rays in this area accounting for a total of 133 individuals (8.01% of all sets that interacted with the species). Fifty-nine percent of interactions had only 1 giant manta present per set, 11% had 2 mantas (*n* = 6), 15% had 3 mantas (*n* = 8), 6% had 4 mantas (*n* = 3), 2% had 5 and 6 individuals per set (2 sets), 4% had 7 or 8 mantas per set (*n* = 2 sets each), and one set had 19 individuals (2%).

Set times ranged from 4:51 am to 17:55 pm (5:55 pm) and of those 53 sets, 23 were before noon (12 pm) with 20 after noon. Temporally, when assessing all interactions by month (Figure 62), the data show a small peak in February with a lower number of sets in the remaining months.

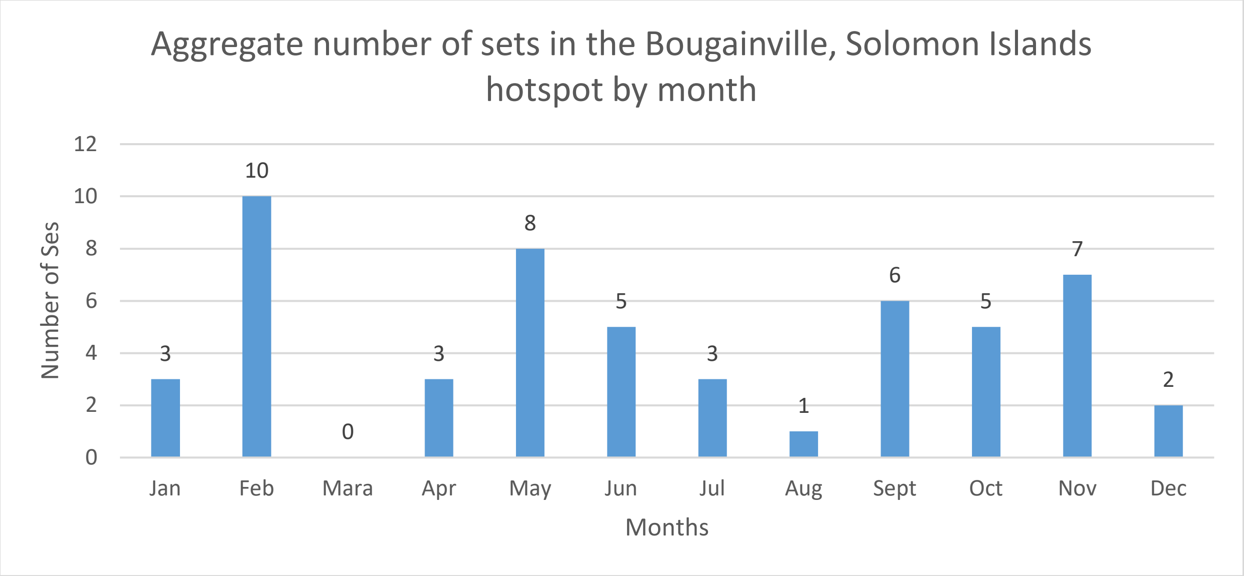


Figure 62. Aggregate sets (*n* = 53) per month from 2010 to 2018 at the second identified hotspot for observed giant manta rays north of Bougainville, Solomon Islands in the eastern portion of PNG's EEZ (PNG East).

When assessing the data by year, the majority of interactions with observed giant manta rays occurred in 2010 and 2011 accounting for the majority of individuals peaking at 65 individuals (Figure 63). In 2010 interactions occurred from September through January, with all interactions in 2011 occurring in February (*n* = 10) and October (*n* = 5). Since then, the number of sets which interacted with the species, and the number of individuals captured per set has decreased. In 2013 a comparable number of sets interacted with the species in this geographical area with a reduced number of individuals per set and when looking at the aggregate total for the year. These data suggest local giant manta ray subpopulations in this area may have declined. However, we also note that all sets in this area (those that interacted with giant manta rays and those did not) have also declined over time (Figure 64) and while the CPUE appears to have declined, this is primarily driven by two years with anomalously high catch rates (2011 and 2014) and without those CPUE appears stable (Figure 65).

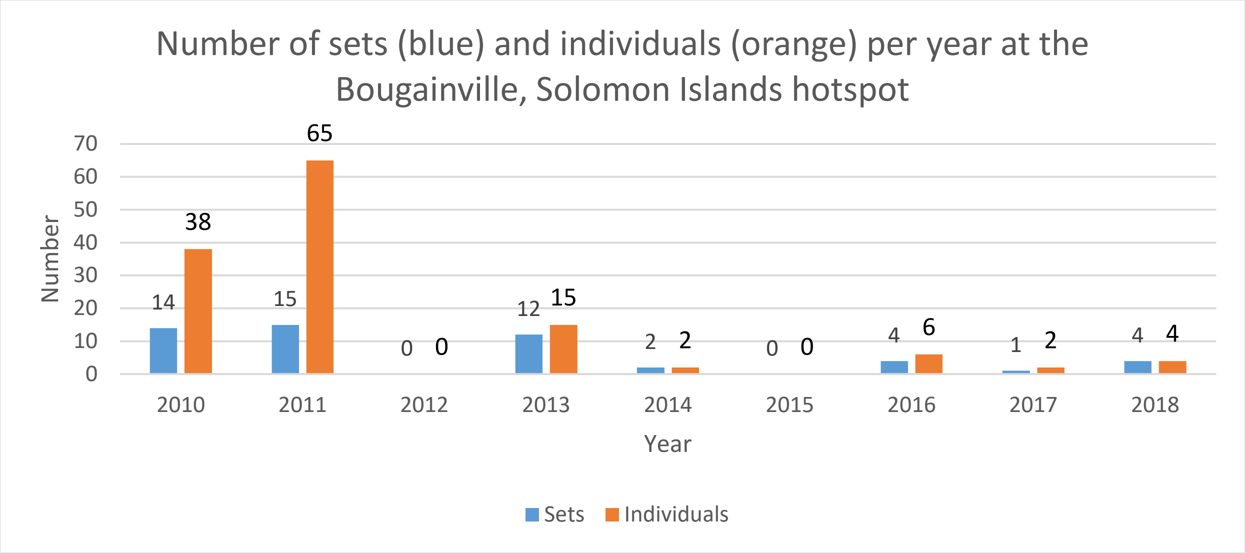


Figure 63. Number of sets and interactions with observed giant manta rays per year from 2010 to 2018 at the second hotspot north of Bougainville, Solomon Islands in the eastern portion of PNG's EEZ (PNG East).

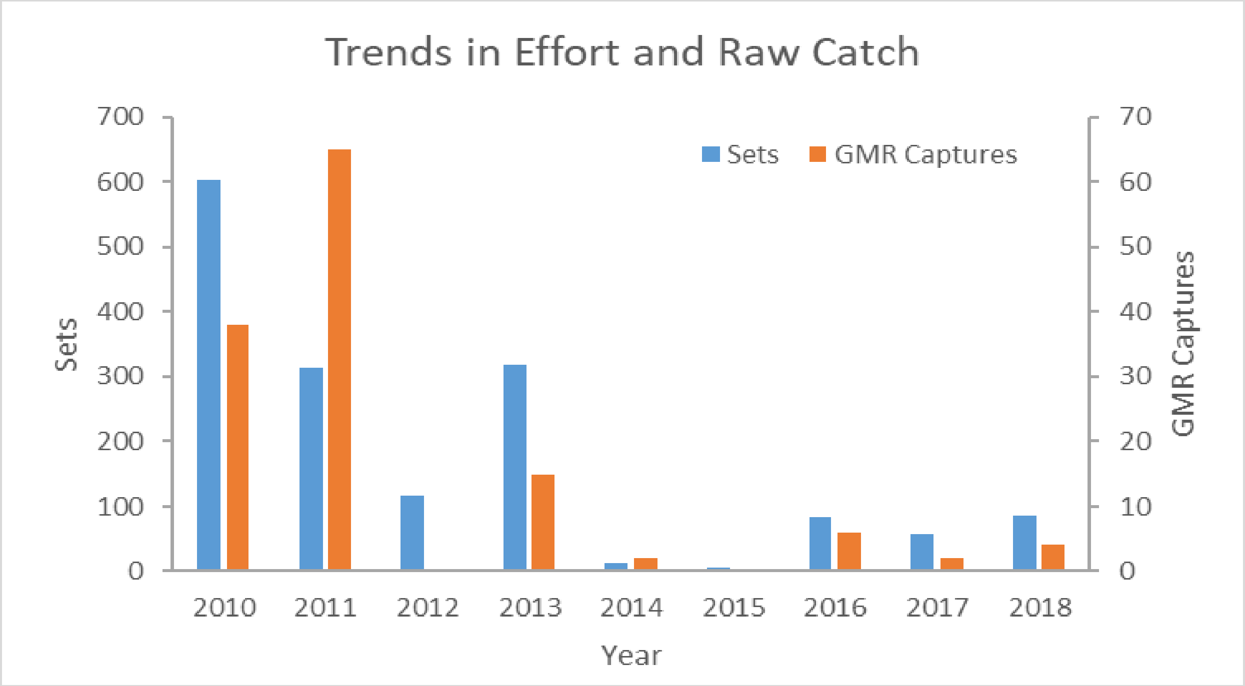


Figure 64. Total number of sets and interactions with observed giant manta rays per year from 2010 to 2018 at the second hotspot north of Bougainville, Solomon Islands in the eastern portion of PNG's EEZ (PNG East).

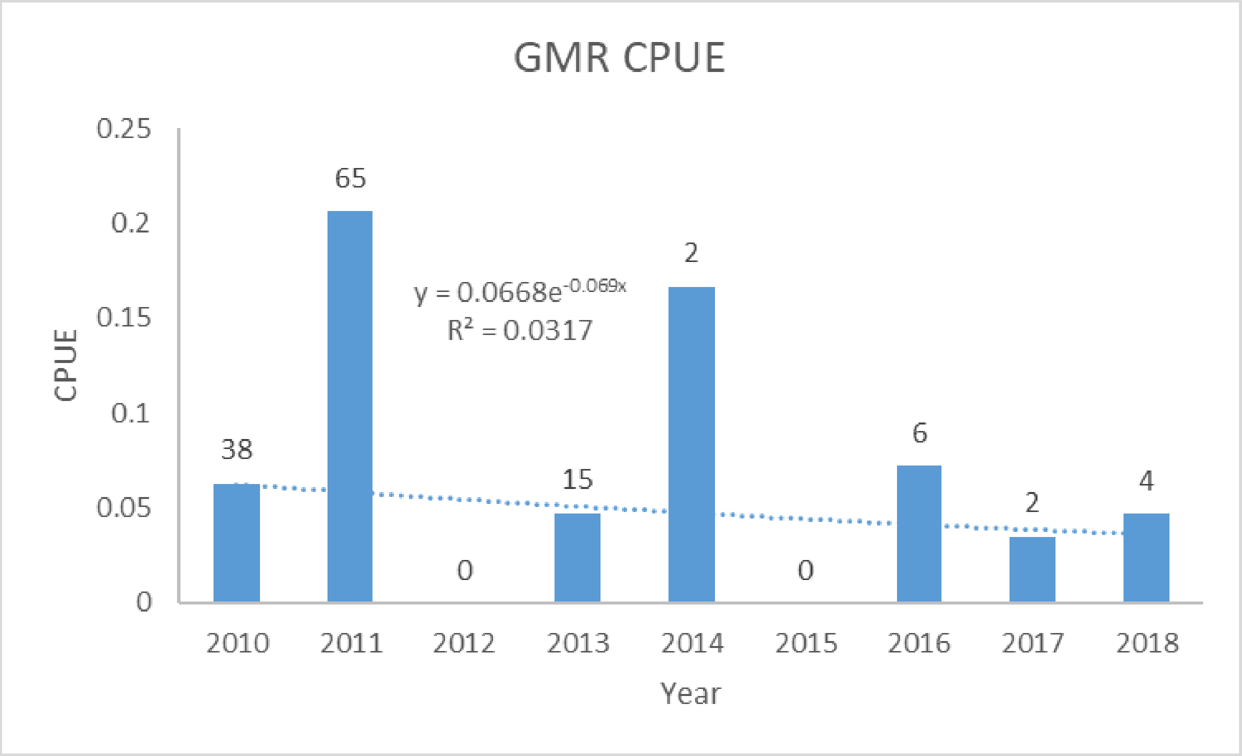
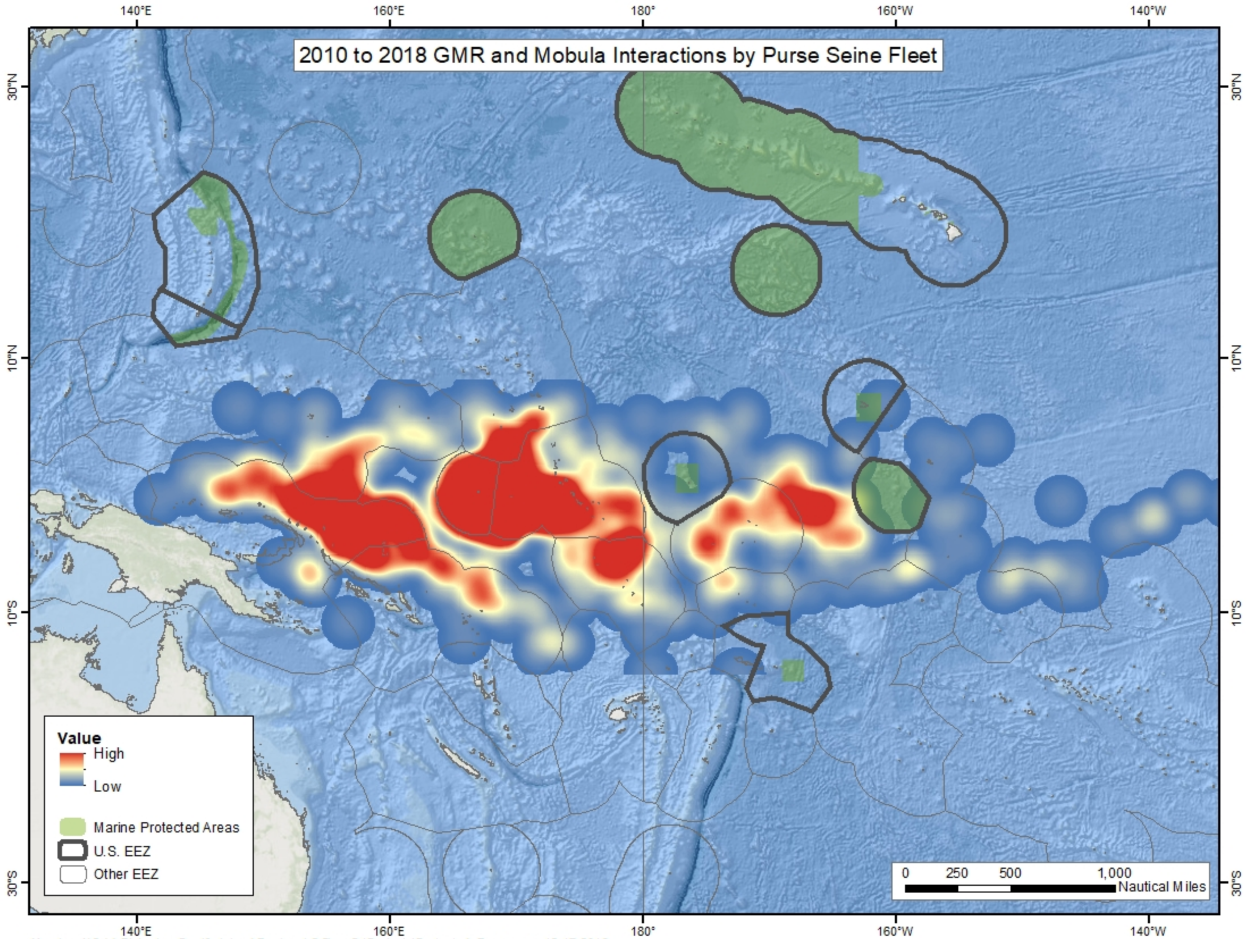


Figure 65. Catch per unit effort for observed giant manta rays from 2010 to 2018 at the second hotspot north of Bougainville, Solomon Islands in the eastern portion of PNG's EEZ (PNG East). Numbers above bars indicate total number of giant manta ray captures.

While giant manta rays are known to aggregate around bathymetric features such as seamounts and ridges (Kashiwagi et al. 2011), bathymetric features did not correlate to the observed interactions in our analysis. However, considering this fishery may interact with mostly juveniles (if identifications were correct and not other smaller species of rays), then these areas may be important for early life stages.

When all unidentified records of *Mobula* nei were overlaid with observed interactions (Figure 66), the spatial analysis becomes more compelling and delineates the hotspot(s) with more clarity, as well as determines other areas with high densities of interactions between the fleet and mobulid species within the *Action Area*. Geographical separation was also noted in interactions from the main hotspots (Nauru and PNG East) by 123 nm from the linear band extending southeast to northwest along the Solomon Islands consistent with the high seas closure areas seen in the fishery effort map (Figure 59). This analysis shows important geographical areas (as identified by the kernel density analysis) where rays are being caught in this fishery and clarifies that it is not just a reflection of where fishing effort is occurring (Figure 59). However, not all of those individuals are likely giant manta rays. Our proration method suggests that approximately 37% of unidentified mobulids would be giant manta rays, and we anticipate that up to 75% of individuals identified as giant manta ray were misidentified. Thus, this heatmap must be interpreted with caution in terms of impacts to giant manta ray. When compared to oceanic whitetip sharks and fishing effort, *Mobula* species appear to be caught on the west side of the *Action Area* with higher densities of interactions. Specifically, west of 180° longitude, densities of interactions with all mobulids species is much more consistent with the fishery effort compared to east of 180° longitude where densities of mobulid interactions are generally lower and more separated. Furthermore, while the densities of interactions for both captures identified as giant manta rays and oceanic whitetip sharks in the Nauru EEZ are spatially similar, in Kiribati's middle EEZ, manta are captured more in the north and east, whereas oceanic whitetips are caught in the south of the EEZ. The records of unidentified *Mobula* nei out number observed giant manta rays in this area such that this may not represent the distribution of giant manta rays.



Map by: NOAA Fisheries, Pacific Island Regional Office, G:\Projects\Protected_Resources, 12-17-2019.

Figure 66. Aggregate heat map of observed giant manta ray interactions and all *Mobula* nei (unidentified records) from 2010 to 2018 in the United States WCPO purse seine fishery.

We also looked at the total number of observed sets from 2008 to 2018 (*n* = 661) and the number of sets versus the number of individuals per set. Table 54 provides a summary of the total number of observed sets and observed individuals that interacted with the fishery. Out of 661 sets that interacted with observed giant manta rays during this time frame, 462 sets had only one individual documented per set (69.79%; 462/661*100), 85 sets had two individuals pet set, 34 sets had 3 individuals, 19 sets had 4 each, 15 sets had 5 animals, and 24 sets had between 6 and 9 individuals a set. More significantly, out of 661 sets, 21 sets had greater than 10 giant manta rays per set (3.18% of sets that interacted with the species; 21/661*100). However, these 21 sets accounted for 453 individuals, that is 29.74% of the total number of interactions with the species in this fishery from 2008 to 2018 (453/1,523*100)). Trujillo-Cordova et al. (2020) presents aerial photographic evidence that the species aggregates in large numbers and documented 125 individuals in a single location.

Table 54. The number of observed sets by the number of observed individuals per set, and the percent of the total sets and total individuals caught from 2008 to 2018 in the United States WCPO purse seine fishery.

Categorized Number of Individuals per Set	Total Number of Sets which Encountered Number of Individuals	Percent of the Total Sets	Number of Individuals	Percent of Total Individuals
1	462	70%	462	30%
2	85	13%	170	11%
3	34	5%	102	7%
4	19	3%	76	5%
5	16	2%	80	5%
6 to 9	24	4%	180	12%
>10	21	3%	453	30%

Potential Spatial Structuring of Giant Manta Rays in the Action Area

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. At a minimum, evidence suggests that giant manta ray in the Atlantic and Indo-Pacific Oceans are independent of one another. However, most studies indicate that, giant manta ray likely occur in much smaller independent demographic units that are important to their viability (Lewis et al. 2015; Stewart et al. 2016a; Moazzam 2018; Beale et al. 2019.

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, personal communication cited in Beale et al. 2019).

The number of giant manta ray subpopulations exposed to the United States WCPO purse seine fishery and the size of those subpopulations is unknown; however, a first and key step to assessing the risk to listed species from an action or threat is describing the population structure, and the viability of the populations that comprise the species. Absent empirical observations of trends and behavior, we would rely on proxy indicators like genetics, demographics and habitat use to identify populations (McElhaney et al. 2000). For some species, like sea turtles this is easier to do than for other species because we rely, in part, on nesting beach aggregations to delineate independent demographic units. For marine pelagic species, it is much more difficult to identify proxies that can assist in identifying putative population boundaries. As noted in the *Status of Listed Resources* Section 3.3.11, since there is a preponderance of growing evidence that giant manta rays likely occur in small subpopulations, and to be viable the abundance of each subpopulation likely needs to be at least 1,000 individuals (Frankham et al. 2014), we used a combination of information from the literature and the distribution of observed giant manta ray captures by the United States WCPO purse seine fishery, the ASLL fishery, and the Hawaii-based deep set longline fishery to approximate the number of giant manta ray subpopulations that may occur in the *Action Area* and may be affected by the fishery.

Based on our analysis we estimate that purse seine fishery interacts with 19 subpopulations of giant manta rays (Table 56). Known giant manta ray subpopulations are found in oceanic waters near productive coastlines, continental shelves, offshore pinnacles, and 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 feature such as an island or seamount. We recognize this approach is not ideal and does not incorporate the possibility of misidentification of giant manta ray in the observer databases, however, given the lack of information on the distribution of giant manta ray in the *Action Area*, we consider this the best approach available. 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. Note that the subpopulations Nauru and Kiribati, Gilbert Islands coincide with the main hotspot while Papua New Guinea East (PNG East) coincides with the secondary hotspot detected in Figure 58.

As noted in the *Status of Listed Resources* section, several studies have tracked individual giant manta rays and provide information on the spatial extent of giant manta ray populations. We used ArcGIS to estimate the longest straight-line distance of the area of occupancy from maps presented in Hearn et al. (2014) and Stewart et al. (2016a) because these publications represent the best available information to illustrate the range of distances that can be expected for subpopulations with the *Action Area*. The resulting values ranged from 457 km in Bahia de Banderas, Mexico to 590 km in Isla de la Plata Ecuador (Figure 67). Therefore, a circle with a minimum diameter of approximately 600 km or a radius of 300 km would encompass these subpopulations. However, as this is a small sample size of studied subpopulations, we increased the radius to 500 km in order to have a higher level of confidence that we would encompass all fishery captures that are impacting specific subpopulations. Between 5% (Stewart et al. 2016a) and 11% (1 out of 9 tagged animals; Hearn et al. 2014) of individuals can be expected to be outside of the core area of occupancy but would be encompassed within a 1,500 km radius. Again, given the low sample size of the studied subpopulations, we assumed that up to 15% of individuals belonging to a specific subpopulation would be between 500 and 1,500 km away from the center point of the aggregation. Therefore, to estimate the number of captures by the United States WCPO purse seine fishery that impact individual subpopulations, we created 500 km radius circles around the center points of the subpopulations identified in Table 56 (Figure 68) and assumed that all captures within that circle impacted that subpopulation (note that some captures are assigned to more than one subpopulation). We then created 1,500 km radius circles around the center points of the subpopulations and assumed that 15% of all individuals between the 500 km circle and the 1,500 km circle impacted that population. The sum of these two values represent the total number of individuals captured from the individual subpopulations (Table 56).

As noted in Section 1.1; *Consultation History*, the action agency provided supplemental information on the characteristics of the United States WCPO purse seine fishery that are likely to occur for the reasonably foreseeable future. Two key characteristics are 1) because there are currently only 20 vessels in the fleet and no new vessels are anticipated to enter the fleet in the near future; and future effort will be lower than the action agency's initial estimate of 7,200 sets per year to 3,100 sets per year and 2) a substantial eastward shift in the expected spatial distribution of the fishing effort is anticipated. Specifically, IFD suggests the change in distribution specified in Table 55.

Table 55. Expected current/future distribution of fishing effort for the United States WCPO purse seine fishery based on the vessels that remain in the fleet.

Block	**1**	**2**	**3**	**4**
Longitude Range	**135° E to 160° E**	**160° E to 175° E**	**175° E to 175° W**	**175° W to 130° W**
Expected distribution	7%	20%	27%	45%
2008-2018 distribution (from IFD)	19%	33%	22%	26%
2010-2018 distribution (PRD)	15%	33%	22%	30%
Adjustment multiplier	0.456	0.610	1.208	1.525

Within these blocks, IFD anticipates the overall spatial distribution of the fishery will remain the same. In other words, a grid that has historically contained high fishery effort will continue to receive high fishery effort relative to the block it is in, but the total effort within each block will follow that proposed in Table 55. In order to assess the number of giant manta ray from each subpopulation anticipated to be captured based on this new distribution, we conducted the following analysis:

1. Bin all sets from 2010 to 2018 into 1° by 1° bins.
2. Determine the percent of sets that occurred in each bin then sum those percents within each block (Table 55).
3. Adjust for the new distribution by dividing the expected percent distribution by the 2010 to 2018 percent distribution to get a multiplier for each block (Table 55)
4. Within each 1° by 1° bin, multiply the percent of sets that occurred in each bin from 2010 to 2018 by the multiplier calculated for the block in which the bin occurs. This gives the redistributed effort in terms of the percent of total sets anticipated to occur in each bin and block.

5. Multiply the percents in each 1° by 1° bin by the number of anticipated annual sets (3100) to get anticipated sets per year per bin.
6. Bin all observed giant manta ray captures from 2010 to 2018 into 1° by 1° bins.
7. Divide observed giant manta ray captures from 2010 to 2018 in each bin by the total sets per 1° by 1° bin (from step 1) to get an interaction rate per bin.
8. Multiply the interaction rate per bin by the anticipated sets per bin (from step 5) to get anticipated giant manta ray captures per bin. This only accounts for individuals identified as giant manta ray so we must account for the proportion of unidentified mobulids that are expected to be giant manta ray.
9. Similar to step 2 for sets, we use the results from step 8 to determine the percent of total giant manta ray captures for each bin by dividing giant manta ray captures per bin from step 8 by the sum of giant manta ray captures across all bins.
10. Multiply the percent of giant manta ray captures per bin by the anticipated annual giant manta ray captures (step 11 below) to get anticipated captures per bin.
11. Anticipated annual giant manta ray captures: Using the interaction rates for giant manta ray and unidentified mobulids along with the proportion of unidentified mobulids anticipated to be giant manta ray as previously calculated for the fishery, we anticipate a mean of 165 (95th percentile 199) giant manta ray captures per year without accounting for any possible misidentification. Again using the same method as was previously applied, if we assume a 75% misidentification rate, the anticipated annual captures are a mean of 45 (95th percentile 50).
12. For bins that fall within the 500 or 1500 km radius of subpopulations, sum the anticipated captures per bin across all bins within each subpopulation, assume 100% of captures within 500 km and 15% of captures between 500 and 1500 km (as described above) are the total captures anticipated for each subpopulation (Table 56).

Table 56. Putative giant manta ray subpopulations likely to be impacted by the United States WCPO purse seine fishery. The number of observed giant manta ray and the number of anticipated future captures per year impacting individual populations were calculated as described in the text.

Giant Manta Ray Subpopulation	Center Point Latitude	Center Point Longitude	Observed Giant Manta Ray Captures	Anticipated Annual Exposures 5-yr Max RA (95th percentile)	Source
Nauru	-0.596	166.893	597	64 (69)	Inferred from capture locations, centered on the island of Nauru
Kiribati, Gilbert Islands	1.454888	172.9776	470	57 (61)	Inferred from capture locations, centered on the island of Tarawa
Papua New Guinea East	-4.472	159.431	320	12 (12)	Lawson et al. 2017; center location inferred from capture locations
Solomon Islands	-9.509	163.877	241	21 (23)	O'Malley et al. 2013, Lawson et al. 2017
Tuvalu	-6.084	177.367	230	39 (40)	Inferred from capture locations, centered on the island of Tuvalu
Papua New Guinea West	-1.465	149.665	147	33 (35)	Lawson et al. 2017; center location inferred from capture locations
Pohnpei FSM	6.876	158.245	120	11 (13)	O'Malley et al. 2013, capture locations

Giant Manta Ray Subpopulation	Center Point Latitude	Center Point Longitude	Observed Giant Manta Ray Captures	Anticipated Annual Exposures 5-yr Max RA (95th percentile)	Source
Kiribati Phoenix Islands East	-4.713	-174.376	83	17 (18)	Inferred from capture locations, centered on seamount locations from Macmillan-Lawler and Harris (2016)
Kiribati Phoenix Islands West	-3.032	-167.468	74	19 (21)	Inferred from capture locations, centered on seamount locations from Macmillan-Lawler and Harris (2016)
Papua New Guinea	-8.624	145.17	44	4 (4)	O’Malley et al. 2013
Fiji	-14.115	179.021	35	6 (6)	O’Malley et al. 2013
American Samoa	-14.2316	-169.463	18	4 (4)	Observed captures in the ASLL fishery (NMFS unpublished data)
Kiribati, Line Islands	1.897469	-157.424	17	5 (6)	O’Malley et al. 2013, centered on the island of Kiritimati (Christmas Island)
Palmyra	6.452	-162.345	14	3 (3)	Observed captures in the Hawaii-based deep-set longline fishery (NMFS unpublished data)
Cook Islands	-13.026	-163.608	13	3 (3)	O’Malley et al. 2013
New Caledonia	-21.365	167.44	8	1 (1)	O’Malley et al. 2013, Lawson et al. 2017
Yap FSM	9.55968	138.1251	5	1 (1)	O’Malley et al. 2013, capture locations

Giant Manta Ray Subpopulation	Center Point Latitude	Center Point Longitude	Observed Giant Manta Ray Captures	Anticipated Annual Exposures 5-yr Max RA (95th percentile)	Source
Johnston Atoll	13.121	-165.782	2	0 (0)	Observed captures in the Hawaii-based deep-set longline fishery (NMFS unpublished data)
French Polynesia	-9.43034	-139.499	1	0 (0)	O'Malley et al. 2013, Carpentier et al. 2019

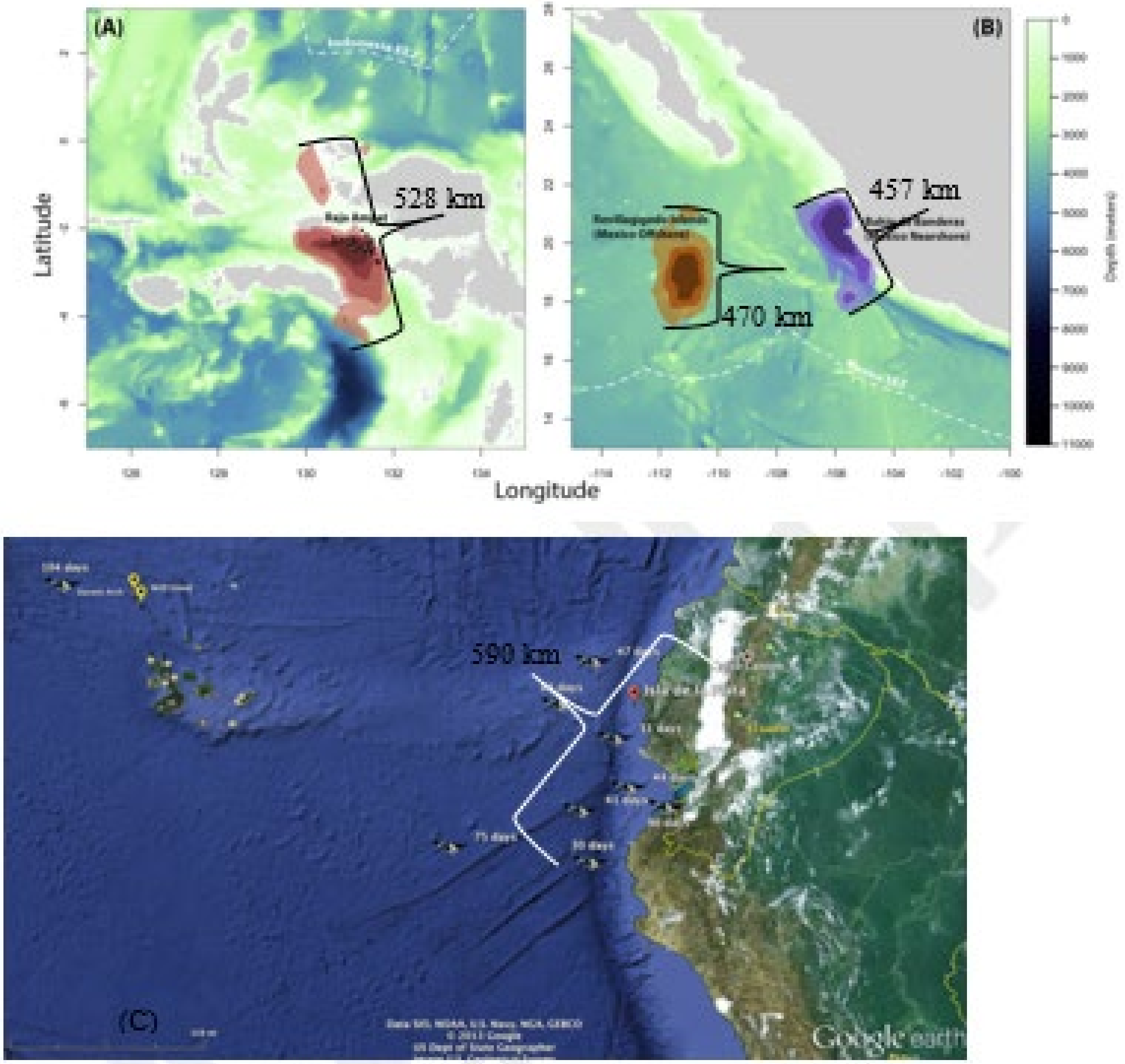


Figure 67. Extent of giant manta ray subpopulations based on tracking tagged individuals. Black brackets (white brackets in C) highlight the longest dimension of the area of occupancy with the length noted in kilometers (km). Locations: A) Raja Ampat, Indonesia (Stewart et al. 2016a); B) Revillagigedo Islands (left) and Bahia de Banderas (right), Mexico (Stewart et al. 2016a); C) Isla de la Plata, Ecuador (Hearn et al. 2014).

As described above, we estimate that the action affects 19 giant manta ray subpopulations. We would anticipate that these are discrete subpopulations with little or no movement between them (Lewis et al. 2015; Stewart et al. 2016a; Moazzam 2018; Beale et al. 2019). However, there is a relatively continuous distribution of giant manta ray captures throughout most of the *Action Area*, which may suggest that the subpopulations are not discrete. Three possible explanations for the relatively continuous distribution of captures are: (1) there are more giant manta ray subpopulations throughout the WCPO that have not been identified, 2) a metapopulation may exist within the *Action Area* that consists of multiple subpopulations that may have some degree of movement between them; or (3) a portion of the individuals were misidentified as giant manta rays and are another *Mobula* species; reef manta rays appear to have relatively continuous distribution throughout the *Action Area* (Lawson et al. 2017). As noted above, studies indicate that giant manta rays likely have discrete subpopulation structure with limited movement between subpopulations (Lewis et al. 2015; Stewart et al. 2016a; Moazzam 2018; Beale et al. 2019), and we focus our analysis on the 19 putative subpopulations in the *Action Area*. We account for any uncertainty in this assumption by addressing each of the three explanations defined above in our analysis of the impacts of the United States WCPO purse seine fishery on giant manta ray subpopulations within the *Action Area*. While the fishery may have interacted with up to 2 individuals from the subpopulation centered around Johnston Atoll and up to 1 individual with the subpopulation centered around French Polynesia from 2010 to 2018 (Table 56), these individuals only occurred in the 500 to 1,500 km area, and assuming only 15% of those individuals can be assigned to a subpopulation results in less than 0.5 individuals assigned to each of those subpopulations. Therefore, we estimate zero interactions will be ascribed to these to subpopulations in the future. Therefore, the scenarios presented below address future anticipated captures for the remaining 17 putative subpopulations of giant manta rays.

Scenarios to Understand the Potential Impacts of Fishery Interactions on Giant Manta Ray Population

To overcome the absence of information on the number of manta ray subpopulations that are likely to be exposed to the United States WCPO purse seine fishery, the size of those subpopulations, their connectivity, and identification uncertainty, we developed several reasoned scenarios to assess the probable effects of the United States WCPO purse seine 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. Maximum intrinsic population growth rates (r_{max}) for giant manta rays have been estimated at 0.019 (reported mean, Rambahiniarison et al 2018), 0.042 (Ward-Paige et al. 2013; Mardhiah et al. 2019) 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 (Table 57), 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; Table 57), NMFS has concluded that the r_{max} values published by Rambahiniarison et al. (2018) likely represent the best r_{max} value available for this consultation ($r_{max} = 0.019$). We concluded that the r_{max} value published by Ward-Paige et al. (2013; $r_{max} = 0.042$) is also reasonable, but left us with some concerns over the methodology because the density dependent assumptions used by Ward-Paige et al. (2013) to estimate survival to maturity result in somewhat illogical r_{max} values, whereby the lowest adult mortality rate (0.02; Table 57) and highest fecundity (0.36; Table 57) would combined 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 those values. Lastly, we are concerned that the r_{max} estimated by Dulvy et al. (2014) is likely an overestimate because the Euler-Lotka equation that they used is not consistent with other derivations in the literature (i.e. it is missing a term for survival to maturity; Myers et al. 1997; Smith et al. 1998) and may be overestimating r_{max}, which may underestimate fishery impacts to giant manta ray subpopulations. As this is a published value for r_{max} we consider the range of

0.019 to 0.116 in a subset of our analyses, however in our assessment we will rely most on the estimates of 0.019 and 0.042.

Table 57. Values used for adult instantaneous natural mortality rate (M), age at maturity (α), number of female pups per adult female per year (b), survival to maturity (l_α), and longevity (ω) to calculate the maximum intrinsic population growth rate (r_{max}) for giant manta ray. The survival to maturity and longevity terms were not incorporated in the Dulvy et al. (2014) model (NA).

Reference	**M**	**α**	***b***	***l_α***	**ω**	**r_{max}**
Rambahiniarison et al 2018	0.087	8.6	0.25	0.473	40	0.019
Ward-Paige et al. 2013	0.15	9	0.3	0.865	30	0.042
Dulvy et al. 2014	0.012-0.04 (0.02)	8-10 (9)	0.1 to 0.5 (0.36)	NA	NA	0.116

For our analyses, all scenarios considered a range of initial subpopulation 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 considered in our scenarios was designed to capture the range of counts of individuals from CITES (2013) and subsequent abundance estimates based on mark and recapture analyses (Burgess 2017; Beale et al. 2019). While these abundance estimates are primarily from subpopulations that occur outside the *Action Area*, and the methods used to estimate subpopulation size varies somewhat, they represent the best scientific information available to estimate the size of subpopulations in the *Action Area,* and the species writ large. As noted in the *Status of Listed Resources* Section 3, we assume that 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. The smaller population sizes, while in the literature, are likely a result of either incomplete sampling of individuals, or representative of exploited populations that are unlikely to persist. Within each scenario, we projected a future subpopulation abundance over 10, 20 and 40 year using three different published rates of intrinsic growth. Subpopulation abundance in each year was estimated using the equation above and subtracting anticipated United States WCPO purse seine fishery mortalities, assuming constant fishing mortalities regardless of subpopulation size. The annual fishery mortalities for each subpopulation projection were calculated as the future exposures per year per subpopulation (Table 56) multiplied by the 0.96 at vessel mortality rate.

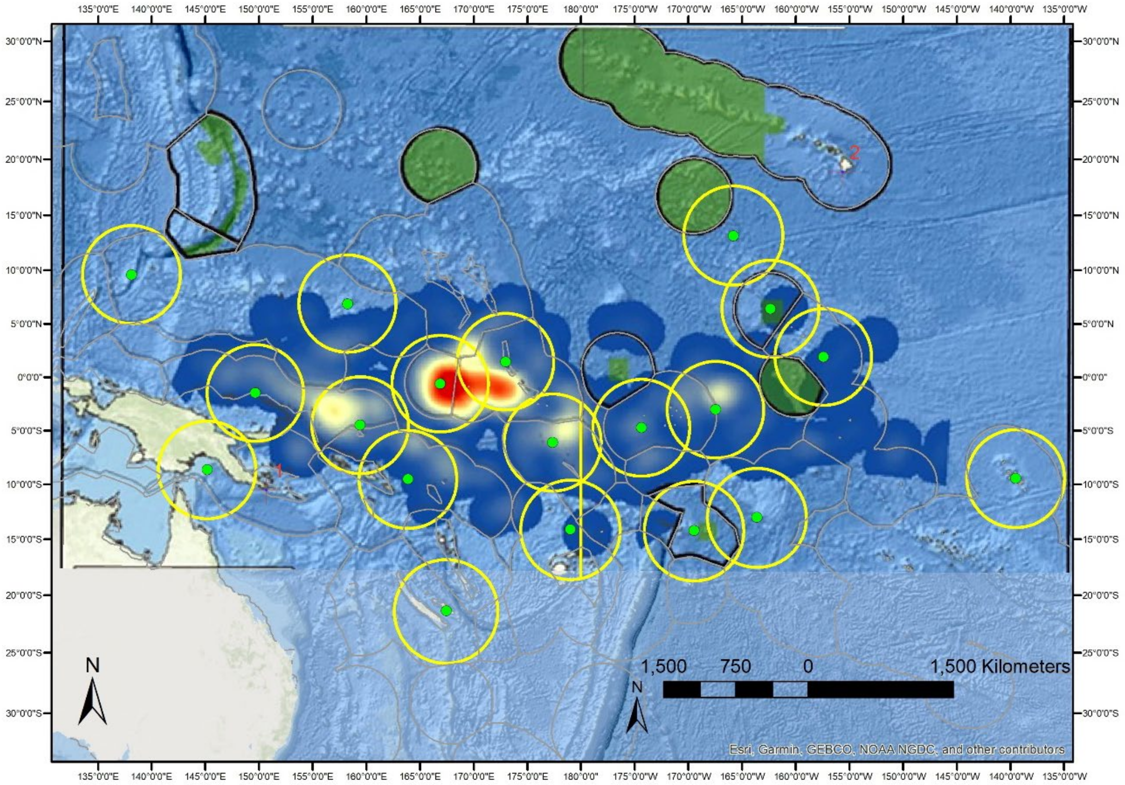


Figure 68. Map of fishing area for the United States WCPO purse seine fishery. Small green circles are the center points of the subpopulations in Table 56. Yellow circles represent an area around the subpopulation center points with a radius of 500 km. The area shaded from dark blue to red is the hot spot analysis for observed giant manta ray captures by the fishery from 2010 to 2018.

Scenario 1 considers the 17 putative subpopulations identified in Table 56 (excluding Johnston Atoll and French Polynesia as no future annual fishery exposures are anticipated for those subpopulations) as discrete subpopulations with no connectivity. Scenario 1 assumes that all individuals identified as giant manta ray were identified correctly (i.e. no misidentification). Subpopulations were projected for 10, 20 and 40 years using the maximum 5-yr running average as annual anticipated future fishery exposures for each subpopulation (Table 56) and initial abundances as described above. This scenario relies on mean exposure because the projections are long enough to include values above and below the mean. Although it is possible for the 95th percentile of exposures to occur repeatedly over these time intervals, it is not likely. Therefore, the running average value strikes a balance between under- and over-estimating the risk over a long time interval.

Scenario 2 is the same as Scenario 1 however here we use the 95th percentile of fishery exposures to understand the worst-case situations. Scenario 2 was also projected for 10, 20 and 40 years. In this and all other scenarios going forward, we eliminated r_{max}=0.116 from the analysis as we consider this value implausible given giant manta ray life history.

Scenario 3 is also essentially the same as Scenario 1; however, in this case we reduce the fishery exposures for each subpopulation (Table 56) by 50% (Scenario 3a) and 75% (Scenario 3b) to account for the effect of species misidentification between giant manta rays and other mobulids. We have several lines of evidence that suggest not all animals identified as giant manta ray were correctly identified. These include:

- There is substantial range overlap among ray species, and we know that there are other ray species like the *M. tarapacana* and *M. alfredi* in the *Action Area* for this consultation, which suggests that mixed ray catches would be more likely than singular species catches particularly for this large *Action Area*.
- Morphological similarities make it difficult to distinguish to species (Stevens et al. 2018). Morphological variation occurs not only among species, but within populations and between populations of the same species occur. For instance, *M. birostris* has a coloration that ranges from melanistic (black) to leucistic (pale) with the most common color variation occurring in between the two (Stevens et al. 2018).
- Based on the catch weights provided by observers, 76% (95% confidence interval= 0.66; 0.87) of observed giant manta ray captures have weight estimates less than known size at birth of giant manta rays (0.0685 mt; Matsumoto and Uchida 2008; Figure 54; although see the caveats associated with that figure). Nowlis (2021) concurs with this assessment and states that the misidentification rate is at least 75%.
- The limited number of other mobulid species besides giant manta ray identified in the observer database meant that 36.9% of unidentified mobulids were assigned to the giant manta ray category, which is likely to be an overestimate.
- FFA provided NMFS with 15 photographs, and based on the date and time stamp IFD believes the 15 photos represent 6 or 7 individual rays. Although we do not know exactly what FFA observers may have cataloged the identification of these species as, the collective consensus of several experts (IFD, PRD, Manta Trust, and others; pers. comm. E. Crigler to A. Garrett, Oct 2020) is that they are not *M. birostris* and are instead, *M. tarapacana* (the Chilean devil ray).
- The IATTC had a manta species identification problem where large numbers of giant manta rays were reportedly captured. We have reason to believe that this may be a substantial issue for the United States WCPO purse seine fishery too. While vessels can request WCPFC-IATTC cross-endorsed observers when they intend to operate in both RFMOs on a single trip (85 FR 66513), however the reporting forms and data collected differ between the two fisheries. After the IATTC improved identification techniques and protocols for rays, reef mantas were found to be caught with higher frequency than giant mantas (Stewart pers. comm to J. Rudolph, 11 May 2020).

While there is strong evidence to suggest that *Mobula* captured in this fishery may not be properly identified to species, we found no information in the observer data that we could use to calculate the magnitude of the problem. We are, however, confident that there is a high probability that the number of individuals recorded by observers as giant manta rays overestimates the number of interactions with giant manta rays. Although it should be interpreted with caution because the weight of *Mobula* species that interact with this fishery are estimated by observers; our analysis of observed and recorded weights suggests that there are a large number of animals in this fishery that are smaller than giant manta rays at birth (specifically a mean of 76% and 95% confidence interval of 66 to 87%).

In the IATTC regions, even after training in correct identification, observers continue to have a misidentification rate of about 25% for giant manta rays (J. Stewart, pers. comm. to M. Snover May 2021). Prior to the training, the misidentification rate was unknown but anticipated to be considerably worse. Based on our weight distribution analysis, the highest proportion of individuals identified as giant manta ray that were larger than 0.0685 mt was in 2015 with 48% (Figure 53) and based on our bootstrap analysis, the mean was 24% (95% confidence interval: 66 to 86%). Similarly, in a separate analysis of the United States WCPO purse seine fishery observer data, Nowlis (2021) suggests that, based on reported weights and the weight distribution of all mobulid species that may interact with the fishery, no more than 25% are likely to be giant manta rays. Therefore, we consider it reasonably likely that the misidentification rate is at least 50%, based on the weight analysis we conducted for this consultation, with 75% as a reasonable upper estimate. Note that we acknowledge the misidentification rate may be higher than 75%, but we consider 75% to be a reasonable upper limit. Therefore, we considered misidentification rates of 50 and 75% as not only plausible, but our best estimates given the evidence, while still being precautionary and providing the benefit of the doubt to the species. Under Scenario 3, subpopulation projections were examed for only 10-years. We opted not to carry this analysis over a longer time frame because it is more appropriate to adjust the data for misidentification prior to developing our estimates for each subpopulation, so we added Scenarios 5 and 6 to our analysis.

In Scenario 4, we incorporate the possibility of movement between subpopulations by considering rates of emigration (individuals subtracted from the subpopulation) and immigration (individuals added to the subpopulation) between 0 and 15% of subpopulation abundance. Above we described that 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, and given the low sample size of the studied subpopulations, we assumed that up to 15% of individuals belonging to a specific subpopulation would be between 500 and 1,500 km away from the center point of the aggregation. We use this same assumed value of 15% (rounded up from 11% based on two studies; Hearn et al. 2014, Stewart et al. 2016a) to estimate the maximum amount of movement expected between subpopulations based on movements described in the literature. We used the range of annual future exposures estimated for individual subpopulations (2 to 200; Table 56). We considered all combinations of values for initial abundance, fishery exposures per year, and intrinsic population growth rates. For emigration rate and immigration rates, we held emigration constant at 0% and ranged immigration from 0 to 15%; conversely, we also held immigration constant at 0% and ranged emigration from 0 to 15%. For this scenario, we were not necessarily modeling individual subpopulations but addressing the general question of whether immigration can sustain small subpopulations in the face of fishery mortalities. Hence, we simply used the range of initial subpopulation sizes as described above and the range of fishery interactions estimated for the 15 subpopulations but to not link these to specific subpopulations.

After conducting and analyzing these scenarios, we thought it prudent to further explore the possibility of up to 75% misidentification issues with giant manta ray in the fishery (Scenario 3b). Again, given the evidence, we consider this a reasonable likely misidentification rate that is still provides the benefit of the doubt to the species.

Scenarios 5 and 6 are almost identical to Scenarios 1 and 2; however, for these scenarios we revised the Bayesian analysis for interaction rates and predictions of future anticipated number of interactions (see *Approach to the Assessment* section) under the assumption that 75% of giant manta ray were misidentified and that 75% fewer unidentified mobulids should be assigned to giant manta ray than were in the initial analysis. We reduced the annual observed giant manta ray interactions in Table 52 by 75% to estimate new interactions rates for both FAD and free sets. In our analysis of the raw observer data, with no assumption of misidentification, we estimated that 36.85% of unidentified mobulids were giant manta ray. Under Scenarios 5 and 6 we assumed that this proportion was overestimated by 75%, or that 9.21% ([36.85 x 25]/100) of unidentified mobulids were giant manta ray. The new values are reported in Table 63. In summary, the scenarios capture our uncertainty in giant manta ray population structure and correct identification of giant manta rays:

> Scenario 1: 17 discrete subpopulations; 96% at-vessel mortality rate; 5-yr maximum running average anticipated future annual interactions from the analysis of the observer database; r_{max} = 0.019, 0.042, and 0.119; 10, 20, and 40 years.
>
> Scenario 2: 17 discrete subpopulations; 96% at-vessel mortality rate; 95th percentile anticipated future annual interactions from the analysis of the observer database; r_{max} = 0.019, 0.042; 10, 20, and 40 years.
>
> Scenario 3a and b: 17 discrete subpopulations; 96% at-vessel mortality rate; mean anticipated future annual interactions; r_{max} = 0.019, 0.042; assume 50% (3a) and 75% (3b) of observed giant manta rays were misidentified and were other mobulid species; 10 years.
>
> Scenario 4: Metapopulation with emigration and immigration between subpopulations; 96% at-vessel mortality rate; range of 2 to 100 annual interactions per year per subpopulation; r_{max} = 0.019, 0.042; 10, 20, and 40 years (Table 58).
>
> Scenario 5: 17 discrete subpopulations; 96% at-vessel mortality rate; 5-yr running average anticipated future annual interactions under the assumption of 75% misidentification; r_{max} = 0.019 and 0.042; 10, 20, and 40 years.
>
> Scenario 6: 17 discrete subpopulations; 96% at-vessel mortality rate; 95th percentile anticipated future annual interactions under the assumption of 75% misidentification; r_{max} = 0.019, 0.042; 10, 20, and 40 years.

Table 58. Parameters and values used for Scenario 4 subpopulation projections.

Parameter	**Values**
Initial Subpopulation Abundance	60, 100, 200, 400, 600, 800, 1000, 1500, 1875, 2464
Fishery Captures per Year per Population	2, 5, 10, 20, 30, 40, 50, 100
At-vessel Mortality	0.96
Maximum intrinsic Population Growth Rate (r_{max})	0.019, 0.042
Immigration Rate	0%, 5%, 10%, 15%

Parameter	Values
Emigration Rate	0%, 5%, 10%, 15%

Scenario 1 examined a total of 510 combinations of annual fishery captures, initial abundance and r_{max} to represent the impact of the fishery on the 17 identified subpopulations assuming giant manta rays were correctly identified and our proration of unidentified mobulids was correct. Table 59 summarizes the resulting subpopulation trajectories for these 510 simulations. All subpopulations with starting size of 60 individuals and 12 or more annual fishery exposures were extirpated within 10 years (Appendix B). Figure 69 summarizes the impact of the intrinsic rate of population growth on three subpopulations, Nauru with the highest estimated fishery exposures, Yap with the lowest estimated fishery exposures, and Solomon Islands with intermediate fishery exposures. For Nauru, all subpopulations with initial abundances of less than 1,000 or less than 800 reach extirpation within 20 years if $r_{max} = 0.019$ or 0.042, respectively. If $r_{max} = 0.116$, subpopulations with initial abundance greater than 600 have increasing trajectories while smaller subpopulations all reach extirpation within 20 years. For PNG West, subpopulations with initial abundance less than 1,500 decline if $r_{max} = 0.019$; for $r_{max} = 0.042$, subpopulations with initial abundance greater than 600 have increasing trajectories while all smaller population sizes decline. If $r_{max} = 0.116$, subpopulations with initial abundance greater than 200 individuals have increasing trajectories while smaller subpopulations decline. For Yap, with only 2 estimated annual fishery exposures, subpopulations continue to increase regardless of r_{max} or initial abundance (Figure 69). Overall, for all subpopulation projections at 10 years, 42% ([23+49]/170, Table 59) of them were extirpated or exhibited severe declines if $r_{max} = 0.019$ (Table 59). As noted in the *Status of Listed Resources* section, to be viable, subpopulations most likely consist of at least 1,000 individuals. If we focus only on initial population abundances greater than 1,000, 15% ([0+10]/68, Table 59) of them were extirpated or exhibited severe declines if $r_{max} = 0.019$ (Table 59). At an initial population abundance of 1,000, under Scenario 1, 10 ($r_{max} = 0.019$), 2 ($r_{max} = 0.042$), or 0 ($r_{max} = 0.116$) of the 17 subpopulations were extirpated or exhibited severe declines in 10 years (Appendix B). See Appendix B for the full results of Scenario 1 for all subpopulations in Table 56. Similar to Scenario 1, in Scenario 2, all subpopulations with starting size of 60 individuals and 12 or more annual fishery exposures were extirpated within 10 years, see Appendix B for full results of this scenario.

Table 59. Summary of projections for 170 modeled subpopulations (interactions range from 61 to 1 depending upon the subpopulation) at three values of r_{max} for Scenario 1 at 10 years. Numbers in parentheses represent projections with initial abundances of more than 1,000 (*n*= 68 at each r_{max}).

r_{max}	Number Extirpated (= 100% or 1)	Number exhibiting substantial declines (>5% and <100%)	Number exhibiting declines of $\leq$5%	Number exhibiting stability or positive growth
10 Years				
0.019	23 (0)	49 (10)	9 (4)	89 (54)

r_{max}	Number Extirpated (= 100% or 1)	Number exhibiting substantial declines (>5% and <100%)	Number exhibiting declines of ≤5%	Number exhibiting stability or positive growth
0.042	22 (0)	26 (2)	2 (0)	120 (66)
0.116	18 (0)	4 (0)	0 (0)	148 (68)

To summarize the impact of fishery interactions across all subpopulations, we calculated the maximum number of captures for each combination of initial abundance and r_{max} that would allow for stable or increasing subpopulation growth (Table 60). When $r_{max} = 0.019$, this occurs at 49 annual exposures for subpopulations with initial abundance of 2,464, while for subpopulations with initial abundance of 400, this occurs at seven annual exposures. For $r_{max} = 0.116$, subpopulations with initial abundance of 2,464 have positive population growth at 316 annual exposures, however subpopulations with initial abundance of 400 individuals will decline if captures exceed 51 annual exposures.

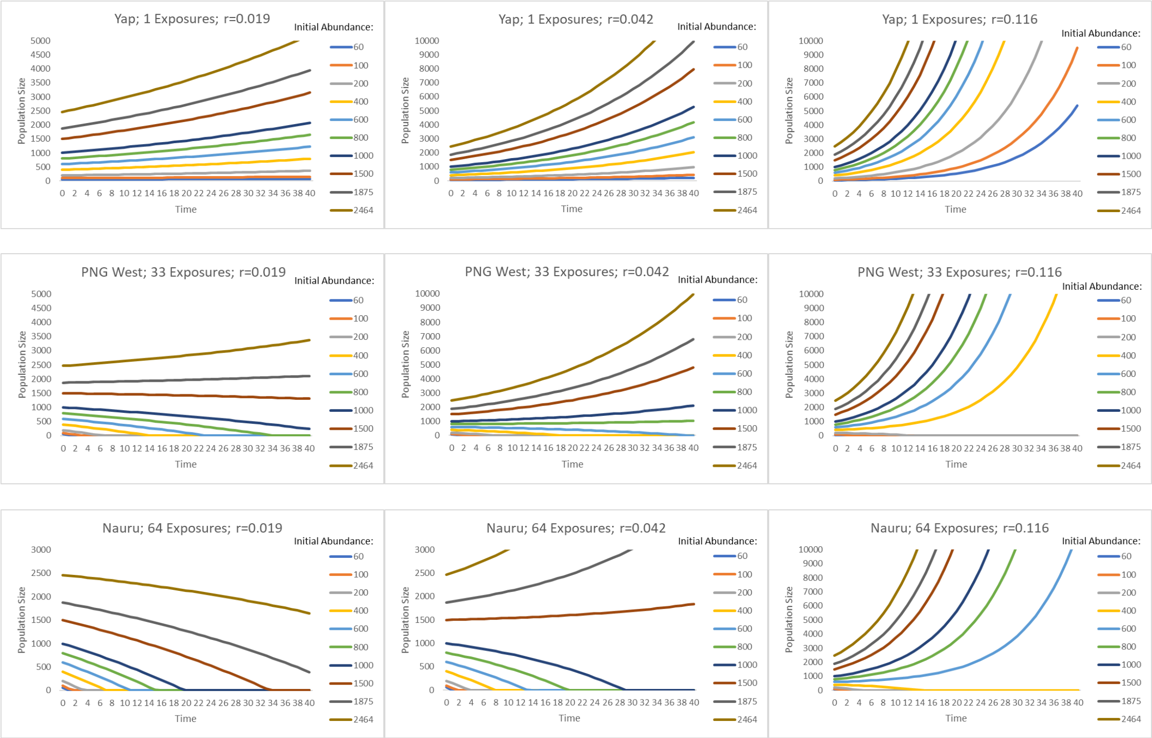


Figure 69. Results of Scenario 1 for the subpopulations with the highest estimated annual fishery exposures (Nauru), the lowest estimated annual fishery exposures (Yap), and a median number of annual fishery exposures (Papua New Guinea (PNG) West; see Appendix B for results of all subpopulations). Each graph shows trajectories for the range of initial abundances (60 to 2,464).

Table 60. Maximum annual numbers of giant manta ray captures that would allow for stable or increasing population growth rates, assuming 96% mortality, for each initial subpopulation abundance and each maximum intrinsic rate of population increase considered.

	Maximum intrinsic rate of population increase (r_{max})		
Initial Abundance	**0.019**	**0.042**	**0.116**
60	1	2	7
100	2	4	13
200	3	8	25
400	7	18	51
600	11	26	76
800	16	35	102
1000	20	44	127
1500	29	67	192
1875	36	83	240
2464	49	109	316

In Scenario 3, we assumed that 50% (Scenario 3a) and 75% (Scenario 3b) of manta rays were misidentified as giant manta rays. Scenario 3 examined a total of 340 combinations of annual fishery captures, initial abundance and two values of r_{max} to represent the impact of the fishery on the 17 identified subpopulations assuming giant manta rays were incorrectly identified as least half of the time and that our proration of mobulids not identified to species over-estimated giant manta ray interactions. Table 61 summarizes the resulting population trajectories for these 340 simulations. Overall, for all subpopulation projections assuming a 50% misidentification rate at 10 years, 29% ([13+36]/170; Table 61) of them were extirpated or exhibited severe declines if $r_{max} = 0.019$ (Table 61). For a 75% misidentification rate, 17% ([6+23]/170; Table 61) of them were extirpated or exhibited severe declines if $r_{max} = 0.019$ (Table 61). As noted in the *Status of Listed Resources* section, to be viable, subpopulations most likely consist of at least 1,000 individuals. If we focus only on initial population abundances greater than 1,000, 3% ([0+2]/68,Table 61) of the projections assuming a 50% misidentification rate were extirpated or exhibited severe declines if $r_{max} = 0.019$ (Table 61). Assuming a 75% misidentification rate, none of the projections were extirpated or exhibited severe declines if $r_{max} = 0.019$ (Table 61). At an initial population abundance of 1,000, under Scenario 3a, two ($r_{max} = 0.019$) or 0 ($r_{max} = 0.042$) of the 17 subpopulations were extirpated or exhibited severe declines in 10 years (Appendix B). Under Scenario 3b, at an initial abundance of 1,000 individuals, none of the 17 subpopulations were extirpated or exhibited severe declines in 10 years at either value of r_{max} (Appendix B). As noted previously, we did not carry this analysis further into the future because we determined it was more prudent to make the misidentification adjustment before calculating the impact of the fishery, rather than after than after subpopulation level effects were calculated.

Table 61. Summary of projections for 170 modeled subpopulations at two r_{max} values (interactions range from 100 to 1 for 50% and 50 to 1 for 75% depending upon the subpopulation) for Scenario 3 at 10 years. Numbers in parentheses represent projections with initial abundances of more than 1,000 (*n*= 68 at each r_{max}).

r_{max}	Number Extirpated (= 100% or 1)	Number exhibiting substantial declines (>5% and <100%)	Number exhibiting declines of <5%	Number exhibiting stability or positive growth
Assume 50% of GMR were misidentified				
0.019	13 (0)	36 (2)	8 (1)	113 (65)
0.042	13 (0)	17 (0)	2 (0)	138 (68)
Assume 75% of GMR were misidentified				
0.019	6 (0)	23 (0)	4 (0)	137 (68)
0.042	6 (0)	12 (0)	0 (0)	152 (68)

Figure 70 summarizes the impact of the intrinsic rate of population growth on three subpopulations, Nauru with the highest estimated fishery exposures, Yap with the lowest estimated fishery exposures, and PNG West with intermediate fishery exposures for Scenario 3a assuming 50% of giant manta ray were misidentified. For Nauru, all subpopulations with initial abundances less than 1,500 individuals decline and abundances less than 800 are extirpated within 40 years if r_{max} = 0.019. If r_{max} = 0.042, subpopulations with initial abundance greater than 800 have increasing trajectories while smaller subpopulations are extirpated within 40 years. For PNG West, subpopulations with initial abundance of 800 or less decline if r_{max} = 0.019, larger subpopulations have increasing population sizes. For r_{max} = 0.042, subpopulations larger than 400 individuals have increasing trajectories while smaller subpopulations decline. For Yap, with only 1 estimated annual fishery exposure, subpopulations continue to increase regardless of r_{max} or initial abundance (Figure 70). See Appendix B for the full results of Scenario 3 for all subpopulations in Table 56.

Figure 71 summarizes the impact of the intrinsic rate of population growth on two subpopulations, Nauru with the highest estimated fishery exposures and PNG West with intermediate fishery exposures for Scenario 3b assuming 75% of giant manta ray were misidentified. For Nauru, subpopulations reach extirpation within 40 years if initial abundance is less than 400 individuals if r_{max} = 0.019. If r_{max} = 0.042, subpopulations with initial abundance greater than 400 have increasing trajectories while subpopulations smaller than 200 individuals are extirpated within 20 years. PNG West, all subpopulations less than 400 individuals decline if r_{max} = 0.019, although only subpopulations with 200 or fewer initial abundance are extirpated within 40 years. For r_{max} = 0.042, subpopulation sizes larger than 200 have increasing trajectories while smaller subpopulations decline. See Appendix B for the full results of Scenario 3b for all subpopulations in Table 56.

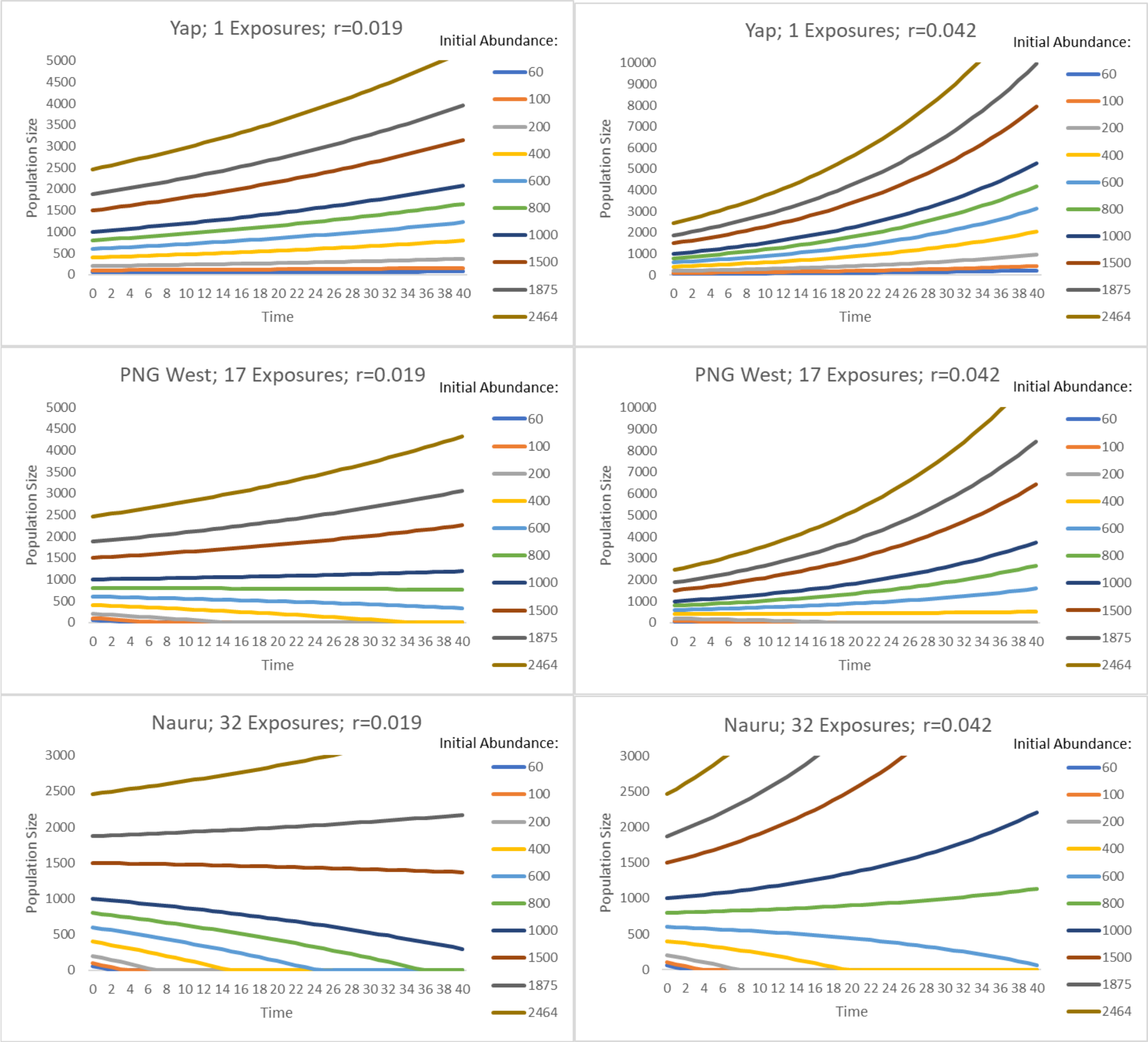


Figure 70. Results of Scenario 3a for the subpopulations with the highest estimated annual fishery exposures (Nauru), the lowest estimated annual fishery exposures (Yap), and a subpopulation with a median number of annual fishery exposures (Papua New Guinea [PNG] West) (see Appendix B for results of all subpopulations). Each graph shows the trajectories over time for the range of initial abundances (60 to 2,464), and a 50% reduction in fishery mortalities compared to Scenario 1.

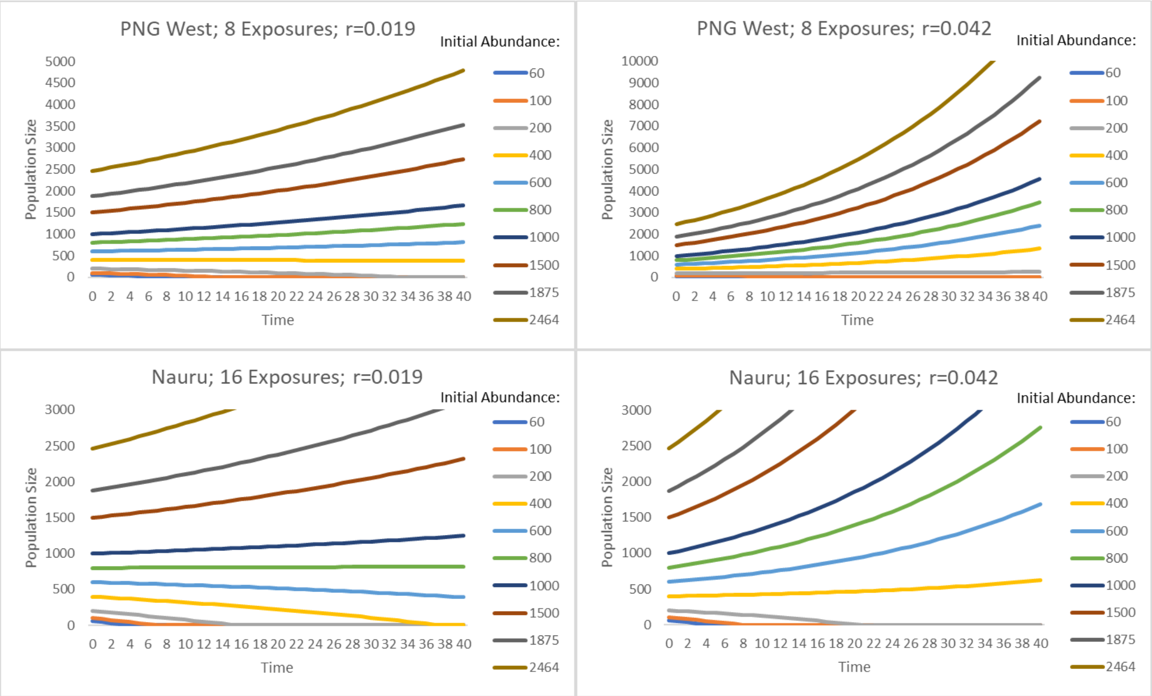


Figure 71. Results of Scenario 3b for the subpopulations with the highest estimated annual fishery exposures (Nauru), and a subpopulation with a median number of annual fishery exposures (Papua New Guinea [PNG] West). We do not show the results for the subpopulation with the lowest estimated annual fishery exposures (1) as the results are identical to those shown in Figure 70 for Yap (see Appendix B for results of all subpopulations). Each graph shows the trajectories over time for the range of initial abundances (60 to 2,464), and a 75% reduction in fishery mortalities compared to Scenario 1.

In Scenario 4, we allowed for connectivity between subpopulations to see if immigration from source subpopulations could maintain smaller subpopulations under fishery pressure. In this Scenario we looked at 8 fishery interaction rates spanning the range of those estimated for the subpopulations in Scenario 1, 10 initial abundances, 4 immigrations rates, 4 emigration rates and 2 values of r_{max} for a total of 2,560 subpopulation simulations. Table 62 summarizes the resulting population trajectories for the subset of these simulations where either emigration or immigration was held at 0% (see Appendix B for full results).

Figure 72 and Figure 74 illustrate the impacts of immigration into small subpopulations with no emigration out of those subpopulation for r = 0.019 and 0.042 respectively. Immigration rates of up to 15% of the subpopulation size had little impact on whether or not subpopulations declined, if a subpopulation was in decline at 0% immigration, it was also in decline with 15% immigration, although there were a few instances where this did not occur. For example with initial abundance of 200, r = 0.019, and fishery exposures = 5, the 15% immigration rate did change the trajectory from declining to increasing (Figure 72).

We also considered whether large subpopulations are able to act as source subpopulations under fishing pressure. For this we considered 0% immigration rates and emigration rates of 5, 10, and 15% of subpopulation abundance (Figure 73, Figure 75). Similar to immigration rates, emigration rates of up to 15% of the population size had little impact on whether or not subpopulations declined, if a subpopulation was in decline at 0% emigration, it was also in decline with 15% emigration, although there were a few instances where this did not occur. For example with initial abundance of 800, r = 0.019, and fishery exposures = 10, the 15% emigration rate did change the trajectory from increasing to declining (Figure 73). See Appendix B for the full results of Scenario 4.

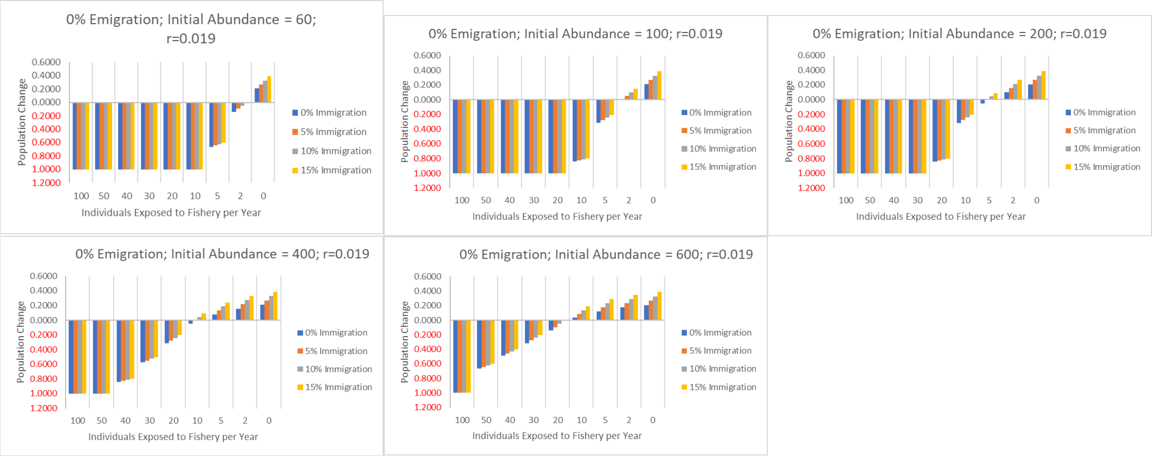
The results for 0% emigration and 0% immigration would be comparable to Scenario 1 and for the range of fishery interactions considered in Scenario 4, 68% of subpopulations would exhibit substantial declines or be extirpated under no immigration or emigration, while 59% of subpopulations would exhibit substantial declines or be extirpated with 15% immigration rates (essentially and increase of 15% of the subpopulation each year; Table 62). However, as these manta rays would emigrate from other subpopulations, we also looked at emigration rates, and under 15% emigration, 73% of subpopulations would exhibit substantial declines or be extirpated (Table 62).

Our results for Scenario 4 indicate that immigration rates up to 15% of the subpopulation size did little to change the overall trajectories (increasing or decreasing), that immigration is unlikely to sustain smaller subpopulations, and that larger population are unlikely to be an adequate source of individuals for smaller subpopulations to offset losses from fishery mortalities as they themselves cannot maintain subpopulation sizes with losses from both fishery mortality and emigration (Figure 72, Figure 74).

Table 62. Partial summary of projections for modeled subpopulations (interactions range from 200 to 2 depending upon the subpopulation) for Scenario 4 at 10 years.

r_{max}	**Number Extirpated (= 100% or 1)**	**Number exhibiting substantial declines (>5% and <100%)**	**Number exhibiting declines of <5%**	**Number exhibiting stability or positive growth**
0% Emigration; 0% Immigration				
0.019	19	29	5	27
0.042	17	19	0	44
0% Emigration; 5% Immigration				
0.019	19	24	5	32
0.042	17	13	6	44
0% Emigration; 10% Immigration				
0.019	19	23	1	37
0.042	17	12	1	50
0% Emigration; 15% Immigration				
0.019	19	20	3	38
0.042	19	12	1	50

r_{max}	Number Extirpated (= 100% or 1)	Number exhibiting substantial declines (>5% and <100%)	Number exhibiting declines of <5%	Number exhibiting stability or positive growth
5% Emigration; 0% Immigration				
0.019	19	34	3	24
0.042	17	19	1	43
10% Emigration; 0% Immigration				
0.019	19	37	7	17
0.042	17	21	1	41
15% Emigration; 0% Immigration				
0.019	19	44	8	9
0.042	17	22	4	37


0% Emigration; Initial Abundance = 60; r=0.019

0% Emigration; Initial Abundance = 100; r=0.019

0% Emigration; Initial Abundance = 200; r=0.019

0% Emigration; Initial Abundance = 400; r=0.019

0% Emigration; Initial Abundance = 600; r=0.019

Figure 72. Results of Scenario 4 for small subpopulations (less than 600 initial abundance), no emigration and r = 0.019. Different rates of immigration are considered: 0% (blue bars), 5% (orange bars), 10% (gray bars), and 15% (yellow bars). Each graph represents a different initial abundance.

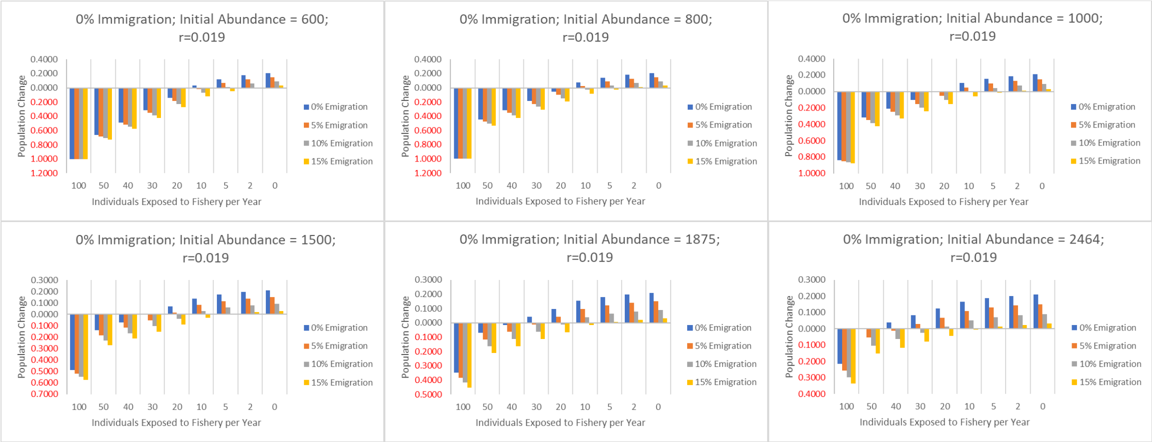


Figure 73. Results of Scenario 4 for larger subpopulations (greater than 600 initial abundance), no immigration and r = 0.019. Different rates of emigration are considered: 0% (blue bars), 5% (orange bars), 10% (gray bars), and 15% (yellow bars). Each graph represents a different initial abundance.

Figure 74. Results of Scenario 4 for small subpopulations (less than 600 initial abundance), no emigration and r = 0.042. Different rates of immigration are considered: 0% (blue bars), 5% (orange bars), 10% (gray bars), and 15% (yellow bars). Each graph represents a different initial abundance.

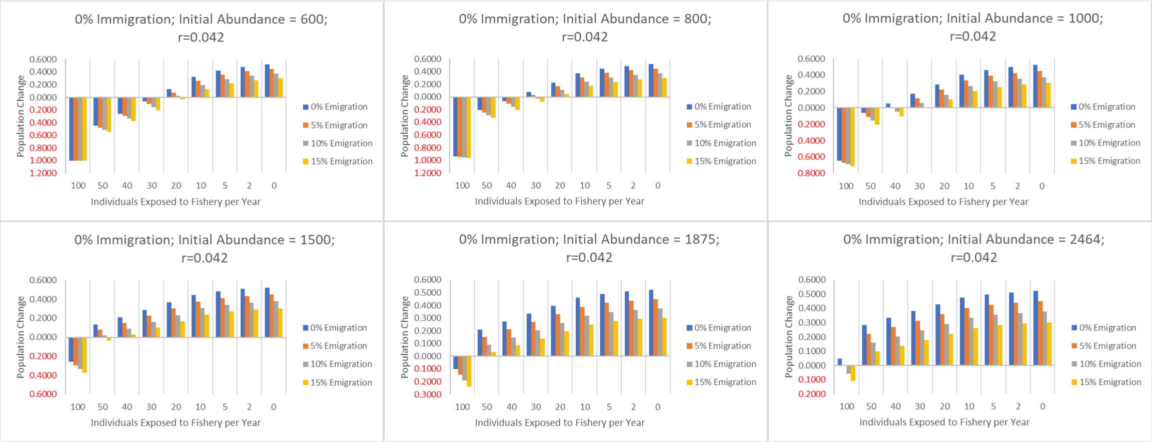


Figure 75. Results of Scenario 4 for larger subpopulations (greater than 600 initial abundance), no immigration and r = 0.042. Different rates of emigration are considered: 0% (blue bars), 5% (orange bars), 10% (gray bars), and 15% (yellow bars). Each graph represents a different initial abundance.

Given that misidentification is likely, we added two additional Scenarios, 5 and 6, to further explore a 75% reduction in number of interactions and considered population projections over 10, 20 and 40 years, for both 5-year running averages (Scenario 5) and 95th percentile (Scenario 6) anticipated interaction levels as was done in Scenarios 1 and 2. To assess the anticipated interactions under this assumption, we repeated the Bayesian analysis assuming that 75% of giant manta ray captures were misidentified (that they were not giant manta ray) and that 75% fewer of the unidentified mobulids category are giant manta ray than were assigned to this category. Using the historic distribution of fishing effort, we proportioned this anticipated number of interactions among the 17 subpopulations. The resulting number of anticipated captures for each subpopulation under Scenarios 5 and 6 are summarized in Table 63. The subpopulation projections resulting from Scenario 5 are summarized in Table 64. For comparison with other scenarios, we show the subpopulation trajectories for the Nauru and PNG West subpopulations for Scenario 5 in Figure 76. We do not show the results for Yap in this figure, as under this scenario we anticipate zero interactions for the Yap subpopulation. As shown in Figure 76, if r_{max}=0.019, subpopulations smaller than 800 individuals decline under the 19 anticipated exposures for the Nauru subpopulation while subpopulations larger than 1000 have increasing trajectories. If r_{max}=0.042, initial abundances greater than 600 individuals increase with 400 individuals close to stable. The Nauru subpopulation is the worst-case scenario and experiences the largest number of anticipated annual captures if the distribution of fishery effort is as described in the project description. For a subpopulation with intermediate anticipated annual interactions like PNG West with 9 per year, if r_{max}=0.019, initial abundances greater than 600 would be increasing, and this would be true for initial abundances as small as 400 individuals if r_{max}=0.042 (Figure 76). Due to the similarity in anticipated captures for the 5-year running average and the 95th percentile (Table 63), the results of Scenario 6 are very similar to those of Scenario 5. Initial population abundance of 1,000 or greater had increasing population trajectories at the lowest r_{max} value of 0.019 under Scenario 6, and if r_{max} = 0.042, all subpopulations with initial abundances of 600 or more had increasing population trajectories.

Table 64 summarizes the resulting subpopulation trajectories for the 340 simulations under Scenario 5. Overall, for all subpopulation projections at 10 years, 16% ([7+20]/170, Table 64) of them were extirpated or exhibited severe declines if r_{max} = 0.019 (Table 64). As noted in the *Status of Listed Resources* section, to be viable, subpopulations most likely consist of at least 1,000 individuals. If we focus only on initial population abundances greater than 1,000, none of the subpopulations experience declines (Table 64) regardless of r_{max} (Table 64). If r_{max} = 0.042, 1% ([5+12]/170, Table 64) of the subpopulations were extirpated or exhibited severe declines in 10 years (Appendix B). At the 20 and 40-year time horizons, the numbers of simulations exhibiting stability or population growth were the same as in Table 64. Fourteen and 20 simulations experienced extirpation at 20 and 40 years respectively, compared to the 7 simulations at 10 years (Table 64; Appendix B), while 16 and 10 simulations experienced substantial declines over 20 and 40 years respectively compared to the 20 simulations at 10 years (Table 64; Appendix B). Until we have more information on the extent of the misidentification issues with giant manta ray in the United States WCPO purse seine fishery, we consider this scenario the most plausible; however, we cannot disregard the giant manta ray captures recorded in the observer database and the results of Scenarios 1 and 2, and therefore we will analyze both levels of interactions. Results of Scenario 6 are detailed in Appendix B, briefly, as noted above, given the similarity in captures between Scenarios 5 and 6, the results are very similar. At the 10, 20 and 40-year time horizons, 137 simulations exhibited stability or population growth. Seven, 14 and 23 simulations experienced extirpation at 10, 20 and 40 years respectively, while 21, 16 and 7 simulations experienced substantial declines over 10, 20 and 40 years respectively (Appendix B).

Table 63. 5-yr maximum running average (RA; Scenario 5) and 95th percentile (Scenario 6) anticipated interactions for giant manta rays captured in the United States WCPO purse seine fishery under the assumption of 75% species misidentification (Scenarios 5 and 6).

Giant Manta Ray Subpopulation	Anticipated Annual Exposures – Max 5-yr RA	Anticipated Annual Exposures - 95th percentile
Nauru	19	19
Kiribati, Gilbert Islands	14	16
Papua New Guinea East	3	3
Solomon Islands	7	8
Tuvalu	7	8
Papua New Guinea West	9	9
Pohnpei FSM	3	3
Kiribati Phoenix Islands East	4	4
Kiribati Phoenix Islands West	6	6

Giant Manta Ray Subpopulation	Anticipated Annual Exposures – Max 5-yr RA	Anticipated Annual Exposures - 95th percentile
Papua New Guinea	1	1
Fiji	1	1
Kiribati, Line Islands	1	1
Cook Islands	1	1
New Caledonia	0	0
Yap FSM	0	0
American Samoa	1	1
Palmyra	1	1

Table 64. Summary of 170 projections for modeled subpopulations at two values of r_{max} (interactions range from 19 to 0 depending upon the subpopulation) for Scenario 5. Numbers in parentheses represent projections with initial abundances of more than 1,000 (*n* =68 at each r_{max}).

r_{max}	Number Extirpated (= 100% or 1)	Number exhibiting substantial declines (>5% and <100%)	Number exhibiting declines of ≤5%	Number exhibiting stability or positive growth
10 years				
0.019	7 (0)	20 (0)	4 (0)	139 (68)
0.042	5 (0)	12 (0)	2 (0)	151 (68)

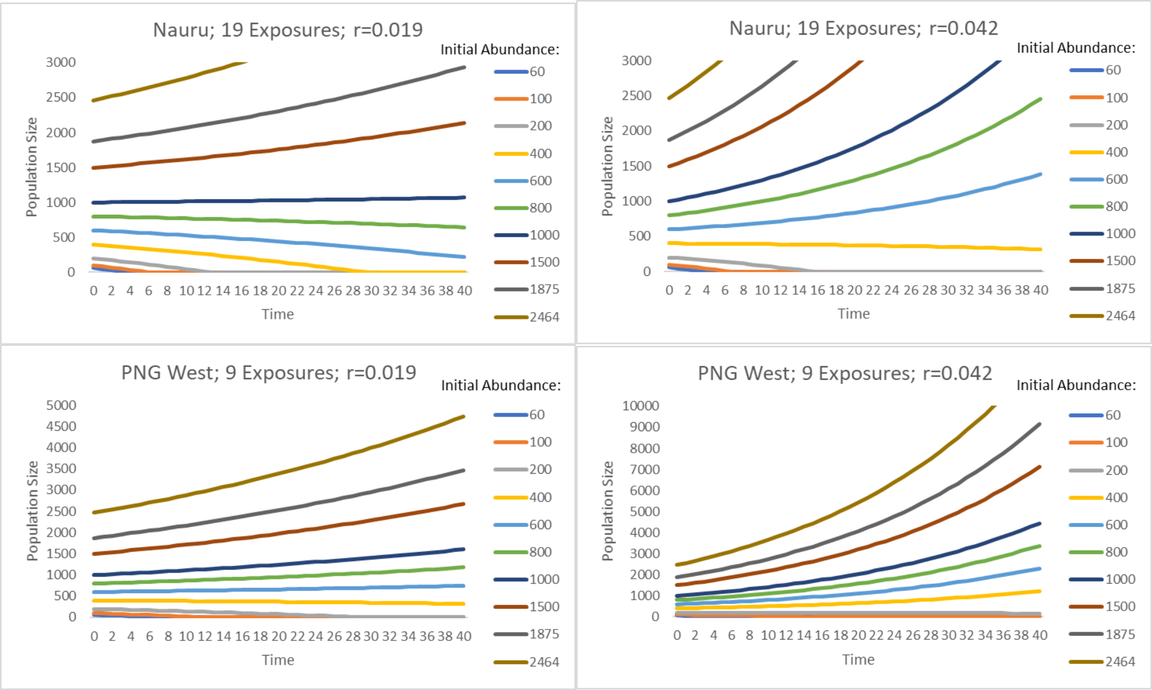


Figure 76. Results of Scenario 5 for the subpopulations with the highest estimated annual fishery exposures (Nauru) and a subpopulation with a median number of annual fishery exposures (Papua New Guinea [PNG] West) (see Appendix B for results of all subpopulations). Each graph shows the trajectories over time for the range of initial abundances (60 to 2,464) with the assumption that 75% of giant manta rays were misidentified and that 75% fewer unidentified mobulids should be considered giant manta ray.

Summary of Scenario Results

We created more than 5,000 reasoned simulations to examine the effect of the action on threatened giant manta rays. The first two scenarios examine the effect of the action on 17 subpopulations with ten initial (pre action) abundances, three values for r_{max}, and looked at multiple time intervals (10, 20 and 40 years) for the level of anticipated number of interactions estimated assuming 100% correct species identification and 36.85% of unidentified mobulids were giant manta ray (i.e. 187 maximum 5-year running average per year for Scenario 1 and 95th percentile of 199 per year for Scenario 2). Scenarios 5 and 6 were conducted in the same manner as Scenarios 1 and 2 but presumed a 75% misidentification rate for giant manta ray. 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.

Based on our literature survey, although it is thought that there is likely very little movement between subpopulations, we considered the possibility of movement given the overlapping spatial distribution of giant manta ray captures in the fishery (Scenario 4); however, this scenario may be less realistic than other scenarios. In any case, the results of Scenario 4 indicate that it is unlikely that smaller populations would be sustained by immigration and unlikely that larger subpopulations can withstand both the loss of individuals through emigration and fishery mortalities, and therefore we do not consider this a viable justification for sustained populations under fishing pressure. Therefore, we do not address this Scenario further in our analysis and conclusions.

Scenarios 3a and 3b were developed as ‘quick’ ways to consider the impact of misidentification and compare those results over 10-year time frames to the results of Scenario 1. Scenario 5 differs from Scenario 3b, and is a more appropriate method for adjusting the estimated number of interactions. In Scenario 3b we simply reduced the captures per subpopulation in Scenario 1 by 75%, whereas under both Scenarios 5 and 6 we conducted a Bayesian analysis to estimate total annual anticipated interactions based on the starting assumption that 75% of giant manta ray captures were misidentified and that 75% fewer of the unidentified mobulids category are giant manta ray. Therefore, Scenario 5 is an improvement to Scenarios 3a and 3b, and replaces Scenarios 3a and 3b

Scenarios 2 and 6 were developed as worst-case scenarios of interactions with giant manta ray equivalent to the 95% percentile for both the no misidentification and 75% misidentification assumptions respectively. Given that the 95th percentile values are the upper limit of anticipated annual captures, we do not anticipate this level of capture to occur each year, and therefore we do not anticipate that these scenarios will be representative of the impact of the fishery on the species, in our analysis and conclusions and do not address Scenarios 2 and 6 further.

We consider Scenario 5 to represent the best scientific and commercial information available to evaluate the effects of the fishery on giant manta ray and therefore, for the remainder of this biological opinion we focus most of our attention on this scenario. Even so, we felt it prudent toalso examine the data as it was presented and recorded by observers, using our typical method for evaluating observer data and without adjustment for the misidentification of giant manta ray (Scenario 1). This allowed us to compare the results of Scenario 5 with Scenario 1 to highlight the impact of species identification. The scenarios are developed from the same data, which as we have said previously is based on legitimate observations in this fishery. Scenario 1, as it represents our typical analytical approach to accept observer data at face value, is a plausible scenario but we believe it to be superficially plausible. While Scenario 1 presents a specious view of the effect of the fishery on giant manta ray, by comparing these estimates to our most plausible estimates of the effect of the fishery on giant manta ray under Scenario 5 we feel that we have conducted the most comprehensive evaluation of the effect of the action possible, provided the benefit of the doubt to the species as appropriate, and done so in a way that does not sacrifice cogent reasoning.

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interaction levels using Bayesian inference techniques as described in the *Approach to the Assessment* section of this biological opinion. The predictions were based on the bycatch of individual rays recorded by observers as giant manta ray and the *Mobula* nei classification in the purse seine fishery. Additionally, the predictions are based on a future effort estimate of 3,100 sets, with 1,581 of those being FAD sets Table 65 contains the resulting predictions of future mean annual interactions for the species as a whole, inclusive of all putative subpopulations. As noted in Section 2.2.2.6 of this biological opinion, we found the 95th percentile from the Bayesian inference model generally underestimated the highest numbers of anticipated captures and we therefore used a nonparametric bootstrap analysis to estimate the 95th percentile of anticipated captures. That value is listed in Table 65 along with the 5-yr running average and sum for giant manta rays in the purse seine fishery (NMFS 2019c). In addition, as described above for Scenarios 5 and 6, we repeated the Bayesian analysis assuming that 75% of giant manta ray captures were misidentified (that they were not giant manta ray) and that 75% fewer of the unidentified mobulids category are giant manta ray. Results of this analysis are provided in Table 65.

Table 65. Estimated number of giant manta ray individuals interacting with the United States WCPO purse seine fishery as represented by the mean, 95th percentile the maximum 5-year running average and 5-year running sum, using the observer data with and without adjustment to account for likely misidentification*.

Estimated numbers of giant manta ray interacting with the fishery	**Annual Mean**	**Annual 95th percentile**
Number based on unadjusted observer data	165	199
Number adjusted based on assumption that 75% are misidentified	45	50
	5-yr Running Average	**5-yr Running Sum**
Number based on unadjusted observer data	187.4	937
Number adjusted based on assumption that 75% are misidentified	47.0	236

* Numbers are based on unadjusted observer data

5.2.2 Entanglement in FADs

5.2.2.1 Marine Mammals

The large whales considered in this biological opinion have not been documented to have an interaction with FADs that lead to an entanglement. Their large body sizes likely limit their vulnerability to entanglement in drifting nets associated with the dFADs used in the United States WCPO purse seine fishery. Similar to ropes associated with pots and traps (Johnson et al. 2005), aFADs may pose an entanglement risk to large whales, but ropes and netting used in dFADs are not materials known to entangle large whales (Stelfox et al. 2016). There were no observations of ESA-listed whales associated with FADs in the WCPO purse seine fishery observer data.

5.2.2.2 Sea Turtles

Interaction with purse seine fishery FADs independent of a fishery set may also pose a hazard to sea turtles and as discussed in the *Interactions with purse seine fishing operations* section of this biological opinion, these interactions were categorized separately from the fishery set interactions. Since 2008, a total of 20 sea turtles have been observed interacting with FADs by United States WCPO purse seine fishery observers (Table 66). Green sea turtles have the highest interaction rate, followed by hawksbills. In comparison to Fishery Set interactions, FAD interactions comprise a small fraction of the United States WCPO purse seine fishery sea turtle interactions (Table 66). It is important to note, however, that the observed sea turtles interacting with the fishery are a minimum estimate as it cannot be known how many sea turtles interact, or become entangled on unobserved FADS, or if sea turtles are entangled underneath observed FADS that cannot be seen. Of the sea turtles observed interacting with FADs, 88% were recorded as uninjured while 4% (1 sea turtle) were dead (Table 67 and Table 68).

There are regulations in place for the United States WCPO purse seine fishery for releasing sea turtles observed entangled in FADs. According to the WCPFC Conservation and Management Measure 2018-04 and 50 CFR 300.223(f)(2), fishermen are required, to the extent practicable, release all sea turtles observed entangled in FADs.

Table 66. Number of the different turtle species observed interacting with FAD used by the United States WCPO purse seine fishery between 2008 and 2018, not associated with a fishery set. Reported values for interaction types are the numbers of observed sea turtles. Also reported are the proportion of FAD interactions across species and the proportion of FAD interactions in relation to Fishery Set interactions.

Species	**FAD Interactions**	**Proportion of all FAD Interactions**	**Proportion of all observed interactions**
Green	10	0.42	0.034
Leatherback	0	0	0
Loggerhead	3	0.125	0.01
Olive Ridley	3	0.125	0.01
Hawksbill	8	0.334	0.027
Unidentified	0	0	0
Grand Total	24	1	0.08

Table 67. Number of sea turtles observed interacting with United States WCPO Purse Seine fishery FADs between 2008 and 2018 by species and release condition.

	Release Condition				
Species	**A1** **Alive and healthy**	**A2** **Alive but injured**	**A3** **Alive but unlikely to survive**	**D** **Dead**	**Grand Total**
Green	8	2	0	0	10
Leatherback	0	0	0	0	0
Loggerhead	3	0	0	0	3
Olive Ridley	3	0	0	0	3
Hawksbill	7	0	0	1	8
Grand Total	21	2	0	1	24

Table 68. Proportion of sea turtles observed interacting with United States WCPO Purse Seine fishery FADs between 2008 and 2018 by species and release condition.

	Release Condition				
Species	**A1** **Alive and healthy**	**A2** **Alive but injured**	**A3** **Alive but unlikely to survive**	**D** **Dead**	**Grand Total**
Green	0.33	0.08	0	0	0.42
Leatherback	0	0	0	0	0
Loggerhead	0.125	0	0	0	0.125
Olive Ridley	0.125	0	0	0	0.125
Hawksbill	0.29	0	0	0.04	0.33
Grand Total	0.88	0.08	0	0.04	1.00

5.2.2.3 Elasmobranchs

Shark entanglement in FADs has been documented only for *Carcharhinus* species (i.e. silky and oceanic whitetip sharks; Chanrachkij and Loog-on 2003; Filmalter et al. 2013; Murua et al. 2017) and Filmalter et al. (2013) indicate that those species are the only shark species known to regularly associate with floating objects. There was one set that noted an oceanic whitetip shark entangled in a FAD, and this is the only interaction observed from 2008 to 2018 in the United

States WCPO purse seine fishery. No FAD interactions were observed for scalloped hammerhead sharks or giant manta rays.

5.3 Response Analyses

As discussed in the *Approach to the Assessment* section of this biological opinion, response analyses determine how listed resources are likely to respond after being exposed to an Action's effects on the environment or directly on listed species themselves. For the purposes of consultations on fishing activities, our assessments try to detect the probability of responses that might result in reducing the fitness of listed individuals. Ideally, our response analyses consider and weigh evidence of adverse consequences, beneficial consequences, or the absence of such consequences.

The most significant hazard the United States WCPO purse seine fishery presents, is to ESA-listed species being encircled by and retained in the net through pursing, hauling, and brailing, which can injure or kill sea turtles, oceanic whitetip sharks, scalloped hammerhead sharks and giant manta rays. Direct impacts to sea turtles, sharks and rays as a result of these interactions often include some element of internal or external wounding, crushing, and hydrostatic effects. Capture in the fishing gear can result in abrasion against nets or against other catch. Poor handling by the crew may also lead to stress or injury, or delayed mortality (Davis 2002). For sharks and rays, the need for sustained swimming can lead to exhaustion. Studies tend to show high mortality rates for sharks and rays due to the varying degrees of physical trauma associated with purse seine gear, and mobility restrictions during the brailing process, among the other impacts observed for bycatch (Poisson et al. 2014b; Hutchinson et al. 2015; Croll et al. 2016).

Although survivability studies have been conducted on some listed species captured in purse seine fisheries, long-term effects are nearly impossible to monitor; therefore, a quantitative measure of the effect of purse seining on sea turtle, elasmobranch, and cetacean populations is very difficult. Even if listed species are not injured or killed after being entrapped or entangled, 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, elasmobranchs, and cetaceans, are likely to respond to interactions with purse seine gear.

Those individuals that are considered either uninjured (A1) or injured (A2) by the observer may still suffer impaired swimming or foraging abilities, altered migratory behavior, and altered breeding or reproductive patterns, and latent mortality as a result of their interaction with the fishery. For the large whales considered in this biological opinion, their direct interaction with the fishery typically ends after they are encircled by the net and are then released by fishermen, or swim free on their own. Some individuals break through the net to escape. Direct impacts from these interactions are likely behavioral, with temporary, short term interruptions of behavioral patterns such as foraging, migrating, or socializing, but can also include elevated stress to the animals. Marine mammals that break through the net to escape may sustain abrasions similar to scars rated as minor injuries for whales and documented from light entanglements (Frost et al. 1983; Knowlton et al. 2005). NMFS has criteria for making serious injury determinations under the MMPA for marine mammals after sustaining injuries caused by injuries from fishing gear interactions and vessel strikes (NMFS 2012a). These criteria do not align exactly for the types of interactions that occur in the purse seine fishery because they are more specific for entanglement, hookings and vessel strikes, but they are helpful. Based on the description of the types of injuries that large whales sustain by breaking through the net as described in the literature and by the observers it appears that the injury category aligns with the L5b category which is considered a superficial laceration; described as not deeper than blubber layer, does not affect major artery, or cut bone. This type of injury is considered non-serious. NMFS defines serious injury in regulation (50 CFR 229.2) as an "injury that is more likely than not to result in mortality" (Andersen et al. 2008).

No survivability studies have been conducted for sea turtles interacting with purse seine fisheries, therefore, a quantitative measure of the effect of capture by the purse seine fishery on sea turtles is very difficult. Even if sea turtles are not injured or killed after being entangled or landed, 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 and other species considered in this consultation are likely to respond to these interactions with fishing gear.

5.3.1 Interactions from Purse Seine Operations

5.3.1.1 Marine Mammals

Capture by entrapment can result in various levels of stress or injury, and potential mortality whether acute or delayed. Smaller species (i.e. odontocetes) are more often encountered compared to baleen whales with more severe effects noted in the literature. The WCPFC (2010) notes that observer records portray that most baleen whales are able to escape the net on their own accord before the net wall is closed, while others punch holes through the net wall. However, mortalities do occur. WCPFC (2010) reported 22 observed sets were made on baleen whales from 2007 to 2009 with an estimated mortality of 3 individual Bryde's whale, which is the smallest baleen whale that this fishery has had known interactions with. There are no reported mortalities for any ESA-listed whales in the United States portion of the fishery.

Sei Whale

As described above in the *Exposure Analysis* section, from 2008 to 2018 there were 129 interactions with sei whales. We anticipate that 3 (95th percentile: 7) sei whales will interact with the United States WCPO purse seine fishery per year. Most sei whales interacting with the United States WCPO purse seine fishery were released alive and uninjured (96%; Table 21), while five individuals, or 4%, were released visibly distressed or injured. Two of the injured or distressed whales were in the net as it was hauled and released once the net was near the vessel; one was chased out of the net; one ripped a hole in the net to escape, and two were released together from the net and were classified as distressed by the observer. The observer reported seeing scratches on one of the whale's rostrum. It is unlikely any of these whales sustained life-threatening injuries from these interactions. The response to the exposures to the fishery will primarily be behavioral. The one thing that we cannot assess from the information we have is whether or not the same individuals are captured multiple times in the *Action Area* and what that impact may be. Sei whales are the most frequently encountered large whale in the *Action Area* and if individuals have multiple encounters these interruptions to their normal behaviors may have larger effects on individuals.

Fin Whale

As described above in the *Exposure Analysis* section, from 2008 to 2018 there were six interactions with fin whales. We anticipate that 0 (95th percentile: 1) fin whale will interact with the United States WCPO purse seine fishery per year. Most fin whales interacting with the United States WCPO purse seine fishery were released alive and uninjured (5 out of the 6), while one individual was released likely distressed or injured based on the observer comments. The one injured or distressed whale was in the net as it was hauled and released once the net was near the vessel, the observer was unable to ascertain its condition upon release. It is unlikely any of these whales sustained life-threatening injuries from these interactions. The response to the exposures to the fishery are likely behavioral as described above for sei whales. We expect individual whales will experience stress during the interaction and immediately following post release which may temporarily alter essential behaviors such as feeding, breeding, or migration but would not affect on the individual's lifetime fitness.

Sperm Whale

As described earlier in the *Exposure Analysis* section, from 2008-2018 there were four observed interactions with sperm whales with an additional estimate of three more caught after accounting for unobserved sets and unidentified marine mammals. Based on the general description of how large whales interact with the gear and the released condition code of alive and healthy we assume that the sperm whales were either released from the net or broke through. We expect individual whales will experience stress during the event and immediately following post release which may temporarily alter essential behaviors such as feeding, breeding, or migration but would not affect an individual's lifetime fitness.

We would expect these same sublethal responses from any future interactions with sperm whales in this fishery, which is estimated to be 0 (95th percentile: 1) annually.

5.3.1.2 Sea Turtles

Entrapment by encirclement in the purse seine net is the primary driver for all interactions between sea turtles and the purse seine fishery. Sea turtles are particularly prone to being entangled in fishing gear because of their body shape 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 (Chaloupka et al. 2008c). Sea turtles have been observed entangled in the netting used by the purse seine fishery.

Some sea turtles entangled in purse seine nets may be forcibly submerged but surface before drowning. Studies have shown that sea turtles that are allowed time to stabilize after being forcibly submerged have a higher survival rate (NMFS 2006). However, this depends on the physiological condition of the turtle (e.g., overall health, age, size), time of last breath, time of submergence, environmental conditions (e.g., sea surface temperature, wave action, etc.), and the nature of any sustained injuries at the time of submergence (NMFS 2006). Currently, the WCPFC's Conservation and Management Measure 2018-04 requires fishermen to implement, as appropriate, the FAO Guidelines to Reduce Sea Turtle Mortality in Fishing Operations which include bringing aboard, if practicable, any captured hard-shell sea turtle that is comatose or inactive as soon as possible to foster its recovery before returning it to the water. Fishermen are also required, if a sea turtle is entangled in the net, to stop the net roll as soon as the turtle comes out of the water; disentangle the turtle without injury before resuming the net roll; and to the extent practicable, assist the recovery of the turtle before returning it to the water. These measures would improve survivability of any forced submergence of sea turtles due to entanglement in the purse seine net.

Sea turtles that are either entangled or entrapped within the net and are not released prior to pursing and hauling are subject to additional stress and potential injury during the net roll and sacking up process, the subsequent removal of the catch with a brailer and sorting. From observer records with adequate detail, we estimate that approximately 72% of sea turtles observed interacting with United States WCPO purse seine fishery sets are landed and subject to brailing. While still in the net, sea turtles can be crushed or have difficulty breathing under the weight of the catch and crushing and other injuries are possible during the brailing and sorting processes. Even if sea turtles are uninjured during this process, the experience is likely to result in elevated stress levels.

Capture may result in a range of outcomes including the animal being released unhurt, injured, or killed depending on each unique interaction. In purse seine operations, turtles can be identified by crew and removed from the purse by hand or dip-net before the turtle has the opportunity to become entangled in the net wall. We estimate that approximately 20% of sea turtles observed interacting with United States WCPO purse seine fishery sets are removed from the net, or disentangled from the net and set free, prior to hauling. If turtles are not removed from the net, they may be lifted and could fall contacting the deck of the vessel with force or may be crushed by the block as the net is hauled (Hall and Roman 2013; Hall et al. 2017). Drowning may also occur if the turtle cannot reach the surface to breathe. Stress responses would be expected during entrapment due to confinement and could potentially affect an individual’s fitness depending on the severity of the interaction.

5.3.2 Post Interaction Survival

5.3.2.1 Sea Turtles

As noted previously, there are no post-interaction survival and mortality estimates for sea turtles interacting with purse seine fisheries. For longline fisheries, NMFS used Ryder et al. (2006) to estimate post-hooking survival and mortality of turtles captured and released alive. These criteria were developed by a panel of experts in biology, anatomy/physiology, veterinary medicine, satellite telemetry and longline gear deployment and has been revisited as more data and studies have become available. The criteria were last assessed by experts in 2011 (Swimmer and Gilman 2012) and no changes were made to the mortality categories which are provided in at the beginning of the *Exposure Analysis* section of this biological opinion.

As part of the biological opinion on the Hawaii-based shallow-set longline fishery (NMFS 2019b), NMFS conducted a literature review to update the criteria of Ryder et al. (2006). Based on this review, NMFS determined that the criteria and rates described in Ryder at al. (2006) are still the most appropriate criteria to use to assess post-hooking mortality of turtles caught in the Hawaii-based shallow-set longline fishery. The criteria establish a mortality estimate based on whether the turtle was lightly hooked (flipper, jaw or mouth), or deeply hooked (ingested) and how much gear was left on the animal when it was released. While injuries from hooking are not an issue with the purse seine fishery, Ryder et al. (2006) incorporate the expert opinion of veterinarians and sea turtle physiologist to account for factors such as stress from the interaction that are applicable to purse seine interactions. It was noted, for example, there may be no detectable difference in behavior between and injured and uninjured sea turtles while on deck, both may appear strong and robust, while post-release tracking show differences in diving behavior and movement that can impact survivability. As sea turtles that interact with the purse seine fishery are released with no gear attached, we use the post-hooking mortality criteria from Ryder et al. (2006) that correspond to being released with all gear removed.

Sea turtles interacting with the United States WCPO purse seine fishery that are released with condition code A1 (alive and healthy) had a range of intensity of interactions with the purse seine fishery, from being encircled by the net and either finding their own way out or being released by fishermen, which likely resulted in a temporary elevation of stress levels to being retained in the net through hauling and brailing before being landed on the deck, which likely results in much more elevated stress levels and the potential for internal injuries not apparent to the observer. The post-hooking mortality rates for sea turtles that are hooked externally with or without entanglement or hooked in the upper or lower jaw with or without entanglement and in both situations release with no gear attached are 5 and 10% respectively for hardshell sea turtles and 10 and 15% for leatherback sea turtles (Ryder et al. 2006). Again, these mortality rates are not solely reflective of the observed injuries caused by hooking but are the result of a more holistic assessment of the stress and potential for unobservable injuries associated with being captured and landed by the longline fishery.

The captured sea turtles that are released from the net prior to the haul are unlikely to have sustained lethal injuries, and they likely have a post-interaction mortality rate of 0%. However, the majority of sea turtles classified as uninjured were landed with the catch and at a minimum suffered stress comparable to externally-hooked and landed sea turtles in the longline fishery. A subset of these sea turtles may have sustained internal injuries not apparent to the observer that may be lethal after release. Therefore, we consider it reasonable to assume that sea turtles interacting with the United States WCPO purse seine fishery sets that were released uninjured as judged by the observer would experience post-interaction mortality rates within the range of 0% to 10% for hardshell and 0% to 15% for leatherback sea turtles, consistent with injuries in the longline fishery that including hooking in the jaw, being landed, having gear removed and being released.

Similarly, the post-hooking mortality rate for sea turtles hooked in cervical esophagus, glottis, jaw joint, soft palate, tongue, and/or other jaw/mouth tissue parts not categorized elsewhere, with or without entanglement and released with no gear attached is 25% for hardshell sea turtles and 35% for leatherback sea turtles. As sea turtles released with a condition code of A2 were observed to either be in distress or to have sustained an injury, we categorize them as at least equivalent to sea turtles that are hooked externally and released with no gear attached and at most equivalent to the more severe hooking injury described above. We therefore consider the range of 5 to 25% post-release mortality for hardshell sea turtles and 10 to 35% mortality for leatherback sea turtles released with condition code A2. We further assume a 100% mortality for sea turtles released with condition code A3 (dying).

Hawksbill Sea Turtle

As described above in the *Exposure Analysis* section, from 2008-2018 there were 59 interactions with hawksbill sea turtles, 51 of those associated with a fishery set and eight associated with a FAD not involving a set. We anticipate that 4 (95th percentile: 6) hawksbill sea turtles will interact with the United States WCPO purse seine fishery per year with a maximum 5-yr running average of 4.5. Most hawksbill sea turtles interacting with the United States WCPO purse seine fishery were released alive and uninjured (78%; Table 30), while four individuals, or 7%, were released injured and eight were released dead or dying (15%). While we do not have details on all of the injuries, two of them reported that the turtle was in distress. One of the dying turtles had a large wound in its shoulder and inability to use the flipper. In addition, the majority of the uninjured hawksbill sea turtles were reported as landed which means that they were pursed in the net with the remainder of the catch, likely resulting in forced submergence and exposure to crushing for an unknown amount of time, before being brailed out of the net. At a minimum these processes would result in elevated stress levels and potentially turtles may have sustained injuries that were not apparent to the observer including internal injuries from compaction in the catch/net. The response to exposures to this fishery will range from primarily behavioral, with a temporary, short-term interruption of behavioral patterns such as foraging, migrating, or socializing to sub-lethal impacts that can result in delayed growth or reproduction to death, either at the vessel or post-release.

Using the mortality coefficients of Ryder et al. (2006) as described above, of the 51 captured turtles an estimated 9 to 15 hawksbill sea turtles died over 11 years from interactions with the United States WCPO purse seine fishery sets (Table 69). The remaining hawksbill sea 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 69. Estimated number of hawksbill sea turtle mortalities by release condition for interactions with the United States WCPO purse seine fishery from 2008 through 2018. Numbers are based on the total number of interactions and injury mortality coefficients of Ryder et al. (2006) as described in the text. Total anticipated mortality rates for the range of mortality coefficients considered provided in the last row.

Release Condition	Number of Turtles Observed	Anticipated Proportion of Captures per Release Condition	Mortality Coefficient Low	Mortality Coefficient High
Alive and healthy (A1)	46	0.78	0	0.10
Alive but injured (A2)	4	0.07	0.05	0.25
Alive but unlikely to survive (A3)	7	0.12	1	1
Dead (D)	2	0.03	1	1

Grand Total	59	1	**Total Mortality Rate – Low:** 0.156	**Total Mortality Rate – High:** 0.247

We applied the estimated mortality rates from Table 69 to the anticipated exposure numbers for hawksbill sea turtles (see Table 32). We calculated the mortality for the mean and 95th percentile for the annual interaction predictions for hawksbill sea turtles as well as the maximum 5-yr running average and running sum (Table 70). NMFS predicts that the United States WCPO purse seine fishery would capture a 4 (95th percentile: 6) hawksbill sea turtles each year. The estimated number of hawksbill sea turtles that would likely die from their capture would be 1 (95th percentile: 1) in any given year (Table 70). Based on the maximum 5-yr running sum, we anticipate that between 4 and 6 hawksbill sea turtles will die over 5 years (Table 70).

We would also expect sublethal responses from hawksbill sea turtles that are captured, landed and released alive. The responses of animals may range from those that are temporary in nature such as elevated stress levels to more significant injuries that may affect feeding, movement, or even breeding success. These effects may decline over time as the surviving turtles heal from their injuries, although the extent to which such sublethal injuries occur or persist are not known.

Table 70. Anticipated mortalities for hawksbill sea turtles interacting with the United States WCPO purse seine fishery. Anticipated mortality estimates are provided for the estimated mean and 95th percentile of hawksbill sea turtle posterior estimated exposures in terms of anticipated annual interactions with fishing gear (inclusive of FAD interactions) and the maximum 5-year running average and running sum. Number outside of parentheses are the anticipated mortalities based on the low mortality coefficients in Table 69, numbers inside the parentheses are based on the high mortality coefficients in Table 69.

Bayesian Inference Model and Nonparametric Bootstrap		
	Annual Mean	**Annual 95th percentile**
Predicted Exposures	1 (1)	1 (2)
Maximum Running Average and Sum		
	5-yr Running Average	**5-yr Running Sum**
Predicted Exposures	1 (1)	4 (6)

Leatherback Sea Turtle

As described above in the *Exposure Analysis* section, from 2008 to 2018 there were eight interactions with leatherback sea turtles, and all of those were associated with a fishery set. We anticipate that 1 (95th percentile: 2) leatherback sea turtles will interact with the United States WCPO purse seine fishery per year with a maximum 5-yr running average of 0.48 (i.e. it is anticipated that zero leatherback sea turtles will be captured in some years, with up to one in other years for an average of 0.48). Seven of the leatherback sea turtles observed interacting with the United States WCPO purse seine fishery were released alive and uninjured (87%; Table 33), while one individual, or 13%, was released injured. The one injured individual sustained a wound on its shoulder due to the interaction with the net sack. All of the captured leatherback sea turtles were reported as landed which means that they were pursed in the net with the remainder of the catch, likely resulting in forced submergence and exposure to crushing for an unknown amount of time, before being brailed out of the net. At a minimum these processes would result in elevated stress levels and potentially turtles may have sustained injuries that were not apparent to the observer. The response to the exposures to the fishery will range from primarily behavioral, with a temporary, short-term interruption of behavioral patterns such as foraging, migrating, or socializing to sub-lethal impacts that can result in delayed growth or reproduction.

Using the mortality coefficients of Ryder et al. (2006) as described above, of the eight captured animals an estimated zero to one leatherback sea turtle died over 11 years in the United States WCPO purse seine fishery (Table 71). The remaining seven to eight leatherback sea 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 71. Estimated number of leatherback sea turtle mortalities by release condition for interactions with the United States WCPO purse seine fishery from 2008 through 2018. Numbers are based on the total number of interactions and injury mortality coefficients of Ryder et al. (2006) as described in the text. Total anticipated mortality rates for the range of mortality coefficients considered provided in the last row.

Release Condition	Number of Turtles Observed	Anticipated Proportion of Captures per Release Condition	Mortality Coefficient Low	Mortality Coefficient High
Alive and healthy (A1)	7	0.87	0	0.15
Alive but injured (A2)	1	0.13	0.10	0.35
Alive but unlikely to survive (A3)	0	0	1	1
Dead (D)	0	0	1	1

Grand Total	8	1	**Total Mortality Rate – Low:** 0.013	**Total Mortality Rate – High:** 0.175

We applied the estimated mortality rates from Table 71 to the anticipated exposure numbers for leatherback sea turtles (Table 71). We calculated the mortality for the mean and 95th percentile for the annual interaction predictions for leatherback sea turtles as well as the maximum 5-yr running average and running sum (Table 72). NMFS predicts that the United States WCPO purse seine fishery would capture 1 (95th percentile: 2) leatherback sea turtles each year. The estimated number of leatherback sea turtles that would likely die from their capture would be 0 (95th percentile: 0) in any given year (Table 72). Based on the maximum 5-yr running sum, we anticipate no mortalities for leatherback sea turtles over 5 years. In spite of this, we considered one mortality in our analysis as this species has died in sets from the greater WCPO purse seine fishery and it would therefore be reasonable to assume an animal could drown during haulback.

We would also expect sublethal responses from leatherback sea turtles that are captured, landed and released alive. The responses of animals may range from those that are temporary in nature such as elevated stress levels to more significant injuries that may affect feeding, movement, or even breeding success. These effects may decline over time as the surviving turtles heal from their injuries, although the extent to which such sublethal injuries occur or persist are not known.

Table 72. Anticipated mortalities for leatherback sea turtles interacting with the United States WCPO purse seine fishery. Anticipated mortality estimates are provided for the estimated mean and 95th percentile of leatherback sea turtle estimated exposures in terms of anticipated annual interactions with fishing gear and the maximum 5-year running average and running sum. Number outside of parentheses are the anticipated mortalities based on the low mortality coefficients in Table 71, numbers inside the parentheses are based on the high mortality coefficients in Table 71.

Bayesian Inference Model and Nonparametric Bootstrap		
	Annual Mean	**Annual 95th percentile**
Predicted Exposures	0 (0)	0 (0)
Maximum Running Average and Sum		
	5-yr Running Average	**5-yr Running Sum**
Predicted Exposures	0 (0)	0 (0)

South Pacific Loggerhead Sea Turtle

As described above in the *Exposure Analysis* section, from 2008 to 2018 there were 68 interactions with loggerhead sea turtles, 65 of those associated with a fishery set and three associated with a FAD not involving a set. We anticipate that 5 (95th percentile: 8) South Pacific loggerhead sea turtles will interact with the United States WCPO purse seine fishery per year with a maximum 5-yr running average of 4.96. Most South Pacific loggerhead sea turtles interacting with the United States WCPO purse seine fishery were released alive and uninjured (76%;), while 11 individuals, or 16%, were released injured and five were released dead or dying (8%). While we do not have details on all of the injuries, two of them went through the power block during the haul and were tangled in the net, one was brailed, and one was judged distressed. There are no details on the dying sea turtle but all injured and dying turtles, and dead sea turtles were landed. In addition, the majority of the uninjured loggerhead sea turtles were reported as landed which means that they were pursed in the net with the remainder of the catch, likely resulting in forced submergence and exposure to crushing for an unknown amount of time, before being brailed out of the net. At a minimum these processes would result in elevated stress levels and potentially turtles may have sustained injuries that were not apparent to the observer. The response to the exposures to the fishery will range from primarily behavioral, with a temporary, short term interruption of behavioral patterns such as foraging, migrating, or socializing to sub-lethal impacts that can result in delayed growth or reproduction to death, either at the vessel or post-release.

Using the mortality coefficients of Ryder et al. (2006) as described above, of the 68 captured animals an estimated six to 13 South Pacific loggerhead sea turtles died over 11 years in the United States WCPO purse seine fishery (Table 73). The remaining South Pacific loggerhead sea 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 73. Estimated number of loggerhead sea turtle mortalities by release condition for interactions with the United States WCPO purse seine fishery from 2008 through 2018. Numbers are based on the total number of interactions and injury mortality coefficients of Ryder et al. (2006) as described in the text. Total anticipated mortality rates for the range of mortality coefficients considered provided in the last row.

Release Condition	**Number of Turtles Observed**	**Anticipated Proportion of Captures per Release Condition**	**Mortality Coefficient Low**	**Mortality Coefficient High**
Alive and healthy (A1)	52	0.76	0	0.10

Alive but injured (A2)	11	0.16	0.05	0.25
Alive but unlikely to survive (A3)	2	0.03	1	1
Dead (D)	3	0.05	1	1
Grand Total	68	1	**Total Mortality Rate – Low:** 0.082	**Total Mortality Rate – High:** 0.190

We applied the estimated mortality rates from Table 73 to the anticipated exposure numbers for South Pacific loggerhead sea turtles (Table 74). We calculated the mortality for the mean and 95th percentile for the annual interaction predictions for South Pacific loggerhead sea turtles as well as the maximum 5-yr running average and running sum (Table 74). NMFS predicts that the United States WCPO purse seine fishery would capture 5 (95th percentile: 8) South Pacific loggerhead sea turtles each year. The estimated number of South Pacific loggerhead sea turtles that would likely die from their capture would be 0 to 1 (95th percentile: 1 to 2) in any given year (Table 74). Based on the maximum 5-yr running sum, we anticipate that between 2 and 5 South Pacific loggerhead sea turtles will die over 5 years (Table 74).

We would also expect sublethal responses from South Pacific loggerhead sea turtles that are captured, landed and released alive. The responses of animals may range from those that are temporary in nature such as elevated stress levels to more significant injuries that may affect feeding, movement, or even breeding success. These effects may decline over time as the surviving turtles heal from their injuries, although the extent to which such sublethal injuries occur or persist are not known.

Table 74. Anticipated mortalities for loggerhead sea turtles interacting with the United States WCPO purse seine fishery. Anticipated mortality estimates are provided for the estimated mean and 95th percentile of loggerhead sea turtle posterior estimated exposures in terms of anticipated annual interactions with fishing gear (inclusive of FAD interactions) and the maximum 5-year running average and running sum. Number outside of parentheses are the anticipated mortalities based on the low mortality coefficients in Table 73, numbers inside the parentheses are based on the high mortality coefficients in Table 73.

Bayesian Inference Model		
	Annual Mean	**Annual 95th percentile**
Predicted Exposures	0 (1)	1 (2)
Maximum Running Average and Sum		

	5-yr Running Average	5-yr Running Sum
Predicted Exposures	0 (1)	2 (5)

Olive Ridley Sea Turtle

As described above in the *Exposure Analysis* section, between 2008-2018 there were 57 interactions with olive ridley sea turtles, 54 of those associated with a fishery set and three associated with a FAD not involving a set. We anticipate that 4 (95th percentile: 7) olive ridley sea turtles will interact with the United States WCPO purse seine fishery per year with a maximum 5-yr running average of 6.29. Most olive ridley sea turtles interacting with the United States WCPO purse seine fishery were released alive and uninjured (74%; Table 40), while 15 individuals, or 26%, were released injured, none were released dead or dying. While we do not have details on all of the injuries, three of them went through the power block during the haul and were tangled in the net, two were removed from the brail, one was tangled in the net and sustained an injury to a front flipper, and one was considered stressed. One of the injured turtles that was removed from the brail was mishandled by the crew and thrown to forcefully to the deck, injuring the turtle. The remaining two were judged to be in a distressed condition by the observer. In addition, the majority of the uninjured olive ridley sea turtles were reported as landed which means that they were pursed in the net with the remainder of the catch, likely resulting in forced submergence and exposure to crushing for an unknown amount of time, before being brailed out of the net. At a minimum these processes would result in elevated stress levels and potentially turtles may have sustained injuries that were not apparent to the observer. The response to the exposures to the fishery will range from primarily behavioral, with a temporary, short-term interruption of behavioral patterns such as foraging, migrating, or socializing to sub-lethal impacts that can result in delayed growth or reproduction. From 2008 to 2018, no olive ridley sea turtles have been reported as dead or dying but it is possible that some of the injured turtles died post-release.

Using the mortality coefficients of Ryder et al. (2006) as described above, of the 57 captured turtles, an estimated one to eight olive ridley sea turtles died over 11 years in the United States WCPO purse seine fishery (Table 75). The remaining olive ridley sea 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 75. Estimated number of olive ridley sea turtle mortalities by release condition for interactions with the United States WCPO purse seine fishery from 2008 through 2018. Numbers are based on the total number of interactions and injury mortality coefficients of Ryder et al. (2006) as described in the text. Total anticipated mortality rates for the range of mortality coefficients considered provided in the last row.

Release Condition	**Number of Turtles Observed**	**Anticipated Proportion of Captures per Release Condition**	**Mortality Coefficient Low**	**Mortality Coefficient High**
Alive and healthy (A1)	42	0.74	0	0.10
Alive but injured (A2)	15	0.26	0.05	0.25
Alive but unlikely to survive (A3)	0	0.0	1	1
Dead (D)	0	0.0	1	1
Grand Total	57	1	**Total Mortality Rate – Low:** 0.013	**Total Mortality Rate – High:** 0.14

We applied the estimated mortality rates from Table 75 to the anticipated exposure numbers for olive ridley sea turtles (Table 76). We calculated the mortality for the mean and 95th percentile for the annual interaction predictions for olive ridley sea turtles as well as the maximum 5-yr running average and running sum (Table 76). NMFS predicts that the United States WCPO purse seine fishery would capture 4 (95th percentile: 7) olive ridley sea turtles each year. The estimated number of olive ridley sea turtles that would likely die from their capture would be 0 to 1 (95th percentile: 0 to 1) in any given year (Table 76). Based on the maximum 5-yr running sum, we anticipate that between 0 and 4 olive ridley sea turtles will die over 5 years (Table 76).We would also expect sublethal responses from olive ridley sea turtles that are captured, landed and released alive. The responses of animals may range from those that are temporary in nature such as elevated stress levels to more significant injuries that may affect feeding, movement, or even breeding success. These effects may decline over time as the surviving turtles heal from their injuries, although the extent to which such sublethal injuries occur or persist are not known.

Table 76. Anticipated mortalities for olive ridley sea turtles interacting with the United States WCPO purse seine fishery. Anticipated mortality estimates are provided for the estimated mean and 95th percentile of olive ridley sea turtle posterior estimated exposures in terms of anticipated annual interactions with fishing gear (inclusive of FAD interactions) and the maximum 5-year running average and running sum. Number outside of parentheses are the anticipated mortalities based on the low mortality coefficients in Table 75, numbers inside the parentheses are based on the high mortality coefficients in Table 75.

Bayesian Inference Model and Nonparametric Bootstrap		
	Annual Mean	**Annual 95th percentile**
Predicted Exposures	0 (1)	0 (1)
Maximum Running Average and Sum		
	5-yr Running Average	**5-yr Running Sum**
Predicted Exposures	0 (1)	0 (4)

Green Sea Turtle

As described above in the *Exposure Analysis* section, from 2008 to 2018 there were 97 interactions with green sea turtles, 87 of those associated with a fishery set and 10 associated with a FAD not involving a set. We anticipate that 7 (95th percentile: 11) green sea turtles will interact with the United States WCPO purse seine fishery per year with a maximum 5-yr running average of 6.26. Most green sea turtles interacting with the United States WCPO purse seine fishery were released alive and uninjured (79%; Table 44), while 14 individuals, or 15%, were released injured, and six (6%) were released dead or dying. While we do not have details on all of the injuries, three of them entangled in the net, disentangled and were considered in distress upon release. One was landed and considered in distress upon release. One of the dying green sea turtles was brailed into the fish hopper before being removed and released, the observer reports that the turtle sank upon release. In addition, the majority of the uninjured olive ridley sea turtles were reported as landed which means that they were pursed in the net with the remainder of the catch, likely resulting in forced submergence and exposure to crushing for an unknown amount of time, before being brailed out of the net. At a minimum these processes would result in elevated stress levels and potentially turtles may have sustained injuries that were not apparent to the observer. The response to the exposures to the fishery will range from primarily behavioral, with a temporary, short term interruption of behavioral patterns such as foraging, migrating, or socializing to sub-lethal impacts that can result in delayed growth or reproduction to death, either at the vessel or post-release.

Using the mortality coefficients of Ryder et al. (2006) as described above, of the 97 captured animals an estimated 7 to 16 green sea turtles died over 11 years in the United States WCPO purse seine fishery (Table 77). The remaining green sea 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 77. Estimated number of green sea turtle mortalities by release condition for interactions with the United States WCPO purse seine fishery from 2008 through 2018. Numbers are based on the total number of interactions and injury mortality coefficients of Ryder et al. (2006) as described in the text. Total anticipated mortality rates for the range of mortality coefficients considered provided in the last row.

Release Condition	**Number of Turtles Observed**	**Anticipated Proportion of Captures per Release Condition**	**Mortality Coefficient Low**	**Mortality Coefficient High**
Alive and healthy (A1)	77	0.79	0	0.10
Alive but injured (A2)	14	0.15	0.05	0.25
Alive but unlikely to survive (A3)	4	0.04	1	1
Dead (D)	2	0.02	1	1
Grand Total	97	1	**Total Mortality Rate – Low:** 0.059	**Total Mortality Rate – High:** 0.17

We applied the estimated mortality rates from Table 77 to the anticipated exposure numbers for green sea turtles. We calculated the mortality for the mean and 95th percentile for the annual interaction predictions for green sea turtles as well as the maximum 5-yr running average and running sum (Table 78). NMFS predicts that the United States WCPO purse seine fishery would capture 7 (95th percentile: 11) green sea turtles each year. The estimated number of green sea turtles that would likely die from their capture would be 0 to 1 (95th percentile: 1 to 2) in any given year (Table 78). Based on the maximum 5-yr running sum, we anticipate that between 2 and 6 green sea turtles will die over 5 years (Table 78).

We would also expect sublethal responses from green sea turtles that are captured, landed and released alive. The responses of animals may range from those that are temporary in nature such as elevated stress levels to more significant injuries that may affect feeding, movement, or even breeding success. These effects may decline over time as the surviving turtles heal from their injuries, although the extent to which such sublethal injuries occur or persist are not known.

Table 78. Anticipated mortalities for green sea turtles interacting with the United States WCPO purse seine fishery. Anticipated mortality estimates are provided for the estimated mean and 95th percentile of green sea turtle posterior estimated exposures in terms of anticipated annual interactions with fishing gear (inclusive of FAD interactions) and the maximum 5-year running average and running sum. Number outside of parentheses are the anticipated mortalities based on the low mortality coefficients in Table 77, numbers inside the parentheses are based on the high mortality coefficients in Table 77.

Bayesian Inference Model and Nonparametric Bootstrap		
	Annual Mean	**Annual 95th percentile**
Predicted Exposures	0 (1)	1 (2)
Maximum Running Average and Sum		
	5-yr Running Average	**5-yr Running Sum**
Predicted Exposures	0 (1)	2 (6)

5.3.2.2 Elasmobranchs

As with sea turtles and other species under consideration in this biological opinion, exposure to elasmobranchs by the fishery is primarily through entrapment by encirclement. All interactions with elasmobranchs occur due to this initial mechanism of exposure. Hutchinson et al. (2015) suggested that the action of simply encircling sharks will not affect their survival. However, subsequent actions reflecting measures taken to remove sharks and rays from the purse or during sorting of the catch can alter the overall outcome for the individual (Poisson et al. 2012). Sharks or giant manta rays may be released alive, injured, or dead.

Hall and Roman (2013) discussed lifting techniques used by purse seine fleets to remove manta rays from the deck, which include puncturing the animal’s cephalic fin to pass a rope or bridle through the tissue to lift the animal and return it to the ocean. Hook pulleys are also passed through the gills to lift the animals from the deck (Hall and Roman 2013). Tissue and muscle damages, tearing injuries, and death (either acute or delayed) can occur as a result from these types of interactions. Injuries to the gills can cause minor bleeding to acute mortality, and if the animal is returned to the water alive may affect the animal’s ability to respire potentially resulting in delayed mortality. Elasmobranchs lifted out of the water, or present on deck during the sorting process, can also suffer from acute hypoxia due to exposure to air (Cook et al. 2019). Hypoxic conditions can also occur as the net is retrieved, progressively reducing the area fish have to swim until they are no longer able to do so thereby reducing the available oxygen in the surrounding water from the large number of fish in close proximity with increased respiratory rates (Raby et al. 2014; Cook et al. 2019). Cook et al. (2019) further review other injuries, barotrauma (fish being pulled from depth too quickly- resulting in organ damage or death), exhaustion, and predation effects that occur to fish from capture stress. Specifically related to purse seine operations, Cook et al. (2019) rated capture stressors low to variable for actual capture activities, however handling and release techniques can result in high stressor severity particularly related to hypoxia and exhaustion. Additionally, elasmobranchs like the oceanic whitetip, scalloped hammerhead sharks, and giant manta ray are obligate ram ventilators and must move forward to maintain oxygen requirements. If they are trapped, confined by other catch, or entangled, acute mortality could occur. Dapp et al. (2015) noted that obligate ram-ventilating elasmobranchs suffer from relatively rapid asphyxiation.

Limited studies exist for purse seine operations in respect to shark mortality estimates. No data exists for oceanic whitetip sharks or Indo-West Pacific scalloped hammerhead sharks; therefore, we examined the effect of the fisheries on surrogate species that are relevant to the listed species. However, we only found only one such study that may be pertinent for the Indo-West Pacific scalloped hammerhead shark and it has a very small sample size of only three sharks. As with other fisheries, many variables affect the outcome of any particular interaction. While purse seine fishing results in higher mortality of elasmobranchs than other fisheries, mortality rates are species-specific and much of the research is focused on underlying physiology for handling stress, and some species may not be physiologically similar in resilience.

Physiological responses and effects to sharks from stress associated with capture have been studied in other shark species. These studies reveal adverse reactions to sharks' fitness after interaction with the gear. Sudden and delayed mortality in individuals of multiple other species of sharks across different ocean basins have been quantified. However, the literature suggests that conspecific sharks generally respond adversely to capture and handling expectations are the same for both shark species. However, this and subsequent sections will provide an overview of the best available scientific and commercial data and is a detailed overview of information found in Ellis et al. (2017) with additional references as noted.

Ellis et al. (2017) reviewed 6 studies (De Silva et al. 2001; Capietto et al. 2014; Poisson et al. 2014a; Hutchinson et al. 2015; Eddy et al. 2016; Francis and Jones 2017) pertaining to mortality metrics in various species of elasmobranchs from purse seine fisheries (see Table II, p. 691). Francis and Jones (2017) is the only study that assessed physiological responses in rays and will be discussed in further detail in the following *Entanglement* discussion as giant manta rays are more susceptible to injury from those stressors than entrapment.

Eddy et al. (2016) reported average at-vessel mortality for silky sharks (*Carcharhinus falciformis*; 59%, range 23-70), and for scalloped hammerhead sharks (*Sphyrna lewini*; 0%). Post release mortality was 62% for silky sharks and all three PSAT tagged scalloped hammerhead shark individuals died immediately after release resulting in a 50% total mortality rate. Due to the low number of animals tagged, the authors suggested more research into this species would be recommended. While this study provides some support for our assumption of 100% mortality in the Indo-West Pacific scalloped hammerhead shark, due to the small sample size and the expected handling practices in this specific fishery, we felt it prudent not to rely on this study alone. Eddy et al. (2016) also determined that size of catch dictated shark mortality rates with larger catches having higher mortality rates in sharks (55% at vessel mortality in 30 mt sets; 87% at vessel mortality in 100 mt sets). Overall or total mortality, the combination of at-vessel mortality and post release mortality rates, was estimated to be 92% in the EPO for silky sharks (Eddy et al. 2016). Similar results (81-85% total mortality) were found in purse seine operations in the West and Central Pacific and Indian Ocean (Filmalter et al. 2012; Poisson et al. 2011, 2014a; Hutchinson et al. 2015; Eddy et al. 2016). Eddy et al. (2016) suspects that inability to ram ventilate may have been a major contributing factor which affected the outcome for these individuals as no internal or external damage was noted. Eddy et al. (2016) concluded the time an individual remained in the sack increased the expected mortality rates for these two species (Figure 77).

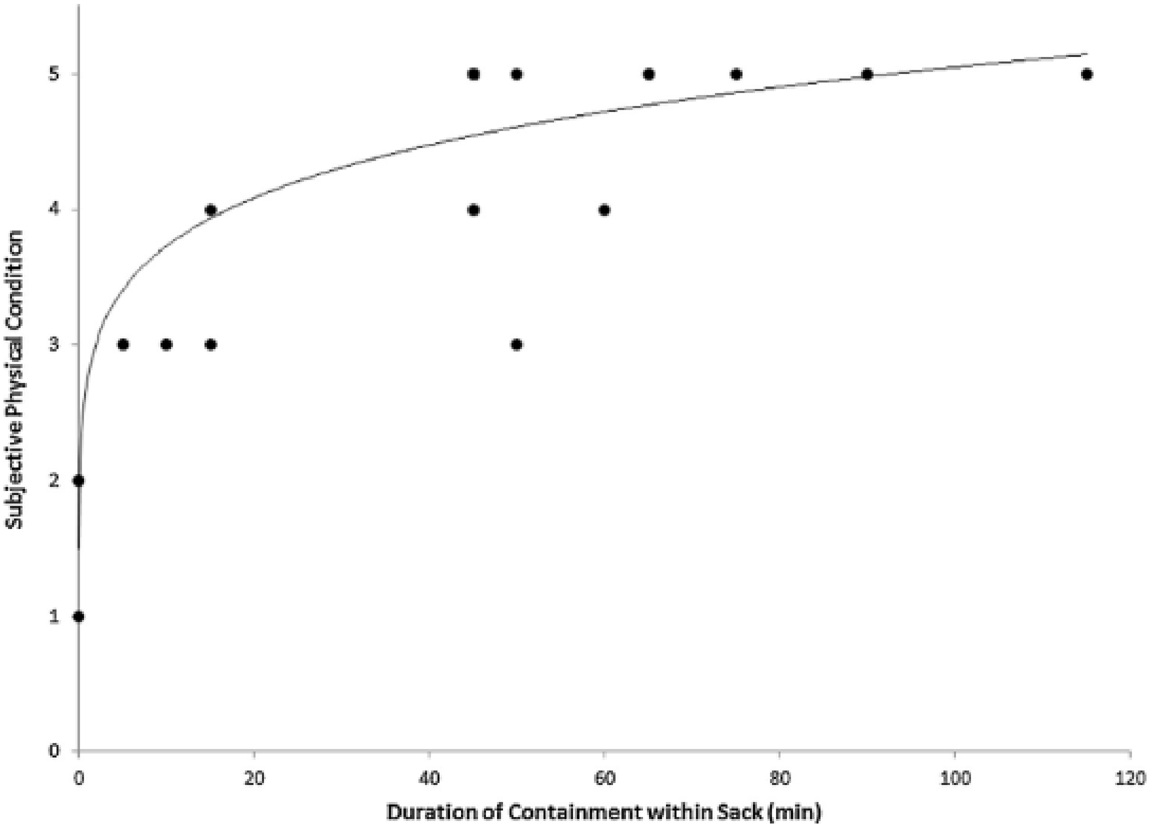


Figure 77. Subjective physical condition of sharks captured by a commercial purse seine over drifting FADs in the Eastern Pacific Ocean with increasing time within the sack. Line represents the change in condition as a function of sack time. Condition = 2.75 (Duration in minutes)$^{0.13}$. R^2= 0.79, $p < 0.05$. (Y axis: 1 = Excellent; 2 = Good; 3 = Fair; 4= Poor; 5 = Moribund or Dead; Eddy et al. 2016).

This is further supported by evidence provided by Hutchinson et al. (2015), who assessed physiological response in silky sharks to capture stress in United States WCPO purse seine operations. They concluded lactate was a predictor of mortality and determined that those sharks with 11.3 mmol l^{-1} and higher were considered moribund (Figure 78; Hutchinson et al. 2015). Many studies have been conducted looking at physiological responses of pelagic sharks, however, these studies provide the best representation for sharks this specific fishery.

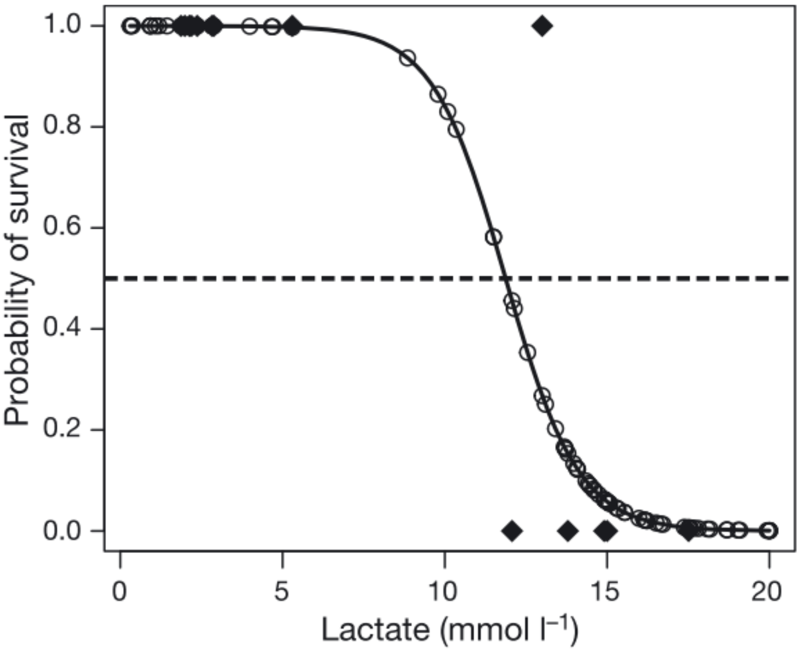


Figure 78. Survival probability curve using lactate mmol 1^{-1} concentrations for satellite tagged silky sharks (♦) that either survived the fishing interaction (1) or died post release (0). Fitted probabilities (o) sharks that had blood drawn but were not satellite tagged. Probability of survival (dashed line = 0.5); (silky) sharks predicted to have survived (above line) the fishing event or suffered mortality (below line; Hutchinson et al. 2015).

There are no data on post-release survival of giant manta rays. However, given our conservative assumption that all giant manta rays released with unknown condition codes are dead, resulting in an estimated at-vessel mortality rate of 96%, we use this value as the anticipated mortality rates of future giant manta ray interactions with the United States WCPO purse seine fishery.

We would expect animals that are released alive and survive may exhibit sublethal responses from their capture. The responses of oceanic whitetip sharks, Indo-West Pacific scalloped hammerhead sharks, and giant manta rays that survive may range from those that are temporary such as elevated stress levels, to injuries that are more significant such as those that may affect feeding, movement, or even breeding success. These effects may decline over time if the surviving sharks heal from their injuries, although the extent to which such sublethal injuries occur or persist are not known.

Oceanic Whitetip Shark

As noted previously, we expect that a portion of the animals assigned to the *unidentified shark* classification are oceanic whitetip sharks that could not be identified to species in the field. The resulting future anticipated exposure estimates are listed in Table 50. For the remainder of this evaluation, we will focus on using the upper adjusted estimates of oceanic whitetip sharks likely to be exposed to the United States WCPO purse seine fishery in our jeopardy analysis. Therefore, we expect 88 (95th percentile: 160) individual oceanic whitetip sharks will interact with the fishery in a given year, and of those, 75 (95th percentile: 135) would be expected to die from their interactions (Table 79). As previously discussed in the *Exposure Analysis* section, survival metrics specific to oceanic whitetip sharks are unknown, therefore a surrogate species, the silky shark was used to determine post release and total mortality for the purposes of this analysis. The predicted future interactions are based on the assumption of 3,100 annual sets with 1,581 of those being FAD sets. The fishery has an observed 47% at vessel mortality, however as previously noted in the *Exposure Analysis*, this is just the at-vessel mortality and does not account for post release mortality which would be expected to occur. Therefore, we conducted a systematic review of the literature to assess what metrics would be relevant to the species from operational processes (e.g. gear, methodology, handling, etc.) which occur in this fishery. It is likely the additional 679 sharks identified as A2 would die from current handling practices resulting in an estimated mortality value of 77%. However, this would not account for other individuals that may die as result of their interaction and is considered a minimum mortality value.

Hutchinson et al. (2015) determined high mortality in silky sharks in this specific fishery, particularly once sharks were brailed. Hutchinson et al. (2015) concluded that sharks landed after brailing will not reduce fishing mortality and predicted a 6.67% survival rate for sharks that are brailed. We would expect oceanic whitetip sharks to suffer the same physiological responses from hypoxia and crushing injuries during this process as they would be handled by the industry in the same manner as silky sharks, and considering they have the same respiratory mode. However, as previously stated, we do not have post release mortality studies on this species in this fishery. Poisson et al. (2011) also reported a 62% at vessel mortality value for silky sharks for French purse seiners in the Indian Ocean with an additional 50% loss of individuals after release. They estimated an overall mortality of 81% for silky sharks. We chose to use Hutchinson et al. (2015) over this value as this research was conducted *in this specific fishery* and operations in the French fleet may differ. However, it provides a comparable metric across fisheries and is consistent with the evidence presented for this fishery. Therefore, we use the mortality values from Hutchinson et al. (2015) as the best scientific and commercial data available for this consultation.

First we assessed the number of individuals expected to die at the vessel based on the number of observed interactions that were identified as dead in the data not including prorated unidentified sharks or those that were alive but released and expected to die (A3s). Of the 88 individuals that are caught in a given year, we estimate 42 sharks are dead at the vessel upon initial discovery by the crew/observer. Considering the overall or total mortality, the combination of at-vessel mortality and post release mortality rates, which was estimated to be 84.2% in the United States WCPO purse seine fishery for silky sharks (Hutchinson et al. 2015) we applied this rate to the total number of estimated interactions. This results in the expectation that of the 88 (95th percentile: 160) individuals caught, a total of 75 (95th percentile: 135) of those sharks would be expected to die as a result of their interaction with the fishery annually. Lastly, when considering the 5-year sum anticipated interactions of 514; we would expect 433 of those individuals would die with the other 81 being released with varying degrees of injuries over that time interval.

Table 79. Anticipated future exposure levels for oceanic whitetip sharks with unidentified records prorated with projected mortality for the United States WCPO purse seine fishery.

Estimated United States WCPO Purse Seine Oceanic Whitetip Shark Mortality		
Rate	**Estimated At Vessel Mortality (47.0%; #Indvs)**	**Estimated Total Mortality (84.2%; #Indvs)**
Annual	42	75
Annual 95th Percentile	76	135
5-Year Running Mean	49	87
5-Year Max Running Sum	242	433

Indo-West Pacific Scalloped Hammerhead Shark

As noted previously, we expect that all Indo-West Pacific scalloped hammerhead sharks would be expected to die from interactions with purse seine fishery. These sharks are not released before sacking the net or brailing occurs. Meaning the sharks will be landed and due to their biological requirement to move to obtain oxygen, will not be able to circulate water over their gills, and will die. Additionally, this species is much more sensitive to others in this regard. For the remainder of this evaluation, we will focus on using the upper adjusted estimates of Indo-West Pacific scalloped hammerhead sharks likely to be exposed to the United States WCPO purse seine fishery in our jeopardy analysis. Therefore, we expect 2 (95th percentile: 2) individual Indo-West Pacific scalloped hammerhead sharks will interact with the fishery in a given year, and all of those would be expected to die from their interactions. Lastly, when considering the 5-year running sum anticipated interaction level of 10; we would expect all individuals to die over this 5-year time interval given the current handling practices of the fleet and considering the poor resiliency of the species.

Giant Manta Ray

Typically, we would expect that a portion of the animals assigned to the *Mobula* nei classification are likely giant manta rays that could not be identified to species; however, the best available information suggests that even individuals identified by observers as giant manta ray are not likely to contain only giant manta ray. We presented our estimates of the number of individual giant manta rays that we expect would be exposed to the United States WCPO purse seine fishery in Table 65. With no misidentification, we would expect 165 (95th percentile: 199) individual giant manta rays will interact with the fishery in a given year. Assuming that 96% of giant manta rays will die as a result of their exposure (capture) as described in Section 5.2.1.3, we would expect 158 (95th percentile: 191) giant manta rays would die from their interactions each year (Table 80). As explained earlier in this biological opinion, we expect that these are likely overestimates of the true number of giant manta rays encountered in this fishery. Our best reasoned estimate is that 75% of giant manta ray captures were misidentified (that they were not giant manta ray) and that 75% fewer of the unidentified mobulids category are giant manta ray than were assigned to this category in the prior analysis. As such, we believe our best estimate of the number of giant manta ray that would be captured in this fishery is 45 each year, and 43 would die as a result of their capture and handling. This results in a 5-year running mean of 47 interactions or exposures resulting in 45 mortalities, and a 5-year running sum maximum of 236 captures, and 227 mortalities of giant manta rays. While we analyzed a scenario in which we assumed a 50% misidentification rate, the data available including the weight analysis (Figure 52) indicates that this value may be much higher and we focus our analysis on a misidentification rate of 75%.

Table 80. Anticipated future incidental captures (exposures) and mortalities of giant manta rays using aggregate unadjusted estimates and under a 75% misidentification scenario for the United States WCPO purse seine fishery.

Estimated Numbers of Giant Manta Ray Captured (exposures) and Killed in the United States WCPO Purse Seine				
Rate	**Exposures**	**Estimated Mortality**	**Exposures with 75% misidentification**	**Estimated Mortality with 75% misidentification**
Annual	165	158	45	43
Annual 95th Percentile	199	191	50	48
5-Year Running Mean	187.4	179.9	47	45.2
5-Year Max Running Sum	937	900	236	227

5.3.3 Entanglement in FADs

As discussed in the *Environmental Baseline* section, the 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 breathe; they may also be predated by sharks. Direct impacts to oceanic whitetip sharks entangled underneath FADs are likely stress, injury, or death from predation or lack of oxygen.

As noted in the *Exposure Analysis* section, shark entanglement in FADs has been documented only for *Carcharhinus* species (i.e. silky and oceanic whitetip sharks; Chanrachkij and Loog-on 2003; Filmalter et al. 2013; Murua et al. 2017) and Filmalter et al. (2013) indicate that those species are the only shark species known to regularly associate with floating objects. Scalloped hammerhead sharks are generally rare in the WCPO purse seine fishery, with all hammerhead shark species representing only 0.03% of all elasmobranch bycatch (Molony 2007). When hammerhead sharks do interact with purse seine fisheries, they are caught at higher rates on unassociated sets as compared to associated sets (Torres-Irineo et al. 2014). There are no records of FAD entanglement for scalloped hammerhead sharks in the WCPO purse seine fishery. Similarly, giant manta rays are caught less frequently on associated sets (Torres-Irineo et al.

2014), for the United States WCPO purse seine fishery we found 37% of giant manta ray bycatch occurred on FAD or other floating object sets. Hence, we do not consider entanglement in FADs a risk to Indo-West Pacific scalloped hammerhead sharks or giant manta rays and do not consider this stressor further for those species.

As noted in the *Exposure Analysis* section, there were no observations of ESA-listed whales associated with FADs in the WCPO purse seine fishery observer data. Hence, we do not have enough information to assess the risk to the marine mammals from FAD entanglement considered in this biological opinion.

5.3.3.1 Sea Turtles

Sea turtles are particularly prone to being entangled in netting because of their body shape and behavior. Records of stranded or entangled sea turtles reveal that fishing debris and lines can wrap around the neck, flippers, or body of a sea turtle and severely restrict swimming or feeding. Depending on the location where the turtle is entangled in the net or FAD, forced submergence could occur resulting in injury or death. Turtles may attempt to escape and become more entangled.

As discussed in the *Environmental Baseline* section, the ropes and nets associated with FADs used by the United States WCPO purse seine fishery pose an entanglement risk to sea turtles (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 breathe; they may also be predated by sharks.

Most (83%) of sea turtles observed in FAD interactions are considered uninjured (Table 81). 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 81).

Table 81. 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.

Release Condition					**-**	**-**
Species	**A1** **Alive and healthy**	**A2** **Alive but injured**	**A3** **Alive but unlikely to survive**	**D** **Dead**	**Total Observed Mortalities**	**Estimated Mortality Rate**
Green	8	2	0	0	0	0 to 0.1
Leatherback	0	0	0	0	0	0
Loggerhead	3	0	0	0	0	0
Olive Ridley	3	0	0	0	0	0
Hawksbill	7	0	0	1	1	0.13
Grand Total	21	2	0	1	1	-

The estimates of future interactions using Bayesian statistical inference techniques, as described in the *Approach to the Assessment* section of this biological opinion, encompass all interaction types considered in the analysis (i.e. FAD set, free set, FAD interaction, and unknown set type), and therefore the tables above for predicted exposures of sea turtles in the United States WCPO purse seine fishery are inclusive of exposures from FAD interactions (Table 70, Table 72, Table 74, and Table 78). Here, we highlight the subset of those anticipated exposures expected to occur as a result of FAD interactions in Table 82. The condition codes for all observed hawksbill, loggerhead, and olive ridley sea turtles interacting with FADs were either uninjured or dead and we used the proportions of uninjured or dead for each species to estimate anticipated mortalities, assuming 0% mortality for uninjured sea turtles (Table 81). We anticipate a mortality rate of 13% and that of the one to three annual mortalities anticipated for hawksbill sea turtles (Table 70), zero of them will be a result of FAD interactions (Table 82). No mortalities resulting from FAD interactions are anticipated for leatherback, South Pacific loggerhead or olive ridley sea turtles. The condition codes for observed green sea turtles interacting with FADs were uninjured (*n* = 8) or injured (*n* = 2). Given the nature of the observed interactions with FADs, we assume 0% mortality for turtles released uninjured. For the injured sea turtles, we use the same range of potential mortality as defined above, 5 to 25%, suggesting that zero to one (rounded from 0.5) injured green sea turtles died as a result of interacting with a FAD. We would therefore anticipate a mortality rate of 0% to 10% (1 mortality divided by 10 total interactions) for green sea turtles interacting with FADs. Hence no mortalities would be anticipated for green sea turtles interacting with FADs.

Table 82. 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.

	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	0
Grand Total			3	6	0	0

5.3.3.2 Oceanic Whitetip Sharks

Sharks can become entangled in the net wall or in dFAD materials. An entanglement as such, could cause the shark to perish 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. The literature on sharks captured in purse seines is primarily focused on the effects the entire catch event from when the shark is encircled until it's released, the handling method(s), and post release mortality metrics. Looking specifically at entanglement metrics, and not at vessel mortality or post release mortality, a few studies on silky sharks provide relevant metrics to these species.

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 quantified. 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).

Marine debris data compiled in NOAA's 2014 Marine Debris Program Report reveals several accounts of sharks entangled in natural fiber rope and monofilament which could be present due to the netting used in purse seines or through materials present in dFADs. Other documented instances of entangled sharks include 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). The Miami Herald recently published an online article by Price (2019) showing a 7-foot shark which had a piece of plastic wrapped through its gills, almost resulting in mortality. As the shark grew, the plastic did not yield and caused extensive physical damage as apparent by the photographic evidence of the encounter (Price 2019). In general, entanglement could directly or indirectly interfere with a shark's mobility, causing impairment in feeding, breeding, or migration, or could cause mortality whether acute or delayed.

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. As such, this interaction was incorporated into the number of exposures as noted in the *Demographic Patterns of Exposure* above. Animals entangled in FADs can naturally degrade or be depredated on before the retrieval of the device (Filmalter et al. 2013). Simple online searches provide photographic evidence of sharks entangled in FADs and Filmalter et al. (2013) has identified this issue as a major source of mortality, at least in the Indian Ocean. However, quantification of these types of events in the WCPO is needed to assess the potential impacts to these populations.

5.4 Cumulative Effects

"Cumulative effects", as defined in the ESA implementing regulations, are limited to the effects of future state, tribal, local, or private actions that are reasonably certain to occur in the *Action Area* considered in this opinion (see 50 CFR 402.02). For an action to be considered reasonably certain to occur, it must be based on clear and substantial information, or otherwise have a firm basis to support a conclusion that a consequence of an action is likely. Some factors we consider when evaluating an action for potential cumulative effects and whether those effects are reasonably certain to occur include our past experiences from similar actions, existing plans for the activity or action, and hurdles, like economic and legal requirements, that must be met before the action can go forward (see 50 CFR 402.17). Future federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to Section 7 of the ESA.

NMFS searched for information on future State, tribal, local, or private actions that were reasonably certain to occur in the *Action Area*. Most of the *Action Area* is outside of territorial waters of the United States of America, which would preclude the possibility of future state, tribal, or local action that would not require some form of federal funding or authorization. NMFS conducted electronic searches of business journals, trade journals, and newspapers using Google scholar, *WorldCat*, and other electronic search engines. Those searches produced no evidence of future private action in the *Action Area* that would not require federal authorization or funding and is reasonably certain to occur.

While we considered various state managed vessel-based fisheries which exist in American Samoa that fish pelagic waters (up to 25 miles offshore). We do not believe they will overlap in geographical space for fishing activities and would only overlap when vessels from this fishery transit to American Samoan ports. 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). Nearshore (mostly recreational) fisheries such as shallow bottomfishing, reef trolling, spearfishing, whipping/casting, trapping, and netting also occur (Loomis et al. 2019). Again, we do not believe any of these fisheries would overlap where fishing activities from this fishery would occur. The same could be said for recreational boating around American Samoa as well. As a result, NMFS is not aware of any actions that are likely to occur in the *Action Area* during the foreseeable future.

6 INTEGRATION AND SYNTHESIS OF EFFECTS

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 effects of the authorization of the purse seine fishery as currently managed, which we expect would continue well into the future.

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 United States WCPO purse seine fishery poses to endangered fin whales, endangered sei whales, endangered sperm whales, endangered leatherback sea turtles, endangered South Pacific loggerhead sea turtles, Eastern Pacific green sea turtles, East Indian – West Pacific green sea turtles, Central West Pacific green sea turtles, Southwest Pacific green sea turtles, Central South Pacific green sea turtles, endangered Mexico breeding population of olive ridley sea turtles and threatened (other) populations of olive ridley sea turtles, threatened oceanic whitetip sharks, threatened giant manta rays, and threatened scalloped hammerhead sharks 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.

In the *Approach to the Assessment* section of this biological opinion, we stated that we focus our assessment on three variables in the jeopardy definition that determine a species likelihood of survival and recovery in the wild: reductions in the species' reproduction, numbers or distribution. We measure risks to individuals of endangered or threatened species using changes in the individuals' "fitness" or the individuals' growth, survival, annual reproductive success, and lifetime reproductive success. When we do not expect listed individuals exposed to an action's effects to experience reductions in fitness, we would not expect the action to have adverse consequences on the viability of the populations that those individuals represent or the species those populations comprise (Mills and Beatty 1979; Stearns 1992; Anderson 2000). As a result, if we conclude that listed animals are not likely to experience reductions in their fitness, we would conclude our assessment. If we conclude that one or more population is likely to experience an increase in its extinction probability (measured using probability of demographic, ecological, or genetic extinction) we assess the effect of that increase on the probable reproduction, numbers, or distribution of the species to determine whether the action appreciably reduces that species' likelihood of survival or recovery in the wild. For these analyses, we combine our knowledge of the patterns that accompanied the decline, collapse, or extinction of populations and species that have experienced these phenomena in the past supplemented by indicators of probable demographic performance produced by population viability analyses.

We recognize that an appreciable reduction in a species' likelihood of surviving and recovering in the wild is not necessarily a quantitative calculation. Rather, since we rarely have information on the abundance and trends, growth rate, spatial structure or other indicators of population and species diversity to place numeric values on a species chances of persistence we use a combination of quantitative and qualitative analytical methods for our risk evaluation. It was our intent to build an analytical framework that appropriately considers the information available to each step of this analysis and uses the best method to evaluate and incorporate that information into our assessment. At the same time, it was important to appropriately insure that the federal action is not likely to jeopardize the continued existence of the endangered and threatened species, and to do so in a way that minimizes uncertainty about the potential impacts listed species can be expected to experience. Underestimating the number of listed individuals that might be exposed to the adverse effects of an action and the number of listed species expected to be "taken" as a result of that exposure poses the greatest risk for endangered and threatened species. The explicit mandate of Section 7(a)(2) of the ESA and congressional direction to provide the benefit of doubt to endangered and threatened species should make the choice of predictive metrics relatively simple: select the metric and interval that provides the greatest confidence of being correct and the lowest risk of being incorrect. The latter would (1) help federal agencies insure that their actions are not likely to jeopardize the continued existence of listed species (for example, by minimizing the chances of underestimating the number of listed species adversely affected by an action); and (2) provide endangered and threatened species with the benefit of the doubt in the face of uncertainty (by minimizing the probability of reaching an incorrect conclusion). This approach recognizes the inherent asymmetrical risk that may be borne by threatened and endangered species because extinction is irreversible and makes appropriate use of many types of best available scientific and commercial data, and is not limited or constrained by numeric information or models. We use such tools where they can best support the information and the analysis that are available, and use other qualitative methods like systematic reviews to compile and evaluate qualitative information into our assessment. As such, this assessment better incorporates available information that is otherwise poorly studied or integrated by quantitative methods.

Our analyses first consider an action's effects on the viability and resilience of the populations the action affects. Second, when a population declines to about 50 individuals: (1) it is not likely to increase in abundance before it becomes extinct; (2) any increases in abundance will be very short-lived; and (3) the persistence of the population depends on the reproductive success of a small proportion of individuals (Fagan and Holmes 2006). That same study suggests that these three phenomena can also affect populations numbering up to 300 individuals. Third, our analyses carefully consider the spatial structure of populations, patterns of immigration and emigration that connect populations, and the existence of population refugia because these will determine a species' resilience (Thomas and Jones 1993; Thomas and Kunin 1999; O'Grady et al. 2004). Finally, we rely on our knowledge of species that have declined, collapsed, and are now extinct and well as those that avoided extinction and those that recovered from endangerment. That knowledge allows us to consider general patterns that have resulted in extinction, avoided it, or that have allowed species to recover from endangerment.

Our analyses find that the proposed action—while it results in the death of individual endangered leatherback sea turtles, endangered North Pacific loggerhead sea turtles, threatened Eastern Pacific green sea turtles, threatened East Indian-West Pacific green sea turtles, endangered Central West Pacific green sea turtles, threatened Southwest Pacific green sea turtles, endangered

Central South Pacific green sea turtles, endangered Mexico breeding population of olive ridley sea turtles, and threatened (other) populations of olive ridley sea turtles, threatened Indo-West Pacific scalloped hammerhead shark, threatened oceanic whitetip sharks; endangered fin whales, endangered sei whales, endangered sperm whales,—has very small effects on the dynamics of the populations those individuals represent or the species those populations comprise. As a result, we conclude that the United States WCPO purse seine fishery does not appreciably reduce these species' likelihood of survival and recovery in the wild. Our examination of the numbers of individual giant manta ray captures that observers have identified interacting with this fishery gives us cause for concern. However, based on evidence available from the IATTC that giant manta ray are routinely misidentified, a lack of verified evidence of giant manta rays interacting with this fishery (e.g., photos or tissue samples, and the photos that we do have are another *Mobula* species, information to suggest that this fishery interacts with more rays that are smaller than the recorded birth weight of giant manta rays, and several other lines of reasoning explained in this biological opinion, we have concluded that the observer data is representative of the *Mobula* species generally, but the category identified as giant manta rays likely contains a significant number of other *Mobula* species. After examination of the observer data and considering these and other lines of reasoning, including consideration of a significant number of presumed subpopulation scenarios, we find that our most plausible analysis for this species suggests the action is likely to have small effects on the dynamics of the populations (or subpopulations those individuals represent or the species those populations comprise. As a result, we conclude that the United States WCPO purse seine fishery does not appreciably reduce the likelihood of survival and recovery of giant manta rays in the wild.

We explain the basis for these conclusions for each species in the following sections. These summaries integrate the results of the exposure and response analyses we presented earlier in this biological opinion with background information from the *Status of Listed Resources*, *Environmental Baseline,* and *Cumulative Effects* sections of this biological opinion to assess the potential risks that the operation of the United States WCPO purse seine fishery is likely to pose to endangered and threatened individuals, the population or populations those individuals represent, and the "species" as it was listed pursuant to the ESA of 1973, as amended.

6.1 Fin Whale

As described in the *Status of Listed Resources*, the fin whale is listed as endangered throughout its range and has a generally anti-tropical distribution centered in the temperate zones. Fin whales were potentially recorded in the Eastern Tropical Pacific in 1996 at a latitude that is within that of the United States WCPO purse seine fishery but relatively far to the east of the eastern-most range of the fishery (Stafford et al. 1999). The population structure of fin whales is poorly understood. In the North Pacific, the best minimum abundance estimates are for California/Oregon/Washington: 8,127 (Nadeem et al. 2016); Hawaii: 75 (Bradford et al. 2017); the Northeast Pacific: 2,554 (Moore et al. 2002; Zerbini et al. 2006) for a total minimum population estimate of 10,756 individuals. Although there is low confidence in the abundance estimate for fin whales in the southern hemisphere, the most current population estimate, calculated for 1978/1979, is 85,200 (no CV given) based on the history of catch trends (IWC 1979). The global population is estimated at 100,000 mature individuals with an increasing trend (NMFS 2019d).

As described in the *Environmental Baseline*, effects from commercial whaling have had the greatest impacts to the populations in the Pacific. From 1904 to 1975, there were 703,693 fin whales harvested in Antarctic whaling operations, and approximately 49,000 were harvested in the North Pacific between 1911 and 1985 (IWC 1990; Mizroch et al. 2009).

Whale migration, feeding, and breeding locations may be influenced by changing ocean currents and water temperatures, although the extent of potential change is unknown.

Given the estimated number of interactions with fin whales in the United States WCPO purse seine fishery from 2008-2018 (n = 14; 95% CI: [9, 19]), as described in the *Effects of the Action*, NMFS predicts future interaction levels, using the adjusted upper estimate, to be a mean of 0 and up to 1 at the 95th percentile annually. As described in the *Response Analysis*, the most significant threat to fin whales by the United States WCPO purse seine fishery is entrapment by gear which can lead to elevated stress levels and scraping wounds from breaking through the net. We do not anticipate any mortality from these interactions. We expect these interactions to lead to stress during the event and immediately following post release which may temporarily impact essential life functions such as feeding, breeding, or migration but are unlikely to affect an individual's lifetime fitness.

Based on the evidence available, a mean of 0 (95th percentile: 1) fin whales may be caught by the purse seine gear in any given year, with a maximum 5-year running average of 0.75 whales per year, or up to 8 whales over 10 years. The evidence available leads us to conclude that these whales will be freed from the net either by the fisherman releasing the net before pursing or from the whale breaking through the net. The best available minimum abundance estimate for fin whales in the North Pacific is 10,756 whales. Assuming the 0 (95th percentile: 1) interaction per year is from the North Pacific, this interaction rate would equal 0% (95th percentile: 0.0093%) of North Pacific fin whales. Over 10 years, 8 interactions would equal 0.074% of North Pacific fin whales. Similarly, the best available abundance estimate for the Southern hemisphere is 85,200 whales. Assuming the 0 (95th percentile: 1) interaction per year is from the Southern hemisphere, this interaction rate would equal 0% (95th percentile: 0.0012%) of the Southern hemisphere fin whales. Over 10 years, 8 interactions would equal 0.0094% of Southern hemisphere fin whales. As the global population is considered to be increasing (NMFS 2019d), these percentages likely overestimate the impact to the species at 10 and 20 years into the future.

Based on our analyses, the United States WCPO purse seine fishery may adversely affect the behavior of up to 1 fin whale annually. However, any behavioral response is unlikely to adversely affect the lifetime fitness of any of those individual fin whales that interact with this fishery. Therefore, we do not expect these interactions to have a measureable affect on fin whale populations in the Pacific Ocean. Because we do not anticipate impacts to fin whales in the Pacific Ocean, we also do not anticipate appreciable reductions in overall reproduction, abundance, or distribution of the fin whale population rangewide. For this reason, the effects of the proposed action are not expected to cause an appreciable reduction in the likelihood of survival and recovery of fin whales in the wild.

Because we do not anticipate impacts to fin whales in the Pacific Ocean, we also do not anticipate appreciable reductions in overall reproduction, abundance, or distribution of the fin whale population rangewide. For this reason, the effects of the proposed action are not expected to cause an appreciable reduction in the likelihood of survival and recovery of fin whales in the wild.

6.2 Sei Whale

As described in the *Status of Listed Resources*, the sei whale is listed as endangered throughout its range and the population structure is poorly understood. There are no trend assessments for the status of any populations in the Pacific. NMFS (2012b) estimates that there are 8,600 individuals in the North Pacific. In the Southern Hemisphere, the most recent population estimates range between 9,800 and 12,000 sei whales (no CV) (Mizroch et al. 1984; Perry et al. 1999).

As described in the *Threats to the Species* section of the *Status of Listed Resources* and in the *Environmental Baseline*, effects from commercial whaling have had the greatest impacts to the populations in the Pacific. Between 1910 and 1975, harvest of 152,233 sei whales occurred in commercial whaling activities in the Southern Hemisphere (Horwood 1987). About 63,500 sei whales were harvested during commercial whaling in the North Pacific between 1910 and 1987 (Committee for Whaling Statistics 1942; Scheffer and Slipp 1948; Rice 1977; Barlow et al. 1997). Japan has continued whaling for sei whales under the auspices of scientific whaling and captured 592 from 1988-2009 in the northwestern Pacific (IWC 2010), and 1,698 sei whales under their special permit in the North Pacific between 2001-2018 (IWC 2021).

The feeding range of sei whales is wide and consequently, it is likely that the sei whale may be more resilient to climate change, should it affect prey, than a species with a narrower range.

Given the estimated number of interactions with sei whales in the United States WCPO purse seine fishery from 2008-2018 (n = 96; 95% CI: [87, 140]), as described in the *Effects of the Action*, NMFS predicts future interaction levels, using the adjusted upper estimate, to be a mean of 3 and up to 7 at the 95th percentile annually, with a maximum 5-year running average of 6 whales and up to 60 whales over 10 years. As described in the *Response Analysis*, the most significant threat to sei whales by the United States WCPO purse seine fishery is entrapment by gear which can lead to elevated stress levels and scraping wounds from breaking through the net. We do not anticipate any mortality from these interactions. We expect these interactions to lead to stress during the event and immediately following post release, which may temporarily alter behaviors such as feeding, breeding, or migration but would not be expected to adversely affect the lifetime fitness of any of those individual sei whales that interact with this fishery.

Based on the evidence available, a mean of 3 (95th percentile: 7) sei whales could be caught by the purse seine gear in any given year. The evidence available leads us to conclude that these whales will be freed from the net either by the fisherman releasing the net before pursing or from the whale breaking through the net. The best available minimum abundance estimate for sei whales in the North Pacific is 8,600 whales. Assuming the 3 (95th percentile: 7) interactions per year are from the North Pacific, this interaction rate would equal 0.035% (95th percentile: 0.081%) of the North Pacific sei whales. Over 10 years, 60 interactions would equal 0.70% of North Pacific sei whales. Similarly, the best available minimum abundance estimate for the Southern hemisphere is 9,800 whales. Assuming the 3 (95th percentile: 7) interactions per year are from the Southern hemisphere, this interaction rate would equal 0.031% (95th percentile: 0.071%) of the Southern hemisphere sei whales. Over 10 years, 60 interactions would equal 0.61% of Southern hemisphere sei whales. While there are no trend estimates available for this species we do not expect the species would decline substantially in the future as the greatest threat to this species, whaling, has been removed. Therefore, we anticipate similar proportions of the population will be impacted 10 and 20 years into the future, and we would not expect these interactions would result in mortality.

Based on our analyses, the United States WCPO purse seine fishery may adversely affect 3 (95th percentile: 7) sei whales on an annual basis by altering their behavior. However, we would not expect this to result in changes to the lifetime fitness of those whales. Therefore, we do not expect these captures to have measureable effects on sei whale populations in the Pacific Ocean. As a result, we also do not expect these captures to appreciably reduce the sei whales' likelihood of surviving and recovering in the wild.

In summary, the impacts expected to occur and affect sei whales in the *Action Area* are not anticipated to result in appreciable reductions in overall reproduction, abundance, or distribution of the sei whale population in the Pacific Ocean. Because we do not anticipate impacts to the sei whale population in the Pacific Ocean, we also do not anticipate appreciable reductions in overall reproduction, abundance, or distribution of the sei whale population rangewide. For this reason, the effects of the proposed action are not expected to cause an appreciable reduction in the likelihood of survival and recovery of sei whales in the wild.

6.3 Sperm Whale

As described in the *Status of Listed Resources,* the sperm whale is listed as endangered throughout its range and the population structure is poorly understood and have generally been assessed at the ocean basin level for assessing populations, and the limited genetic sampling thus far supports that with low genetic differentiation among basins. In the North Pacific, the most reliable estimates of abundance are 1,997 for California/Oregon/Washington (Moore and Barlow 2014) and 4,559 for Hawaii (Bradford et al. 2017) resulting in a total minimum estimate of 6,556 individuals. We are aware of no reliable abundance estimates specifically for sperm whales in the South Pacific Ocean, and there is insufficient data to evaluate trends in abundance and growth rates of sperm whale populations at this time. Branch and Butterworth (2001) estimate 5,400 to 10,000 individuals in the Antarctic based on summer sighting surveys. There are no trend assessments for sperm whales in the Pacific. The only study that references a trend was based on sperm whale off the California/Oregon/Washington coasts. The abundance of this population was estimated at 2,281 (no confidence interval provided) individuals in 2014, with data from 1991 to 2014 showing a non-significant increasing trend and a significant increase in individuals traveling in small groups of 1 or 2 (Moore and Barlow 2017).

Historically the most significant threat to sperm whales was whaling. Ship strikes, marine pollution, entanglement and anthropogenic noise are significant threats today but they are not difficult to observe, for example, most ship strikes go unnoticed or unreported.

As described in the *Environmental Baseline*, effects from commercial whaling have had the greatest impacts to the populations in the Pacific. From 1800 to 1987, at least 436,000 sperm whales were commercially harvested in the North Pacific (Best 1976 as cited in Carretta et al. 2018a; C. Allison pers. comm. as cited in Carretta et al. 2019a). The IWC banned commercial whaling in the North Pacific in 1966, and some Soviet whaling probably continued after the ban (as cited in Forney and Brownell 1996).

Whale migration, feeding, and breeding locations may be influenced by changing ocean currents and water temperatures, although the extent of potential change is unknown.

Given the estimated number of interactions (*n* = 6; 95% CI: [4, 13]) with sperm whales in the United States WCPO purse seine fishery from 2008-2018, as described in the *Effects of the Action*, NMFS predicts future interaction levels to be a mean of zero and up to one at the 95th percentile annually, with a maximum 5-year running average of 0.37 whales and up to 8 whales over 10 years. As described in the *Response Analysis*, the most significant threat to sperm whales by the United States WCPO purse seine fishery is entrapment by gear which can lead to elevated stress levels and scraping wounds from breaking through the net. We do not anticipate any mortality from these interactions. We expect these interactions to lead to high stress during the event and immediately following post release which may impact essential life functions such as feeding, breeding, or migration in the short term but are unlikely to affect individuals in the long term.

Based on the evidence available, up to one sperm whale could be caught by purse seine gear in any given year (mean = 0; 95th percentile = 1). The evidence available leads us to conclude that these whales will be freed from the net either by the fisherman releasing the net before pursing or from the whale breaking through the net. The best available minimum abundance estimate for the North Pacific sperm whales is 6,556 whales. Assuming the one interaction per year is from the North Pacific, this interaction rate would equal 0.015% of North Pacific sperm whales. Over 10 years, 8 interactions would equal 0.12% of North Pacific sperm whales. Similarly, as previously noted in the *Status of Listed Resources*, the best available abundance estimate for the Southern hemisphere is 5,400 to 10,000 whales and is from the Antarctic. Assuming the one interaction per year is from the Southern hemisphere, this interaction rate would equal 0.019% of the Southern hemisphere sperm whales for the lower estimate (i.e. 5,400) and would be lower if the greater abundance estimate of 10,000 was used. Over 10 years, 8 interactions would equal at most 0.15% of Southern hemisphere sperm whales. There are no trend estimates available for this species, however Moore and Barlow (2017) found a non-significant increasing trend for sperm whales off the coasts of California/Oregon/Washington. It is not anticipated that the populations would decline substantially as the greatest threat to this species, whaling, has been removed. Therefore, we anticipate similar proportions of the population will be impacted 10 and 20 years into the future and none of the interactions are anticipated to be lethal.

Based on our analyses, the United States WCPO purse seine fishery may alter the behavior of up to one sperm whale on an annual basis. We expect individual whales will experience stress during the interaction and immediately following post release which may temporarily alter essential behaviors such as feeding, breeding, or migration, but this would not be expected to affect any individual's lifetime fitness. Therefore, we do not expect these captures to affect the sperm whale populations in the Pacific Ocean. As a result, we also do not expect these captures to appreciably reduce the sperm whales' likelihood of surviving and recovering in the wild.

In summary, the impacts expected to occur and affect sperm whales in the *Action Area* are not anticipated to result in appreciable reductions in overall reproduction, abundance, or distribution of the sperm whale population in the Pacific Ocean. Because we do not anticipate impacts to the sperm whale population in the Pacific Ocean, we also do not anticipate appreciable reductions in overall reproduction, abundance, or distribution of the sperm whale population rangewide. For this reason, the effects of the proposed action are not expected to cause an appreciable reduction in the likelihood of survival and recovery of sperm whales in the wild.

6.4 Green Sea Turtle

6.4.1 East Pacific

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 (Seminoff et al. 2015). Green sea turtle nesting is widely dispersed in the Eastern Pacific Ocean. 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 as cited in Seminoff et al. 2015) to Peru (S. Kelez, Oceanica, pers. comm. 2012 as cited in Seminoff et al. 2015). The total for the entire Eastern Pacific green sea turtle is estimated at 20,108 nesting females (Seminoff et al. 2015). We do not have estimates of the species total abundance. However, we expect that the total population of the Eastern Pacific green sea turtle is likely greater than 265,600, which is the estimated total population of the Central North Pacific green sea turtle, since the number of nesting females is greater than the number of nesting females in the Central North Pacific (3,846).

Eastern 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 nesting occurs are threatened by hurricanes, tropical storms, storm surges, sand accretion, and rainfall associated with storms. The largest threat to 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 sea turtle nesting beaches in other regions around the Pacific, several of the secondary green sea turtle nesting beaches in Mexico suffer from coastal development. Incidental capture in artisanal and commercial fisheries is a significant threat to the survival of green sea turtles throughout the Eastern Pacific Ocean. Significant bycatch has been reported in artisanal gill net and longline shark and 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 Eastern Pacific green sea 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 Eastern Pacific green sea turtles are still routinely captured in international, federal and state commercial fisheries that target other species.

Effects of climate change include, among other things, sea surface temperature increases, and 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. Impacts from global climate change induced by human activities are likely to become more apparent in future years (IPCC 2014). However, at 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 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.

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

As described in the *Exposure Analysis*, the United States WCPO purse seine fishery interacts with more green sea turtles than any other species. All individuals that were measured (n = 32) would be considered juveniles with two possible subadults. They were the most common species to be observed interacting with FADs not associated with an active set, however higher numbers of green sea turtles were observed captured by the fishery during free-sets compared to FAD-sets. In 11 years, 97 green sea turtles have been observed in this fishery, with a total estimate of 173 (95% CI: [152, 200]). Of those, 79% were considered by the observers to be alive and healthy at the time of release, 15% were released injured and 6% were released dying or dead.

As previously discussed in the *Status of Listed Resources*, no genetic data have been collected from green sea turtles interacting with the United States WCPO purse seine fishery. Since we cannot determine the percentages of turtles from each population with great certainty, for the purpose of the jeopardy analysis, we assigned 100% of the anticipated interactions between green sea turtles and the United States WCPO purse seine fishery as affecting the East Pacific green sea turtle.

Based on the 173 (95th percentile: 200) green sea turtles that were estimated to be caught by the U.S WCPO purse seine fishery from 2008 to 2018 as described in the *Effects Analysis* section, NMFS predicts that the purse seine fishery will interact with a mean of 7 (95th percentile: 11) green sea turtles per year with a maximum 5-year running average of 7.56 turtles and up to 76 turtles over 10 years.

As described in the *Response Analyses* section, we applied lower and upper post-interaction mortality rates to sea turtles captured by the fishery. Using those mortality rates, the corresponding mortality estimates with an interaction of 7 (95th percentile: 11) green sea turtles is 0 to 1 (95th percentile: 1 to 2). Assuming total abundance for East Pacific green sea turtles is 265,600 individuals, and we assume that an average of 7.56 (95th percentile: 11) individuals from this species may interact with the United States WCPO purse seine fishery annually, that would equal 0.0028% (95th percentile: 0.0041%) of the species total population. When we consider 76 interactions over the next 10 years, the rate of interaction with the fishery would be 0.029% of the total population. As the East Pacific green sea turtle is considered to be increasing (Seminoff et al. 2015), these percentages represent worst-case scenarios at 10 and 20 years into the future as we anticipate higher population sizes at those time points.

The annual mortality of 0 to 1 (95th percentile: 1 to 2) individual green sea turtles assuming total abundance equals 265,600 represents 0% to 0.00038% (95th percentile: 0.00038% to 0.0008%) of the total population. Over 10 years, we would anticipate 4 to 13 green sea turtles to die as a result of their interaction with the fishery. These mortalities would represent 0.0015 to 0.0049% of the species.

While our assessment estimated that up to two mortalities would occur annually, up to nine would be released with varying degrees of injuries. As noted in the *Response Analysis* section, the majority of captured green sea turtles were reported as landed which means that they were pursed in the net with the remainder of the catch, likely resulting in forced submergence and exposure to crushing for an unknown amount of time, before being brailed out of the net. We acknowledge that other responses can occur from these interactions, and can range from short term behavioral changes to longer lasting effects. These types of effects are not easy to monitor. We anticipate that most of the turtles that do not die as a result of their injuries may experience differing levels of harassment or other sublethal effects from the exposure to entrapment, entanglement, or by handling, which range from being temporary in nature such as elevated stress levels to more significant injuries that may heal over time but may affect the individual's lifetime fitness. Of the 16 animals that live through their capture or entanglement, we expect that a small portion of the turtles that survive may experience fitness level impacts.

Our analysis examines the effect of the United States WCPO purse seine fishery on East Pacific green sea turtles, under several lines of reasoning. Under each scenario examined, we conclude that the impact of the United States WCPO purse seine fishery on East Pacific green sea turtles is sufficiently small that the action would not affect the viability of the populations that these individuals represent.

As noted above, because unknown injuries may occur during the process of landing and releasing green sea turtles, we expect that the United States WCPO purse seine fishery will reduce the lifetime fitness of a portion of the individuals exposed to the action. However, we do not expect this to affect the viability of the populations that these individuals represent due to the low number of expected incidental interactions with the species. 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, NMFS' authorization of the United States WCPO purse seine fishery as currently managed would not be expected to appreciably reduce, directly or indirectly, the threatened East Pacific green sea turtles' likelihood of surviving and recovering in the wild.

6.4.2 Central South Pacific

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 although they are considered to be declining (Seminoff et al. 2015). Based on available data, we estimate there are nearly 3,000 nesters. However, the largest nesting site, Scilly Atoll, which comprises roughly one third of the entire nesting abundance, was last monitored in the early 1990s (Balazs et al. 1995b) and has reportedly significantly declined in the past 30 years as a result of commercial exploitation (Conservation International Pacific Islands Program 2013). There are currently no total population abundance estimates for this species, however, we estimate that the total population of the Central South Pacific green sea turtle is 184,870 sea turtles, which is 70% less than the estimated total population of the Central North Pacific green sea turtle, since the number of nesting females is 70% less than the number of nesting females for the Central North Pacific green sea turtle (3,846).

Central South Pacific green sea turtles are exposed to a variety of natural threats both at their nesting beaches and in the open ocean. Nesting beach habitat loss is occurring as a result of coastal armoring and coastal erosion. Directed harvest of turtles and their eggs is an ongoing problem in the Central West Pacific in American Samoa, Fiji, Kiribati (e.g., Phoenix Islands), Tuvalu, Tokelau, and the Cook Islands (Balazs 1983; Tuato'o-Bartley et al. 1993; 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 species. 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 due to the effects of increased human population and increased human consumption of fish products.

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

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 affecting sea turtle nesting habitat (Wilkinson and Souter 2008). Lastly, studies examining the spatiotemporal 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). 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*, the United States WCPO purse seine fishery interacts with more green sea turtles than any other sea turtle species. All individuals that were measured ($n = 32$) would be considered juveniles with two possible subadults. They were the most common species to be observed interacting with FADs not associated with an active set, however higher numbers of green sea turtles were observed captured by the fishery during free-sets compared to FAD-sets. In 11 years, 97 green sea turtles have been observed in this fishery, with a total estimate of 173 (95% CI: [152, 200]). Of those, 79% were considered by the observers to be alive and healthy at the time of release, 15% were released injured and 6% were released dying or dead.

As previously discussed in the *Status of Listed Resources*; no genetic data have been collected from green sea turtles interacting with the United States WCPO purse seine fishery. Since we cannot determine the percentages of turtles from each population with great certainty, for the purpose of the jeopardy analysis we assigned 100% of the green sea turtles interacting with the fishery to the Central South Pacific green sea turtle. Based on the 173 (95th percentile: 200) green sea turtles that were estimated to be caught by the U.S WCPO purse seine fishery from 2008 to 2018 as described in the *Effects Analysis* section, NMFS predicts that the purse seine fishery will interact with a mean of 7 (95th percentile: 11) green sea turtles per year with a maximum 5-year running average of 7.56 turtles and up to 76 turtles over 10 years.

As described in the *Response Analyses* section, we applied lower and upper post-interaction mortality rates to sea turtles captured by the fishery. Using those mortality rates, the corresponding mortality estimates with an interaction of 7 (95th percentile: 11) green sea turtles is 0 to 1 (95th percentile: 1 to 2). Assuming there are currently about 184,870 Central South Pacific green sea turtles, and we assume that an average of 7.56 (95th percentile: 11) individuals from this species may interact with the United States WCPO purse seine fishery annually, that would equal 0.0041% (95th percentile: 0.0060%) of the species total population.

While we do not know the rate at which the Central South Pacific green sea turtle is declining, evidence suggests that it is declining and that the portion of the population interacting with the fishery may increase if fishery interactions stay the same but population numbers decline. Piacenza et al. (2017) used demographic parameters estimated for the Hawaiian green sea turtle to simulate population trends for 50 years prior to their protection under the ESA assuming juvenile and adult harvest pressures, which are the same pressures faced by Central South Pacific green sea turtles. They estimated population declines of approximately 50% over 10 years, which equates to an annual population decline of approximately 7.7% per year. While some Central South Pacific nesting aggregations have some protections from harvest, we use this rate of decline as a worst-case scenario to estimate the impacts of the fishery out to 40 years. At this rate of decline, the population size in 2030 would be 85,597 and in 2060, 8,497. When we consider 76 interactions over the next 10 years, the rate of interaction with the fishery would be 0.089% of the total population sizes estimated for current and 2030. Between 2051 and 2060 if the number of expected interactions stay the same, the fishery would interact with 0.41% to 0.89% of the total Central South Pacific green sea turtle.

The annual mortality of 0 to 1 (95th percentile: 1 to 2) individual green sea turtles assuming total abundance equals 184,870 represents 0% to 0.00054% (95th percentile: 0.00054% to 0.00060%) of the total population. Over 10 years, we would anticipate four to 13 green sea turtles to die as a result of their interaction with the fishery. These mortalities would represent 0.0047 to 0.015% of the species over the 2021 to 2030 time period. From 2051 to 2060, under the assumption of a rate of decline of 7.7% per year, the mortalities would represent 0.047 to 0.15% of the species if all green sea turtle captures are attributed to this species.

While our assessment estimated that up to two mortalities would occur annually, up to nine would be released with varying degrees of injuries. As noted in the *Response Analysis* section, the majority of captured green sea turtles were reported as landed which means that they were pursed in the net with the remainder of the catch, likely resulting in forced submergence and exposure to crushing for an unknown amount of time, before being brailed out of the net. We acknowledge that other responses can occur from these interactions, and can range from short term behavioral changes to longer lasting effects. These types of effects are not easy to monitor.

We anticipate that most of the turtles that do not die as a result of their injuries may experience differing levels of harassment or other sublethal effects from the exposure to entrapment, entanglement, or by handling, which range from being temporary in nature such as elevated stress levels to more significant injuries that may heal over time but may affect the individual's lifetime fitness. Of the 16 animals that live through their capture or entanglement each year, we expect that a small portion of the turtles that survive may experience fitness level impacts.

Our analysis examines the effect of the United States WCPO purse seine fishery on Central South Pacific green sea turtles, under several lines of reasoning. Again, this is considered conservative as all interactions are attributed to one green sea turtle species with a high rate of decline. Under each scenario examined, we conclude that the impact of the United States WCPO purse seine fishery on Central South Pacific green sea turtles is sufficiently small that the action would not affect the viability of the populations that these individuals represent based on the low number of incidental interactions with the fishery and the abundance of the species.

As noted above, because unknown injuries may occur during the process of landing and releasing green sea turtles, we expect that the United States WCPO purse seine fishery will reduce the lifetime fitness of a portion of the individuals exposed to the action. However, we do not expect this to affect the viability of the populations that these individuals represent due to the low number of expected incidental interactions with the species.

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, NMFS' authorization of the United States WCPO purse seine fishery as currently managed would not be expected to appreciably reduce, directly or indirectly, the endangered Central South Pacific green sea turtles' likelihood of surviving and recovering in the wild.

6.4.3 Central West Pacific

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, however they are considered to be declining (Seminoff et al. 2015). 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, there are approximately 51 nesting sites and 6,518 nesting females in the Central West Pacific. There are no total population estimates for this species, however, we estimate that the total population of the Central West Pacific green sea turtle is greater than 265,600, the estimated total population of the Central North Pacific green sea turtle, since the number of nesting females is greater than the number of nesting females in the Central North Pacific (3,846). There are a number of unquantified nesting sites, possibly with small numbers, although specifics regarding these sites are unknown. The largest nesting site is in the FSM, and that particular site hosts approximately 22% of the total annual nesting females (Seminoff et al. 2015).

There is only one site for which 15 or more years of recent data are available for annual nester abundance (one of the standards for representing PVAs in Seminoff et al. 2015). This is at Chichijima, Japan, one of the major green sea turtle nesting concentrations in Japan (Horikoshi et al. 1994). The PVA has limitations, and does not fully incorporate other key elements critical to the decision making process such as spatial structure or threats. It assumes all environmental and anthropogenic pressures will remain constant in the forecast period and it relies on nesting data alone. The PVA suggests the probability that this population will fall below the trend reference or absolute abundance reference in 100 years approaches zero. 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. Chaloupka et al. (2008a) reports an estimated annual population growth rate of 6.8% per year for the Chichijima nesting site.

Central West Pacific green sea turtles are exposed to a variety of natural threats both at their nesting beaches and in the open ocean. 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 (Seminoff et al. 2015)

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 sea 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 species. While mitigation and minimization measures have reduced fisheries sea turtle bycatch in the United States in recent years, large numbers of Central West Pacific green sea turtles may still be captured in international and United States commercial fisheries that target other species. These activities are reasonably likely to continue, and may increase over time due to the effects of increased human population and increased human consumption of fish products.

In addition, warming sea surface temperatures may lead to potential fitness consequences in sea turtles resulting from altered seasonality and duration of nesting (Pike et al. 2006). Sea turtles may also expand their range as temperature-dependent distribution limits change (McMahon and Hays 2006; Poloczanska et al. 2009). Further, sea turtles have temperature-dependent sex determination, and many populations produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Newson et al. 2009; 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 marine 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 spatiotemporal 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). 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*, the United States WCPO purse seine fishery interacts with more green sea turtles than any other species. All individuals that were measured (*n* = 32) would be considered juveniles with two possible subadults. They were the most common species to be observed interacting with FADs not associated with an active set, however higher numbers of green sea turtles were observed captured by the fishery during free-sets compared to FAD-sets. In 11 years, 97 green sea turtles have been observed in this fishery, with a total estimate of 173 (95% CI: [152, 200]). Of those, 79% were considered by the observers to be alive and healthy at the time of release, 15% were released injured and 6% were released dying or dead.

As previously discussed in the *Status of Listed Resources*; no genetic data has been collected from green sea turtles interacting with the United States WCPO purse seine fishery. Since we cannot determine the percentages of turtles from each population with great certainty, for the purpose of the jeopardy analysis, we assigned 100% of the green sea turtles interacting with the fishery to be from the Central West Pacific population. Based on the 173 green sea turtles that were estimated to be caught by the U.S WCPO purse seine fishery from 2008 to 2018 as described in the *Effects Analysis* section, NMFS predicts that the purse seine fishery will interact with a mean of 7 (95th percentile: 11) green sea turtles per year with a maximum 5-year running average of 7.56 turtles and up to 76 turtles over 10 years.

As described in the *Response Analyses* section, we applied lower and upper post-interaction mortality rates to sea turtles captured by the fishery. Using those mortality rates, the corresponding mortality estimates with an interaction of 7 (95th percentile: 11) green sea turtles is 0 to 1 (95th percentile: 1 to 2). Assuming there are currently about 265,600 Central West Pacific green sea turtles, and we assume that an average of 7.56 (95th percentile: 11) individuals from this species may interact with the United States WCPO purse seine fishery annually, that would equal 0.0028% (95th percentile: 0.0041%) of the species total population. When we consider 76 interactions over the next 10 years, the rate of interaction with the fishery would be 0.029% of the total population.

While we do not know the rate at which the Central West Pacific green sea turtle is declining, evidence suggests that it is declining and that the portion of the population interacting with the fishery may increase if fishery interactions stay the same but population numbers decline. As noted previously, Piacenza et al. (2017) used demographic parameters estimated for the Hawaiian green sea turtle to simulate population trends for 50 years prior to their protection under the ESA assuming juvenile and adult harvest pressures, which are the same pressures faced by Central South Pacific green sea turtles. They estimated population declines of approximately 50% over 10 years, which equates to an annual population decline of approximately 7.7% per year. While some Central West Pacific nesting aggregations have some protections from harvest, we use this rate of decline as a worst-case scenario to estimate the impacts of the fishery out to 40 years. At this rate of decline, the population size in 2030 would be 122,976 and in 2040, 56,940. When we consider 76 interactions over the next 10 years, the rate of interaction with the fishery would be between 0.029% and 0.062% of the total population sizes estimated for current and 2030. Between 2051 and 2060 if the number of expected interactions stay the same, the fishery would interact with 0.29 to 0.62% of the total Central West Pacific green sea turtle.

The annual mortality of 0 to 1 (95th percentile: 1 to 2) individual green sea turtles assuming total abundance equals 265,600 represents 0% to 0.00038% (95th percentile: 0.00038% to 0.0008%) of the total population Over 10 years, we would anticipate 4 to 13 green sea turtles to die as a result of their interaction with the fishery. These mortalities would represent 0.0015 to 0.0049% of the species over the 2021 to 2030 time period. From 2051 to 2060, under the assumption of a rate of decline of 7.7% per year, the mortalities would represent 0.033 to 0.11% of the species if all green sea turtle captures are attributed to this species.

While our assessment estimated two mortalities would occur per year, up to nine turtles would be released with varying degrees of injuries each year. As noted in the *Response Analysis* section, the majority of captured green sea turtles were reported as landed which means that they were pursed in the net with the remainder of the catch, likely resulting in forced submergence and exposure to crushing for an unknown amount of time, before being brailed out of the net. We acknowledge that other responses can occur from these interactions, and can range from short term behavioral changes to longer lasting effects. These types of effects are not easy to monitor. We anticipate that most of the turtles that do not die as a result of their injuries may experience differing levels of harassment or other sublethal effects from the exposure to entrapment, entanglement, or by handling, which range from being temporary in nature such as elevated stress levels to more significant injuries that may heal over time but may affect the individual's lifetime fitness. Of the 16 animals that live through their capture or entanglement, we expect that a small portion of the turtles that survive may experience fitness level impacts.

Our analysis examines the effect of the United States WCPO purse seine fishery on Central West Pacific green sea turtles, under several lines of reasoning. Under each scenario examined, we conclude that the impact of the United States WCPO purse seine fishery on Central West Pacific green sea turtles is sufficiently small that the action would not affect the viability of the populations that these individuals represent.

As noted above, because unknown injuries may occur during the process of landing and releasing green sea turtles, we expect that the United States WCPO purse seine fishery will reduce the lifetime fitness of a portion of the individuals exposed to the action. However, we do not expect this to affect the viability of the populations that these individuals represent due to the low number of expected incidental interactions with the species.

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, NMFS' authorization of the United States WCPO purse seine fishery as currently managed would not be expected to appreciably reduce, directly or indirectly, the endangered Central West Pacific green sea turtles' likelihood of surviving and recovering in the wild.

6.4.4 Southwest Pacific

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. 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. 2008a; Limpus 2009). 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 female green turtles 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).There are currently no total population estimates for this species, however, we estimate that the total population of the Southwest Pacific green sea turtle is greater than 265,600, the estimated total population of the Central North Pacific green sea turtle, since the number of nesting females is greater than the number of nesting females in the Central North Pacific (3,846). Southwest Pacific green sea turtle is thought to have increasing population trends although the exact trend is not known (Seminoff et al. 2015).

Southwest Pacific green sea turtles are exposed to a variety of natural threats both at their nesting beaches and in the open ocean including a loss of nesting beach habitat from sea level rise. Southwest Pacific green sea turtles are vulnerable to harvest throughout Australia and neighboring countries such as New Caledonia, Fiji, Vanuatu, Papua New Guinea, and Indonesia (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 species. While mitigation and minimization measures have reduced fisheries sea turtle bycatch in the United States in recent years, large numbers of Southwest Pacific green sea turtles may still be captured in international and United States commercial fisheries that target other species. These activities are reasonably likely to continue, and may increase over time due to the effects of increased human population and increased human consumption of fish products.

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

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 affecting sea turtle nesting habitat (Wilkinson and Souter 2008). Lastly, studies examining the spatiotemporal 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). 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*, the United States WCPO purse seine fishery interacts with more green sea turtles than any other species. All individuals that were measured (n = 32) would be considered juveniles with two possible subadults. They were the most common species to be observed interacting with FADs not associated with an active set, however higher numbers of green sea turtles were observed captured by the fishery during free-sets compared to FAD-sets. In 11 years, 97 green sea turtles have been observed in this fishery, with a total estimate of 173 (95% CI: [152, 200]). Of those, 79% were considered by the observers to be alive and healthy at the time of release, 15% were released injured and 6% were released dying or dead.

As previously discussed in the *Status of Listed Resources*; no genetic data has been collected from green sea turtles interacting with the United States WCPO purse seine fishery. Since we cannot determine the percentages of turtles from each population with great certainty, for the purpose of the jeopardy analysis, we assigned 100% of the green sea turtles interacting with the fishery to the Southwest Pacific population. Based on the 173 (95th percentile: 200) green sea turtles that were estimated to be caught by the U.S WCPO purse seine fishery from 2008 to 2018 as described in the *Effects Analysis* section, NMFS predicts that the purse seine fishery will interact with a mean of 7 (95th percentile: 11) green sea turtles per year with a maximum 5-year running average of 7.56 turtles and up to 76 turtles over 10 years.

As described in the *Response Analyses* section, we applied lower and upper post-interaction mortality rates to sea turtles captured by the fishery. Using those mortality rates, the corresponding mortality estimates with an interaction of 7 (95th percentile: 11) green sea turtles is 0 to 1 (95th percentile: 1 to 2). Assuming total abundance for Southwest Pacific green sea turtles is 265,600 individuals, and we assume that an average of 7.56 (95th percentile: 11) individuals from this species may interact with the United States WCPO purse seine fishery annually, that would equal 0.0028% (95th percentile: 0.0041%) of the species total population. When we consider 76 interactions over the next 10 years, the rate of interaction with the fishery would be 0.029% of the total population. As the Southwest Pacific green sea turtle is considered to be increasing (Seminoff et al. 2015), these percentages represent worst-case scenarios at 10 and 20 years into the future as we anticipate higher population sizes at those time points.

The annual mortality of 0 to 1 (95th percentile: 1 to 2) individual green sea turtles assuming total abundance equals 265,600 represents 0% to 0.00038% (95th percentile: 0.00038% to 0.0008%) of the total population. Over 10 years, we would anticipate 4 to 13 green sea turtles to die as a result of their interaction with the fishery. These mortalities would represent 0.0015 to 0.0049% of the species.

While our assessment estimated that up to two mortalities would occur annually, up to nine would be released with varying degrees of injuries. As noted in the *Response Analysis* section, the majority of captured green sea turtles were reported as landed which means that they were pursed in the net with the remainder of the catch, likely resulting in forced submergence and exposure to crushing for an unknown amount of time, before being brailed out of the net. We acknowledge that other responses can occur from these interactions, and can range from short term behavioral changes to longer lasting effects. These types of effects are not easy to monitor. We anticipate that most of the turtles that do not die as a result of their injuries may experience differing levels of harassment or other sublethal effects from the exposure to entrapment, entanglement, or by handling, which range from being temporary in nature such as elevated stress levels to more significant injuries that may heal over time but may affect the individual's lifetime fitness. Of the 13 animals that live through their capture or entanglement, we expect that a small portion of the turtles that survive may experience fitness level impacts.

Our analysis examines the effect of the United States WCPO purse seine fishery on Southwest Pacific green sea turtles, under several lines of reasoning. Under each scenario examined, we conclude that the impact of the United States WCPO purse seine fishery on Southwest Pacific green sea turtles is sufficiently small that the action would not affect the viability of the populations that these individuals represent.

As noted above, because unknown injuries may occur during the process of landing and releasing green sea turtles, we expect that the United States WCPO purse seine fishery will reduce the lifetime fitness of a portion of the individuals exposed to the action. However, we do not expect this to affect the viability of the populations that these individuals represent due to the low number of expected incidental interactions with the species. 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, NMFS' authorization of the United States WCPO purse seine fishery as currently managed would not be expected to appreciably reduce, directly or indirectly, the threatened Southwest Pacific green sea turtles' likelihood of surviving and recovering in the wild.

6.4.5 East Indian-West Pacific

As described in the Status of Listed Resources, the East Indian-West Pacific green sea turtle is listed as threatened the abundance of nesting females in this species is estimated at 77,009 females over 50 nesting sites. The largest nesting site is in northern Australia and supports about 25,000 nesting females. 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 species have relatively high probabilities of falling below 100 nesting females per year (Seminoff et al. 2015).

There are currently no total population estimates for this species, however, we estimate that the total population of the East Indian-West Pacific green sea turtle is greater than 265,600, the estimated total population of the Central North Pacific green sea turtle, since the number of nesting females is greater than the number of nesting females in the Central North Pacific (3,846).

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 species and continues to occur where it has been banned (Islam 2001; Seminoff et al. 2015).

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 species. While mitigation and minimization measures have reduced fisheries sea turtle bycatch in the United States in recent years, large numbers of East Indian-West Pacific green sea turtles may still be captured in international and United States commercial fisheries that target other species. These activities are reasonably likely to continue, and may increase over time due to the effects of increased human population and increased human consumption of fish products.

In addition, warming sea surface temperatures may lead to potential fitness consequences in sea turtles resulting from altered seasonality and duration of nesting (Pike et al. 2006). Sea turtles may also expand their range as temperature-dependent distribution limits change (McMahon and Hays 2006; Poloczanska et al. 2009). Further, sea turtles have temperature-dependent sex determination, and many populations produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Newson et al. 2009; 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 marine 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 spatiotemporal 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). 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*, the United States WCPO purse seine fishery interacts with more green sea turtles than any other species. All individuals that were measured (n = 32) would be considered juveniles with two possible subadults. They were the most common species to be observed interacting with FADs not associated with an active set, however higher numbers of green sea turtles were observed captured by the fishery during free-sets compared to FAD-sets. In 11 years, 97 green sea turtles have been observed in this fishery, with a total estimate of 173 (95% CI: [152, 200]). Of those, 79% were considered by the observers to be alive and healthy at the time of release, 15% were released injured and 6% were released dying or dead.

As previously discussed in the *Status of Listed Resources*; no genetic data has been collected from green sea turtles interacting with the United States WCPO purse seine fishery. Since we cannot determine the percentages of turtles from each population with great certainty, for the purpose of the jeopardy analysis, we assigned 100% of the green sea turtles interacting with the fishery to be from the East Indian-West Pacific population. Based on the 173 green sea turtles that were estimated to be caught by the U.S WCPO purse seine fishery from 2008 to 2018 as described in the *Effects Analysis* section, NMFS predicts that the purse seine fishery will interact with a mean of 7 (95th percentile: 11) green sea turtles per year with a maximum 5-year running average of 7.56 turtles and up to 76 turtles over 10 years.

As described in the *Response Analyses* section, we applied lower and upper post-interaction mortality rates to sea turtles captured by the fishery. Using those mortality rates, the corresponding mortality estimates with an interaction of 7 (95th percentile: 11) green sea turtles is 0 to 1 (95th percentile: 1 to 2). Assuming there are currently about 265,600 East Indian-West Pacific green sea turtles, and we assume that an average of 7.56 (95th percentile: 11) individuals from this species may interact with the United States WCPO purse seine fishery annually, that would equal 0.0028% (95th percentile: 0.0041%) of the species total population. When we consider 76 interactions over the next 10 years, the rate of interaction with the fishery would be 0.029% of the total population.

While we do not know the rate at which the East Indian-West Pacific green sea turtle is declining, evidence suggests that it is declining and that the portion of the population interacting with the fishery may increase if fishery interactions stay the same but population numbers decline. As noted previously, Piacenza et al. (2017) used demographic parameters estimated for the Hawaiian green sea turtle to simulate population trends for 50 years prior to their protection under the ESA assuming juvenile and adult harvest pressures, which are the same pressures faced by Central South Pacific green sea turtles. They estimated population declines of approximately 50% over 10 years, which equates to an annual population decline of approximately 7.7% per year. While some East Indian-West Pacific nesting aggregations have some protections from harvest, we use this rate of decline as a worst-case scenario to estimate the impacts of the fishery out to 40 years. At this rate of decline, the population size in 2030 would be 122,976 and in 2060, 12,207. When we consider 76 interactions over the next 10 years, the rate of interaction with the fishery would be between 0.029% and 0.062% of the total population sizes estimated for current and 2030. Between 2051 and 2060 if the number of expected interactions stay the same, the fishery would interact with 0.29 to 0.62% of the total East Indian-West Pacific green sea turtle.

The annual mortality of 0 to 1 (95th percentile: 1 to 2) individual green sea turtles assuming total abundance equals 265,600 represents 0% to 0.00038% (95th percentile: 0.00038% to 0.0008%) of the total population Over 10 years, we would anticipate 4 to 13 green sea turtles to die as a result of their interaction with the fishery. These mortalities would represent 0.0015 to 0.0049% of the species over the 2021 to 2030 time period. From 2051 to 2060, under the assumption of a rate of decline of 7.7% per year, the mortalities would represent 0.033 to 0.11% of the species if all green sea turtle captures are attributed to this species.

While our assessment estimated two mortalities would occur per year, up to nine turtles would be released with varying degrees of injuries each year. As noted in the *Response Analysis* section, the majority of captured green sea turtles were reported as landed which means that they were pursed in the net with the remainder of the catch, likely resulting in forced submergence and exposure to crushing for an unknown amount of time, before being brailed out of the net. We acknowledge that other responses can occur from these interactions, and can range from short term behavioral changes to longer lasting effects. These types of effects are not easy to monitor. We anticipate that most of the turtles that do not die as a result of their injuries may experience differing levels of harassment or other sublethal effects from the exposure to entrapment, entanglement, or by handling, which range from being temporary in nature such as elevated stress levels to more significant injuries that may heal over time but may affect the individual’s lifetime fitness. Of the 16 animals that live through their capture or entanglement, we expect that a small portion of the turtles that survive may experience fitness level impacts.

Our analysis examines the effect of the United States WCPO purse seine fishery on East Indian-West Pacific green sea turtles, under several lines of reasoning. Under each scenario examined, we conclude that the impact of the United States WCPO purse seine fishery on East Indian-West Pacific green sea turtles is sufficiently small that the action would not affect the viability of the populations that these individuals represent.

As noted above, because unknown injuries may occur during the process of landing and releasing green sea turtles we expect that the United States WCPO purse seine fishery will reduce the lifetime fitness of a portion of the individuals exposed to the action. However, we do not expect this to affect the viability of the populations that these individuals represent due to the low number of expected incidental interactions with the species. 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, NMFS' authorization of the United States WCPO purse seine fishery as currently managed would not be expected to appreciably reduce, directly or indirectly, the threatened East Indian-West Pacific green sea turtles' likelihood of surviving and recovering in the wild.

6.5 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 there are between 5,430 and 6,165 annual nesting females in Oceania (NMFS and FWS 2013a). The largest nesting concentration occurs on remote islands in Australia's Great Barrier Reef World Heritage Area, and Australia's Torres Strait area. Additional hawksbill nesting in Oceania occurs at Papua New Guinea, Solomon Islands, Vanuatu, Fiji, Federated States of Micronesia, Republic of Palau, the Samoan Islands (Independent Samoa and American Samoa) and the Mariana Islands.

There is currently no total population estimate for this species, however, as detailed in the *Status of Listed Resources* we estimate the total population of the hawksbill sea turtles in Oceania at 3,440,725 sea turtles (juveniles greater than one-year-old and adults). This estimate is based on the lower nester abundance level of 5,430 annual nesting females (NMFS and FWS 2013a) per year and is exclusive of the estimated 2,098,152 hatchlings successfully emerge from the nest in a year. Due to the high rate of mortality, we used juveniles (greater than one year old) and adults for our analysis and this is considered a minimum population size. The addition of estimated hatchlings to our calculations would further reduce the impacts to the populations, which was not necessary according to the results.

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. 2009). Further, sea turtles have temperature-dependent sex determination, and many populations produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Newson et al. 2009; Patricio et al. 2017; Jensen et al. 2018).

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 spatiotemporal 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*, 59 hawksbill sea turtles were observed interacting with the United States WCPO purse seine fishery from 2008 to 2018. All individuals that were measured ($n = 24$) would be considered juveniles with one possible subadult. They were the second-most common species to be observed interacting with FADs not associated with an active set, however higher numbers of hawksbill sea turtles were observed captured by the fishery during free-sets compared to FAD-sets. All interactions that were recorded with sufficient information to evaluate post interaction mortality risk were assessed using the criteria of Ryder et al. (2006). Of those, 76% were considered by the observers to be alive and healthy at the time of release, 8% were released injured and 16% were released dying or dead.

Based on the total of 99 (95% CI: [87, 124]) hawksbill sea turtles estimated to have interacted with the U.S WCPO purse seine fishery from 2008 to 2018 as described in the *Effects Analysis* section, NMFS predicts that the purse seine fishery will interact with a mean of 4 (95th percentile: 8) hawksbill sea turtles per year with a maximum 5-year running average of 4.5 and up to 46 turtles over 10 years.

As described in the *Response Analyses* section, we applied lower and upper post-interaction mortality rates to sea turtles captured by the fishery. Using those mortality rates, the corresponding mortality estimates with an interaction of an average of 4.5 (95th percentile: 8) hawksbill sea turtles annually is 1 to 1 (95th percentile: 1 to 2). Assuming total abundance for hawksbill sea turtles is 2,592,331 individuals not including hatchlings, and we assume that 4.5 (upper 95th percentile: 8) individuals from this species may interact with the United States WCPO purse seine fishery annually, that would equal 0.00017% (95th percentile: 0.00031%) of the species total population.

While we do not know the rate at which the Oceania population of hawksbill sea turtles is declining, evidence suggests that it is declining and that the portion of the population interacting with the fishery may increase if fishery interactions stay the same but population numbers decline. To account for this, we follow the same method we used for green sea turtle species without trend data and considered the possibility of the population declining to 50% of current numbers over a 10-year period which equates to an annual population decline of approximately 7.7% per year and we consider this to be a worst-case scenario. At this rate of decline, the population size in 2030 would be 1,200,283 and 119,142 in 2060. When we consider 46 interactions over the next 10 years, the rate of interaction with the fishery would be between 0.0018% and 0.0038% of the total population sizes estimated for current and 2030. Between

2051 and 2060 if the number of expected interactions stay the same, the fishery would interact with 0.018 to 0.039% of the total Oceania population of hawksbill sea turtles.

The annual mortality of 1 (95th percentile: 1 to 2) individual hawksbill sea turtles assuming total abundance equals 2,592,331represents 0.000039% (95th percentile: 0.000039% to 0.000077%) of the total population. Over 10 years, we would anticipate 7 to 11 hawksbill sea turtles to die as a result of their interaction with the fishery. These mortalities would represent 0.00058 to 0.0092% of the Oceania population over the 2021 to 2030 time period. From 2051 to 2060, under the assumption of a rate of decline of 7.7% per year, the mortalities would represent 0.0059 to 0.0092% of the population.

Although our assessment estimates that up to two hawksbill sea turtles could die each year from interactions with the United States WCPO purse seine fishery, up to six would be released with varying degrees of injuries. While we focus much of our analysis on the number of mortalities that are anticipated to occur from these interactions, we acknowledge that other forms of take occur from these interactions, which range from harassment to other sublethal effects. As noted in the *Response Analysis* section, the majority of captured hawksbill sea turtles were reported as landed which means that they were pursed in the net with the remainder of the catch, likely resulting in forced submergence and exposure to crushing for an unknown amount of time, before being brailed out of the net. These types of effects are not easy to determine but we anticipate that most of the turtles that do not die as a result of their injuries may experience differing levels of harassment or other sublethal effects from the exposure to entanglement and the process of retrieving the net and landing the catch which range from being temporary in nature such as elevated stress to more significant injuries that may heal over time but may affect the individual's fitness. In our *Response Analysis* section, we concluded three mortalities could occur per year. However, our sample size of hawksbill sea turtles is very small and small changes in the number and type of injury recorded by observers with subsequent interactions could significantly affect the predicted number of mortalities. From the remaining seven interactions, we expect a small portion of the turtles that survive may experience fitness level impacts.

Our analysis examines the effect of the United States WCPO purse seine fishery on hawksbill sea turtles, under several lines of reasoning. Under each scenario examined, we conclude that the impact of the United States WCPO purse seine fishery on hawksbill sea turtles is sufficiently small that the action would not affect the viability of the populations that these individuals represent.

As noted above, because unknown injuries may occur during the process of landing and releasing hawksbill sea turtle, we expect that the United States WCPO purse seine fishery will reduce the lifetime fitness of a portion of the individuals exposed to the action. However, we do not expect this to affect the viability of the populations that these individuals represent. 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, NMFS' authorization of the United States WCPO purse seine fishery as currently managed would not be expected to appreciably reduce, directly or indirectly, the endangered hawksbill sea turtles' likelihood of surviving and recovering in the wild.

6.6 Leatherback Sea Turtle

As discussed in the *Status of Listed Resources* section of this biological opinion, leatherback sea turtles are globally listed as endangered. While the species is composed of seven populations, the proposed action adversely affects only the West Pacific population. The West Pacific population exhibits a species that in the Pacific Ocean is in exponential decline. The West Pacific population is genetically and spatially distinct from other populations including the East Pacific population (NMFS and FWS 2020).

Leatherback sea turtles in the West Pacific population have declined by as much as 78.3% over the last 27 years (Tapilatu et al. 2013). The leatherback status review (NMFS and USFWS 2020) conservatively estimates adult female abundance at 1,277 individuals. This value is based only on nesting at Jamursba Medi and Wermon beaches in Papua Barat, Indonesia as these are the only beaches with long-term modeling. These two beaches likely represent between 50 and 75% of all nesting for this population (NMFS and USFWS 2020). In a separate analysis of the same data, Martin et al. (2020) estimate the population consisted of about 790 adult female leatherback sea turtles (95% CI: 666-942) in 2017 using the median values for nest counts from their Bayesian state space analysis imputing missing nest count data. We consider the values from Martin et al. (2020) to be the best available estimates for abundance for two reasons. First, Martin et al. (2020) imputed missing data for months during which data were not collected, providing a more accurate estimate of total nesting. Second, their model evaluated variation due to natural causes (i.e., changes in nesting over time due to environmental or demographic factors) and observational error (i.e., imperfect data collection; Martin et al. 2020). Therefore, to be precautionary, we use the lower 95th CI value of 666 adult females for the Jamursba Medi and Wermon index beaches combined for the rest of this analysis. Assuming that 75% of adults are females and that the two index beaches represent 75% of all West Pacific leatherback nesting, we estimate a total of 1,184 adults for this population. Based on Jones et al. (2012) there were about 2,600 (1,800-3,400) mature leatherback sea turtles in the West Pacific Ocean population in 2004. As noted in the Status of Listed Resources, this suggests that the total West Pacific population is comprised of about 100,000 individual leatherback sea turtles, and ranging between 47,000 and 195,000 individuals as of 2017 or the end of the time series of nest data for Jamursba Medi and Wermon beaches. Based on NMFS' PVA model (Martin et al. 2020), adult female leatherback sea turtles in the West Pacific population are declining at about 6% per year, and the population as indicated by the index beaches is at risk of falling to less than half of its current abundance in a few as five years (range 5-26 years, mean 12.7 years; Martin et al. 2020).

West Pacific leatherback sea turtles exhibit two distinct life history strategies represented by winter boreal nesters and summer boreal nesters. The importance of these two strategies is not well understood, nor is it clear whether individuals switch between the migratory and nesting patterns that define these two strategies. Nevertheless, while it is possible that both life history strategies are impacted by the fishery, winter nesters are more likely to interact with the purse seine fishery given the location of the *Action Area*.

As described in the *Environmental Baseline*, effects from global climate change, marine debris, and international and United States fisheries are of concern for the species' persistence. Leatherback sea turtles likely already experience low hatch success from elevated beach temperatures and beach erosion. The harvest of eggs and adult leatherback sea turtles in the range of the West Pacific population is of particular concern, and while work has been underway to support conservation and monitoring activities, these threats remain of paramount concern for the population. Based on the estimates of WCPFC, over 8,000 leatherback sea turtles were captured between 2003-2017 in the *Action Area* (Peatman et al. 2018b; BDEP data tables). If this is correct, then some leatherback sea turtles are likely exposed to more than one fishery interaction each year. Fishery interactions and loss of eggs are among some of the greatest threats facing this species. These threats 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.

Leatherback sea turtles represent 3% of the turtles captured in the United States WCPO purse seine fishery. Estimated total number of leatherback sea turtles interacting with the United States WCPO purse seine fishery from 2008 to 2018 is 10 (95% CI: [8, 20]). All observed leatherbacks (*n* = 8) were released alive, with one considered injured by the observer. Only three of those eight were measured, and based on those turtles, 66.6% (or two out of three) of leatherbacks interacting with the fishery are juveniles.

Our analyses predict that the mean (95th percentile) annual interaction will be one (two) annually, with a maximum 5-year running average of 0.48 or up to six individuals over 10 years. We do not anticipate any mortalities to occur from these interactions as described in detail in the *Exposure* and *Response Analysis* sections of this biological opinion. Assuming the West Pacific population includes a total of 47,000 leatherback sea turtles of all age classes, which represents our low estimate for the population, our predicted annual interaction of a maximum 5-year running average of 0.48 (95th percentile: 2) leatherback sea turtles would mean that the United States WCPO purse seine fishery could interact with 0.0010% (95th percentile: 0.0043%) of the total West Pacific leatherback sea turtle population. The 6% per year decline reported by Martin et al. (2020) applies only to the adult female portion of the population, however, given that this is the best available information on trends for this population, we apply it to the entire adult population as well as the entire population including all age classes in the calculations below. Based on this trend, we anticipate that the total population size in 2020 was 39,257 and would be 3,562 by 2030. When we consider six interactions over the next 10 years, the rate of interaction with the fishery would be 0.013 to 0.015% of the total population. Between 2051 and 2060 if the number of expected interactions stay the same, the fishery would interact with 0.092% to 0.17% of the total western Pacific population.

Of the six leatherback sea turtles we anticipate may interact with the fishery over the next 10 years, we expect two of them to be adults based on the ratio of adults to juveniles (1:3). This is based on the size class ratio of observed animals caught in the fishery to date. Based on the estimate of 1,184 adults in the West Pacific population in 2017, and a 6% per year decline, we estimate the 2020 adult population size to be 989, the 2030 adult population size to be 542, and the 2060 adult population size to be 89.

Based on these population sizes, over 10 years, from 2021 to 2030, the fishery would be expected to interact with between 0.12 to 0.27% of the adult population. Over the 10-year period from 2051 to 2060, if the population continues to decline at 6% per year, and if fishery interactions remain the same, the fishery would be expected to interact with 1.22 to 2.25% of the adult population. Importantly, we do not anticipate that any of the leatherbacks are going to die as a result of the interactions; however, other responses can occur from these interactions, and can range from short term behavioral changes to longer lasting effects. These types of latent effects are not easy to monitor. We anticipate that most of the turtles may experience differing levels of harassment or other sublethal effects from their exposure to capture, which range from being temporary in nature such as elevated stress to more significant injuries that may heal over time but may affect the individual's lifetime fitness. While we expect that all leatherbacks will live through their capture, we expect that most will heal but there could also be a smaller percentage that could experience fitness level impacts.

As noted in the *Approach to the Assessment* we are concerned about the viability of the species, and the populations that comprise the species. In the case of the West Pacific population, we are concerned about the long-term persistence of this population and the role it plays in the persistence of the species globally, and particularly the Pacific Ocean. The West Pacific population is unique, spatially and demographically, and as noted by the IUCN, the global conservation of the species is dependent upon the populations that comprise the species; this is particularly true of the West Pacific population, which is the more robust of the two populations in the Pacific Ocean. Changes in a population's spatial structure can affect evolutionary processes and extinction risk, and yet may not be detected from short-term metrics of abundance (McElhaney et al. 2000). It was for these reasons, NMFS advised using four key parameters to evaluating population viability: abundance, growth rate, spatial structure, and diversity—because these are reasonable predictors of extinction risk to all populations of all species (McElhaney et al. 2000).

Finally, while there have been no observed at-vessel mortalities or serious injuries of leatherback sea turtles in the United States WCPO purse seine fishery to date, between 2013 and 2018, two leatherback sea turtle mortalities were recorded in the international WCPO purse seine fishery (data from the WCPFC (2019b)) and we cannot discount the possibility of a mortality. As noted in the *Response Analysis* section, all captured leatherback sea turtles were reported as landed which means that they were pursed in the net with the remainder of the catch, likely resulting in forced submergence and exposure to crushing for an unknown amount of time, before being brailed out of the net. At a minimum these processes would result in elevated stress levels and potentially turtles may have sustained injuries that were not apparent to the observer. Therefore, we assess the possibility of one adult mortality occurring over a 10-year time span. If a mortality occurs between 2021 and 2030, it would represent 0.10 to 0.18% of the adult population. Over the subsequent 10 years, if a mortality occurred between 2030 and 2040, it would represent 0.18 to 0.34% of the adult population.

Despite our concerns regarding the viability of the West Pacific leatherback sea turtle populations, our analysis suggests that, while on average we expect less than one individual to interact with the fishery, and over 10 years the fishery would interact with up to 6 individuals. However, there have been no observed mortalities to date and we would expect no more than one mortality over 10 years based on interactions reported in the international portion of the fishery. Therefore, the authorization of the United States WCPO purse seine fishery as currently managed is not likely to reduce the viability of the West Pacific leatherback sea turtle.

As noted above, because unknown injuries may occur during the process of landing and releasing leatherback sea turtles, we expect that the United States WCPO purse seine fishery could affect the lifetime fitness of a portion of the individuals exposed to the action. However, we do not expect this to affect the viability of the populations that these individuals represent. 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, NMFS' authorization of the United States WCPO purse seine fishery as currently managed would not be expected to appreciably reduce, directly or indirectly, the endangered leatherback sea turtles' likelihood of surviving and recovering in the wild.

6.7 South Pacific Loggerhead Sea Turtle

As discussed in the *Status of Listed Resources* section, we explained that the South Pacific loggerhead sea turtle is endangered, and that the species' trend appears to be decreasing in abundance with the most recent estimate at 3% per year decline (Chaloupka and Limpus (2001). In the Pacific, extensive mtDNA studies show that the southern loggerhead populations are isolated from the northern Pacific populations, and that juveniles from these distinct genetic populations do not disperse across the equator (Hatase et al. 2002; Dutton 2007; Conant et al. 2009).

All loggerhead sea turtles inhabiting the South Pacific Ocean are derived from beaches in Eastern Australia and a lesser known number of beaches in southern New Caledonia, Vanuatu, and Tokelau (Limpus and Limpus 2003b; Limpus and Fien 2009). Nesting colonies of the South Pacific are genetically distinct from loggerhead sea turtles in the North Pacific and Indian Ocean. In Australia, there are both major and minor loggerhead sea turtle nesting rookeries (Limpus and Fien 2009), but long-term counts of nesting females are available for 7 index nesting sites that are all located in Queensland (Woongarra Coast, Heron Island, Wreck Island, Tyron Island, Lady Musgrave Island, Northwest Island and Wreck Rock beaches) and have been monitored from the 1970s to 2011 (Limpus et al. 2013). Additionally, there are two primary index foraging areas that have been monitored (via annual tag-capture-recapture sampling) in eastern Australia in the southern Great Barrier Reef at Heron and Wistari Reef from 1984-1999 (Limpus and Limpus 2003b) and Moreton Bay (Limpus and Fien 2009). The size of the annual breeding population (females only) has been monitored at numerous rookeries in Australia since 1968 (Limpus and Limpus 2003b), and these data constitute the primary measure of the current status of the species.

As explained in the *Status of Listed Resources* section, we estimated the total abundance of the South Pacific loggerhead sea turtle population, based on 770 annual nesting females, at 751,000 juveniles and adults. This excludes the estimated 365,750 hatchlings that successfully emerge from the nest each year; due to the high rate of mortality, we used only juveniles (greater than 1-year old) and adults for our analysis and this is considered a minimum. The addition of estimated hatchlings to our calculations would further reduce the impacts to the populations, which was not necessary according to the results.

Reviewing threats from the *Status of Listed Resources*, Chaloupka et al. (2008b) demonstrated that nesting abundance of loggerheads in Australia was inversely related to sea surface temperatures, and suggested that a long term warming trend in the South Pacific may be adversely impacting the recovery potential of this population. Additionally, loggerheads in the South Pacific are substantially effected by periodic environmental perturbations such as the El Nino Southern Oscillation (ENSO). However, at this time, the Services cannot predict the effect of climate change, sea level rise may present a more immediate challenge for this species because of the proportion of beaches with shoreline armoring that prevents or interferes with the ability of nesting females to access suitable nesting habitat. 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 South Pacific loggerhead sea turtles within the *Action Area*. Bycatch of South Pacific sea turtles occurs in many fisheries throughout the geographic oceanic range of this species. While the only United States fishery likely to impact South Pacific loggerhead sea turtles is the United States WCPO purse seine fishery, large numbers of South Pacific loggerhead sea turtles may still be captured in international fisheries that target other species. These activities are reasonably likely to continue, and may increase over time due to the effects of increased human population and increased human consumption of fish products.

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

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 spatiotemporal 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 discussed in the *Exposure Analysis* section, the United States WCPO purse seine fishery has interacted with an observed 68 South Pacific loggerhead sea turtles from 2008 to 2018. The estimated total number of interactions is 121 (95% CI: [107,144]). Sixty-five of the observed interactions were associated with a fishery set and three were associated with a FAD not involving a set. Of the 68 observed interactions, 77% were considered by the observers to be uninjured at the time of release, 16% were released injured and 7% were released either dying or dead. NMFS predicts that the United States WCPO purse seine fishery will interact with a mean of 5 (95th percentile: 8) South Pacific loggerhead sea turtles per year with a maximum running average of 5.0 per year and up to 50 over 10 years.

As described in the *Response Analyses* section, we applied lower and upper post-interaction mortality rates to sea turtles captured by the fishery. Using those mortality rates, the corresponding mortality estimates with an average of 5 (95th percentile: 8) South Pacific loggerhead sea turtle interactions annually is 0 to 1 (95th percentile: 1 to 2). Assuming total abundance for South Pacific loggerhead sea turtles is 751,000 individuals not including hatchlings, and we assume that 5 (95th percentile: 8) individuals from this species may interact with the United States WCPO purse seine fishery annually, that would equal 0.00066% (95th percentile: 0.0011%) of the species total population.

Evidence suggests that the South Pacific loggerhead sea turtle is declining, therefore the portion of the population interacting with the fishery may increase if fishery interactions stay the same but population numbers decline. The most recent estimate of the decline is 3% per year. At this rate of decline, the population size in 2030 would be 556,354 and by 2060 it would be 226,194. When we consider 50 interactions over the next 10 years, the rate of interaction with the fishery would be between 0.0067% and 0.0090% of the total population sizes estimated for current and 2030. Between 2051 and 2060 if the number of expected interactions stay the same, the fishery would interact with 0.016 to 0.022% of the total South Pacific loggerhead sea turtle.

The mortality of 0 to 1 (95th percentile: 1 to 2) individual South Pacific loggerhead sea turtles per year, assuming total abundance equals 751,000 represents 0% to 0.00013% (95th percentile: 0.00013% to 0.00027%) of the species. Over 10 years, we would anticipate 4 to 10 South Pacific loggerhead sea turtles to die as a result of their interaction with the fishery. These mortalities would represent 0.00072 to 0.0018% of the species over the 2021 to 2030 time period. From 2051 to 2060, under the assumption of a rate of decline of 3% per year, the mortalities would represent 0.0018 to 0.0044% of the species.

Although our assessment estimates that up to two South Pacific loggerhead sea turtles could die each year from interactions with the United States WCPO purse seine fishery, up to six would be released with varying degrees of injuries. While we focus much of our analysis on the number of mortalities that are anticipated to occur from these interactions, effects may range from harassment to other sublethal effects. As noted in the *Response Analysis* section, the majority of captured loggerhead sea turtles were reported as landed which means that they were pursed in the net with the remainder of the catch, likely resulting in forced submergence and exposure to crushing for an unknown amount of time before being brailed out of the net. We do not have sufficient information to describe these effects but we anticipate that most of the turtles that do not die as a result of their injuries may experience elevated stress to more significant injuries that may heal over time, some of which may affect an individual’s fitness. In our *Response Analysis* section, we concluded up to three mortalities will likely occur per year. However, our sample size of South Pacific loggerhead sea turtles is very small and small changes in the number and type of injury recorded by observers with subsequent interactions could significantly affect the predicted number of mortalities. From the remaining 17 annual interactions, we expect a small portion of the turtles that survive may experience reductions in their fitness.

Our analysis examines the effect of the United States WCPO purse seine fishery on South Pacific loggerhead sea turtles, under several lines of reasoning. Under each scenario examined, we conclude that the impact of the United States WCPO purse seine fishery on South Pacific loggerhead sea turtles is sufficiently small that the action would not affect the viability of the populations that these individuals represent.

As noted above, because unknown injuries may occur during the process of landing and releasing loggerhead sea turtles, we expect that the United States WCPO purse seine fishery will reduce the lifetime fitness of a portion of the individuals exposed to the action. However, we do not expect this to affect the viability of the populations that these individuals represent. 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, NMFS' authorization of the United States WCPO purse seine fishery as currently managed would not be expected to appreciably reduce, directly or indirectly, the endangered South Pacific loggerhead sea turtles’ likelihood of surviving and recovering in the wild.

6.8 Olive Ridley Sea Turtle

As discussed in the *Status of Listed Resources* section, there are two listed populations that occur in the *Action Area*; the endangered breeding colony populations on the Pacific coast of Mexico and the threatened populations from everywhere but the coast of Mexico endangered population.

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

In the Western Pacific, the largest remaining breeding aggregations are in Northern Australia where it's estimated that a few thousand females nest annually (Limpus 2008b). Considering remigration intervals and scattered nesting in Indonesia together with Australia, we estimate this population at 6,000 adult females. Assuming a 50% sex ratio, this would equate to 12,000 adults. When added to a well-studied nesting population in Alas Purwo National Park, East Java, Indonesia, which is currently increasing; suggests a minimum adult population size of 16,400 for Indo-West Pacific olive ridley sea turtles. Trends of the West Pacific population are uncertain although they appear to be stable or increasing on some beaches.

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. In 2015 a workshop convened to analyze the effectiveness of sea turtle mitigation measures in the tuna RFMOs, 16 countries provided data on observed sea turtle interactions and gear configurations. From 1989 -2015 those sixteen countries reported there were 762 olive ridley sea turtle interactions with approximately 5% observer coverage. Of these, 206 were a result of United States fishery interactions. When extrapolated from 5% observer coverage the estimate is 15,240 olive ridley sea turtles caught in the region from 1989-2016. 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.

As discussed in the *Exposure Analysis* section, there were 57 observed interactions with olive ridley sea turtles in the United States WCPO purse seine fishery from 2008-2018, the estimated total number of interactions was 105 (95% CI: [90, 118])). Fifty-four of those were associated with a set and three were associated with a FAD not involving a set. Of the 57 observed interactions, 72% were considered by the observers to be uninjured at the time of release, 28%

were released injured and none were released dying or dead. NMFS predicts that the United States WCPO purse seine fishery will interact with a mean of 4 (95th percentile: 7) olive ridley sea turtles per year with a maximum 5-year running average of 6.29 and up to 64 turtles over 10 years.

As described in the *Response Analyses* section, we applied lower and upper post-interaction mortality rates to sea turtles captured by the fishery. Using those mortality rates, the corresponding mortality estimates with an annual interaction of an average of 6.29 (95th percentile: 7) olive ridley sea turtles is 0 to 1 (95th percentile: 0 to 1). Assuming total abundance of olive ridley sea turtles for both the endangered population and the East Pacific portion of the threatened population is a minimum of one million individuals, this would correspond to 0% to 0.00010% of the total population for both metrics. When we consider 64 interactions over the next 10 years, the rate of interaction with the fishery would be 0.0064% of the total population for both the endangered Mexico and threatened East Pacific populations. As both of these populations are considered to be increasing (NMFS and FWS 2014), these percentages represent worst-case scenarios at 10 and up to 40 years into the future as we anticipate higher population sizes at those time points.

Assuming a minimum abundance of olive ridley sea turtles for the West Pacific portion of the threatened population is 16,400 individuals (equivalent to the estimated number of adults), and we assume that an average of 6.29 (95th percentile: 7) individuals from this species may interact with the United States WCPO purse seine fishery annually, this would equal 0.038% (95th percentile: 0.043%) of the total population. When we consider 64 interactions over the next 10 years, the rate of interaction with the fishery would be 0.39% of the total population for the West Pacific portion of the threatened population, this percentage is a worst-case scenario as 16,400 is the estimated size of the adult population, and we are assuming all interactions are with adults. However, based on the size distribution of olive ridleys interacting with the fishery, only about 26% of olive ridleys are adults. As this population is likely stable or increasing (NMFS and FWS 2014; Kurniawan and Gitayana 2020), these percentages will be representative at 10 and up to 40 years into the future as we anticipate equivalent or higher population sizes at those time points.

The annual mortality of 0 to 1 (95th percentile: 0 to 1) individual olive ridley sea turtles represents 0% to 0.0001% of the total population for both the endangered Mexico and threatened East Pacific populations, or 0% to 0.0061% of the West Pacific portion of the threatened population for both metrics. Over 10 years, we would anticipate 1 to 9 olive ridley sea turtles to die as a result of their interaction with the fishery. These mortalities would represent 0.00010% to 0.00090% of the endangered Mexico and threatened East Pacific populations or 0.0061% to 0.055% of the West Pacific portion of the threatened population. As these populations are considered to be stable or increasing (NMFS and FWS 2014), these percentages would represent scenarios at 10 and up to 40 years into the future as we anticipate equal or higher population sizes at those time points.

Although our assessment estimates that up to one olive ridley sea turtle could die from interactions with the United States WCPO purse seine fishery annually, up to six would be released with varying degrees of injuries. While we focus much of our analysis on the number of mortalities that are anticipated to occur from these interactions, we acknowledge that other forms of take occur from these interactions, which range from harassment to other sublethal effects. As noted in the *Response Analysis* section the majority of captured olive ridley sea turtles were reported as landed which means that they were pursed in the net with the remainder of the catch, likely resulting in forced submergence and exposure to crushing for an unknown amount of time, before being brailed out of the net. We do not have sufficient information to describe these types of effects but we anticipate that most of the turtles that do not die as a result of their injuries may experience differing levels of harassment or other sublethal effects from the exposure to hooking and entanglement which range from being temporary in nature such as elevated stress to more significant injuries that may heal over time but may affect the individual's fitness. In our *Response Analysis* section, we concluded that up to two mortalities could occur per year. However, our sample size of olive ridley sea turtles is very small and small changes in the number and type of injury recorded by observers with subsequent interactions could significantly affect the predicted number of mortalities. From the remaining 10 interactions, we expect a small portion of the turtles that survive may experience fitness level impacts.

Our analysis examines the effect of the United States WCPO purse seine fishery on olive ridley sea turtles, under several lines of reasoning. Under each scenario examined, we conclude that the impact of the United States WCPO purse seine fishery on olive ridley sea turtles is sufficiently small that the action would not affect the viability of the populations that these individuals represent.

As noted above, because unknown injuries may occur during the process of landing and releasing olive ridley sea turtles, we expect that the United States WCPO purse seine fishery may reduce the lifetime fitness of a portion of the individuals exposed to the action. However, we do not expect this action to affect the viability of the populations that these individuals represent considering the number of interactions expected to occur in this fishery, the corresponding mortality, the species abundance, and the species ability to persist. 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, NMFS' authorization of the United States WCPO purse seine fishery as currently managed would not be expected to appreciably reduce, directly or indirectly, either the endangered or threatened species of olive ridley sea turtles' likelihood of surviving and recovering in the wild.

6.9 Oceanic Whitetip Shark

Oceanic whitetip sharks are listed as threatened throughout their range. They are exposed to fishing activities throughout the *Action Area*. As discussed in the *Status of Listed Resources,* genetic and tagging studies suggest that oceanic whitetip sharks may be philopatric exhibiting regional site fidelity, which may lead to localized depletions of the species (Howey-Jordon et al. 2013; Tolotti et al. 2015; Camargo et al. 2016; Ruck 2016; Young and Carlson 2020). Only two stock assessments have been completed to date which both only pertain to the WCPO. Rice and Harley (2012) estimated the 2010 total biomass at 7,295 metric tons. FAO (2012) suggest this represents approximately 200,000 individuals assuming and average individual body weight of 80.4 pounds (36.5 kilograms). Tremblay-Boyer et al. (2019) updated the stock assessment of Rice and Harley (2012) and concluded that total biomass in 2010 was 19,740 metric tons and that biomass declined to 9,641 metric tons by 2016. Using the same assumptions as FAO (2012), this biomass would equate to approximately 264,318 individuals. Stock assessments have not been conducted for either the Eastern Pacific or for the global population, therefore this number would be considered a minimum.

Overall, the species has experienced significant historical and ongoing abundance declines in all three ocean basins due to overutilization from fishing pressure and inadequate regulatory mechanisms to protect the species (based on CPUE; Hazin et al. 2007; Lawson 2011; Clarke et al. 2012; Hasarangi et al. 2012; Brodziak et al. 2013; Hall and Roman 2013). The significant declining trends observed in all available abundance indices (e.g. standardized CPUE, biomass, and median size) of oceanic whitetip sharks occurred as a result of increased fishing effort in the longline fisheries, with lesser impacts from targeted longline fishing and purse-seining (Young et al. 2017).

The most significant threat to the species results from the combined effect of fisheries bycatch and exploitation for the fin trade. Bycatch-related mortality in longline fisheries is considered the primary driver for these declines (Clarke et al. 2011; Rice and Harley 2012; Young et al. 2017), with purse seine fisheries being secondary sources of mortality. In addition to bycatch-related mortality, the oceanic whitetip shark is a preferred species for retention because its large fins obtain a high price in the Asian fin market and comprises approximately 2% of the global fin trade (Clarke et al. 2006a). This high value and demand for oceanic whitetip fins incentivizes the retention and subsequent 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. As a result, oceanic whitetip biomass has declined by 88% since 1995 (Tremblay-Boyer et al. 2019). Currently, the population is overfished, and overfishing is still occurring throughout much of the species' range (Rice and Harley 2012; Tremblay-Boyer et al. 2019; 85 CFR 46588). Catch per unit effort 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 (see the casual loop diagram in the *Status of Listed Resources*).

As described in the *Environmental Baseline*, effects from international and United States fisheries have resulted in interactions with the oceanic whitetip shark in the *Action Area*. 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.

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 can 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.

The data show that interactions appear to occur throughout the operational range of the fishery, but with higher densities of interactions within specific areas, such as Nauru or Tokelau's EEZ. Considering the total number of sets throughout this time frame within the *Action Area* was 45,905 sets, these hotspots are considered significant as these areas are geographically distinct when compared to fishing effort and represent only 1.48% of the sets (680/45905*100).

The data show a reduction in interactions between July and September with minor peaks in April and November. Overall, the number of sets that interact with the species has increased steadily since 2008, with a minor decline in the number of sharks caught after peaking in 2010 as depicted in the *Exposure* analysis. Comparing these data to aggregated fishing effort shows that sharks are widely distributed throughout the *Action Area*. Based on the available evidence, we conclude the United States WCPO purse seine fishery are likely to interact with a mean of 88 (95th percentile: 160) oceanic whitetip sharks per year.

Of these 88 (95th percentile: 160) interactions, we expect an overall mortality rate (including at vessel and post release mortality) of 84.2% using silky sharks from this fishery a surrogate species (Hutchinson et al. 2015). From these interactions we expect that 75 (95th percentile: 135) oceanic whitetip sharks are likely to die each year. Given the total abundance of the western Pacific oceanic whitetip shark population contains an estimated 264,318 individuals (our analysis based on the evidence presented by FAO (2012), Rice and Harley (2012), and Tremblay-Boyer et al. (2019)) and assuming that 88 (95th percentile: 160) individual oceanic whitetip sharks may interact with the United States WCPO purse seine fishery this represents 0.033% (95th percentile: 0.061%) of the total western Pacific population. For the 75 (95th percentile: 135) oceanic whitetip sharks expected to die each year, and assuming a minimum population estimate of 264,318 individuals gives us an estimate of 0.028% (95th percentile: 0.051%) of the population would be killed by the United States WCPO purse seine fishery in a given year. The 5-year running average of 102.7 would result in interactions with 0.04% of the population with an associated mortality of 0.03%. Finally, the 5-year maximum running sum of 514 would represent 0.21% of the western Pacific population with an associated mortality of 0.17%.

Oceanic whitetip sharks that do not die due to their interactions may experience injuries or other sublethal effects from their exposure to capture. These effects range from being temporary in nature (e.g., elevated stress) to more significant injuries that may heal over time but may reduce an individual's fitness. Of the remaining 13 (95th percentile= 25) interactions that may occur, a portion of those interactions are likely to result in injuries and while we expect that most sharks will heal quickly although a smaller percentage may experience reductions in their lifetime fitness.

Finally, we examined the effect of maintaining the same number of interactions with the species as its numbers declined. As discussed in the *Status of Listed Resources*, Rice et al.[2] (2020) estimates a mean of 13.3% and a median of 14.6% decline in the western Pacific population over the next 10 years. Using the median value of 14.6% over 10 year equates to a decline of 1.6% per year and we use this a worst case scenario. Rice et al. (2020) further estimate that cumulatively, the United States longline fisheries are responsible for upwards of 9% of this decline to the species' Spawning Potential Ratio (SPR) (Rice et al. 2020). Rice et al. (2020) used a post release mortality value of 25% from Hutchinson and Bigelow (2019), which appears to be biased for alive and healthy sharks (condition code AG) and may not be representative of all interactions in this fishery (see limitations presented by Hutchinson and Bigelow 2019). Therefore, if sharks released injured were included in that study, overall mortality would likely be greater, and this value may be considered a best-case scenario if they arrive in good condition at the vessel and trailing gear (again longline focused) is minimized when released. However, at this time we cannot recreate the analysis using a different overall mortality value specific to this (purse seine) fishery.

Given this rate of decline (1.6% per year; Rice et al. 2020), we expect the population would be 228,871 sharks in the year 2028. If we consider 1,028 interactions over the next 10 years the

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[2] We also recognize, and highlight for the reader, that both PIFSC and the Council are coauthors of this research.

WCPO purse seine fishery would interact with 0.45% of the population with an associated total mortality of 0.38% of the total WCPO oceanic whitetip shark population over the course of those 10 years given this population abundance. Likewise, in 20 years we would expect the population to be 195,031 if this rate was consistent. Our analysis shows that the level of interaction over this time with an additional 1,028 interactions (over that next 10-year interval) would result in 0.062% of the population being captured with a mortality of 0.52% of the population. In 30 years, we estimate the population will be 166,193 if this rate of decline continued. An additional 1,028 interactions in that 10-year time frame would result in interactions with 0.62% of the western Pacific population with an estimated mortality of 0.52%. Lastly, in 40 years we estimate the population would be 141,622. Maintaining the same level of interaction would result in interactions with 0.73% of the western Pacific population with estimated mortality of 0.61% of the population using the current estimated mortality. We did not compound the number of interactions through these 40-year projections as that would double count individuals if they were removed from the population, as the population declines.

Finally, our analysis examines the effect of the action against a portion of the Pacific population, represented by the western Pacific. If abundance values were available for the eastern Pacific we would combine these numbers with the western Pacific to better describe the current status of the Pacific population, which would undoubtedly combine to produce a larger estimated population size. We would then compare that number to our estimates of the number of interactions and mortalities, which would result in much lower estimated effect to the Pacific population of oceanic whitetip sharks

The oceanic whitetip shark is globally listed. In our analysis, we assessed the impact of interactions with the United States WCPO purse seine fishery on the estimated size of the population in the WCPO (264,318 individuals) as it represents the best available data for this consultation. There are no global abundance estimates for this species but it is reasonable to assume that it is much larger than 264,318 individuals.

When we take into account the number of expected interactions with the purse seine fishery, the number of sharks estimated to be present within the WCPO (264,318 individuals) from the most recent stock assessment, and the uncertainty regarding population abundance estimates in the Eastern Pacific and globally; we conclude that the number of sharks this fishery interacts with will not have an appreciable effect on the population of oceanic whitetip sharks in the *Action Area.* That is, if the rate of decline in the species abundance is consistent across the entire western Pacific and additional mitigation measures are unsuccessful in other fisheries. Considering the data we do have and the evidence presented herein, NMFS does not expect this action to affect the viability of the populations that these individuals represent. 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, NMFS' authorization of the United States WCPO purse seine fishery as currently managed would not be expected to appreciably reduce, directly or indirectly, the oceanic whitetip shark’s likelihood of surviving and recovering in the wild.

6.10 Indo-West Pacific Scalloped Hammerhead Shark

There are three identified scalloped hammerhead species in the Pacific, two of which are listed under the ESA. The Indo-West Pacific is listed as threatened. They are exposed to fishing activities throughout the *Action Area*. As discussed in the *Status of Listed Resources,* the population is estimated to be 33,600. 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).

As described in the *Environmental Baseline*, effects from international and United States fisheries have resulted in interactions with the Indo-West Pacific scalloped hammerhead sharks in the *Action Area*. It is estimated that the ASLL fishery will interact with up to 33 Indo-West Pacific scalloped hammerhead sharks every three years, with up to 15 of them being mortalities. Between 2013 and 2017, the international WCPO longline fishery is estimated to have captured 2,997 Indo-West Pacific scalloped hammerhead sharks, with 1,096 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 species. 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, reviewing all United States WCPO purse seine data from 2008 to 2018 there have been 14 observed interactions and 41 estimated total interactions with the species (95% CI: [31-51]). All 14 interactions resulted in mortality. Based on the 14 Indo-West Pacific scalloped hammerhead sharks that were caught by the United States WCPO purse seine fishery, NMFS predicts that the fishery will interact with up to two sharks per year. From these we expect all individuals will die.

Assuming total abundance for Indo-West Pacific scalloped hammerhead sharks is 33,600 individuals, and we assume that 2 (95th percentile: 2) individuals from this species could be captured, then the United States purse seine fishery may interact with 0.006% (95th percentile: 0.006%) of the total population. Therefore, the mortality of 2 (95th percentile: 2) Indo-West Pacific scalloped hammerhead sharks represents 0.006% (95th percentile: 0.006%) of the total population. The 5 year running average produces the same values (0.006%) and the 5 year running sum of 10 interactions would result in interaction and subsequent mortality of 0.03% of the species' population.

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. However, Figure 5 of Pacoureau et al. (2020) suggests populations in the South Pacific and Indian Oceans (i.e. Indo West Pacific scalloped hammerheads) have stabilized at a depressed rate. Therefore, we examined the effect of two scenarios: 1) if the population remains stable at this abundance moving forward; and 2) maintaining the same number of interactions with the species as its numbers declined at this (-2.31%) rate.

If the total population of Indo-West Pacific scalloped hammerhead sharks maintained its current estimated abundance of 33,600; and the fishery interactions remain the same over 10 years, the fishery would interact with and result in the mortality of 0.06% of the total population over that time frame. For each additional 10-year interval, the number of interactions would account for the same percentage (0.06%) as the number of interactions would be expected to be the same. When we considered the second scenario with a decreasing population growth trend, the population in 10 years would be expected to be 26,668 individuals. The number of interactions and mortality expected to occur in the next 10 years (20 individuals), results in the WCPO purse seine fishery interacting with and responsible for the mortality of 0.07% of the population. Projecting out to 40 years, the expected population would be 13,335. Twenty interactions over that next 10-year interval would result in interaction and mortality of 0.15% of the total Indo-West Pacific scalloped hammerhead shark population. Our projections for 20 and 30 years were with within this range. Therefore the 40-year projection would be the worst case scenario as the population would be the lowest abundance considered. As with the other species considered herein, we did not compound the number of interactions over the time span which would double count individuals as they were removed from the population.

Our analysis examines the effect of the United States WCPO purse seine fishery on Indo-West Pacific scalloped hammerhead sharks, under several lines of reasoning. Under each scenario examined, we conclude that the impact of the purse seine fishery on Indo-West Pacific scalloped hammerhead shark is sufficiently small that the action would not affect the viability of the populations that these individuals represent.

Based on the available evidence, we expect that the United States WCPO purse seine fishery will reduce the lifetime fitness of a portion of the individuals exposed to the action as all individuals are expected to die from their interactions with the fishery. However, we do not expect this to affect the viability of the populations that these individuals represent considering the number of interactions expected to occur in this fishery, the corresponding mortality, the species abundance, and the species ability to persist. 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, NMFS' authorization of the United States WCPO purse seine fishery as currently managed would not be expected to appreciably reduce, directly or indirectly, the endangered Indo-West Pacific scalloped hammerhead sharks’ likelihood of surviving and recovering in the wild.

6.11 Giant Manta Ray

As described in the *Status of Listed Resources*, the giant manta 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 6); however, estimates of the global population size as well as population sizes within the *Action Area* remain unavailable. The species appears to have a population substructure that is composed of independent demographic units or subpopulations with small distinct home ranges. There are a few available estimates of the size of subpopulations, but many available counts are opportunistic and likely do not represent robust population estimates. In general, while the number of individuals identified in subpopulations are typically small (less than 500 individuals; Table 6).

In recent years, emphasis has been placed by some researchers on determining population structure, which has led to more rigorous population studies (e.g., through mark recapture techniques to estimate pop size as opposed to opportunistic counts). Ongoing research including mark-recapture analyses suggests that typical subpopulation abundances are more likely in the low thousands (e.g. Beale et al. 2019) and in rare cases may exceed 10,000 in areas with extremely high productivity (pers. comm. Joshua Stewart, Manta Trust to A. Garrett 2021). 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). Notwithstanding that available counts may not be representative of a true population or subpopulation; fisheries (direct harvest or bycatch) have likely affected some subpopulations.

While the giant manta ray may be physically capable of longer migrations, and it is globally listed, the viability of the smaller independent demographic units (subpopulations) appears to be important to the species survival and recovery. 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). For example, in Indonesia, gill net fisheries in the Lembeh Strait captured 1,424 manta rays (*Mobula* spp.) in a 10-month period, resulting in an apparent local extirpation of giant manta rays (D. Djalal and A. Doali, personal communication cited in Beale et al. 2019).

CITES (2013) compiled information from Fernando and Stevens (2011), Heinrichs et al. (2011), O’Malley et al. (2013), and Lewis et al. (2015; presented as Setiasih et al. in prep in CITES 2013) to summarize our current knowledge of giant manta ray subpopulation locations. The most comprehensive of these is O’Malley et al. (2013), which presents an overview of the economic value of manta ray watching tourism. They highlight 23 sites globally, and nine of these sites are within the *Action Area* of the United States WCPO purse seine fishery: Indonesia, Papua New Guinea, Federated States of Micronesia, Palau, Solomon Islands, Kiribati, New Caledonia, Fiji and French Polynesia.

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).

According to the status review, giant manta rays’ longevity is estimated to be greater than 20-40 years and their age of maturity ranges from three to >15 years (Miller and Klimovich 2017). Their reproductive periodicity is anywhere from an annual cycle to a 5-year cycle, with an average litter size of only 1 pup, and their generation interval is estimated to be around 25 years (Notarbartolo-Di-Sciara 1988; Marshall and Bennett 2010; Dulvy et al. 2014; Rambahiniarison et al. 2018). Given these life history traits, combined with small subpopulation sizes, giant manta rays are especially vulnerable to threats that reduce its numbers, and the species appears to have little potential to recover from high and sustained levels of mortality.

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*, effects from international and United States fisheries have resulted in interactions with the giant manta ray in the *Action Area*. The ASLL fishery is estimated to capture up to 60 giant manta rays every three years. Only 2 interactions were observed between 2010 and 2017 with 15 total estimated interactions over this time frame (McCracken 2019a). 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. While observer rates are greater in United States longline fisheries, when reviewing the greater WCPO fishery as a whole, the percent coverage is much less as noted by Tremblay-Boyer and Brouwer (2016). However, we do not know what portion of these may be misidentified. Since animals are seen by observers in the water, and the line is likely cut at the rail due to an animals' size and therefore the animals are typically not boarded, we expect that a significant portion of the animals could be misidentified.

Considering Tremblay-Boyer and Brouwer's (2016) purse seine data, we estimate that approximately 2,474 giant manta rays may be captured in the greater WCPO purse seine fisheries each year. As previously noted, this would include the United States portion of the fishery but, based on how the data are combined by the WCPFC, it is not possible to extract the United States data from the total dataset. Based on these estimates, it appears that the United States WCPO purse seine fishery accounts for more than 9% of the total giant manta rays captured each year based on the number of documented and estimated interactions by the United States fleet (see Table 65). Again, while the data would suggest the United States fleet catches upwards of 12% of the giant manta rays once unidentified rays are prorated for the species, this may not be a reliable estimate without also accounting for unidentified rays in the greater purse seine fleet. As FFA observers are placed on vessels throughout the greater purse seine fishery including United States vessels, misidentification issues are expected to occur throughout the WCPO fishery. If the greater WCPO fishery has the same 75% misidentification rate throughout, then we would estimate a total of 619 giant mantas are caught each year by the United States and international WCPO purse seine fisheries. These activities are reasonably likely to continue and may increase over time because of increased human population, increased human consumption of fish products, and the international trade of mobulid gill plates.

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 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 giant manta rays (Miller and Klimovich 2017). Coral reef degradation from anthropogenic causes, particularly climate change, is projected to increase through the future. As declines in coral cover have been shown to result in changes in coral reef fish communities (Jones et al. 2004; Graham et al. 2008), the projected increase in coral habitat degradation may potentially lead to a decrease in the abundance of manta ray cleaning fish (e.g., *Labroides* spp., *Thalassoma* spp., and *Chaetodon* spp.) and an overall reduction in the number of cleaning stations available to manta rays within these habitats. Decreased access to cleaning stations may negatively affect the fitness of the 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.

As described in the *Effects of the Action* section of this opinion, the most significant threat to giant manta rays by the United States WCPO purse seine fishery is the level of mortality that the species incurs due to its interactions with the fishery (96% at vessel mortality). As noted in the *Exposure Analysis* section, CMM 2019-05 will prohibit the use of destructive and injurious measures for releasing mobulids, and encourage the use of slings and the immediate (as practicable) release of landed mobulids. We acknowledge that these measures may improve post-release survival of giant manta ray, however until there are data available to ascertain the effectiveness of these measures at reducing mortality, we will continue to assume high mortality rates for this species.

If observers are correctly identifying giant manta ray, and 36.85% of the unidentified *Mobula* nei category were giant manta ray, we estimate that the United States WCPO purse seine fishery interacted with 3,676 giant manta rays between 2008-2018 (95% CI: [3,119, 4,467]). Observed interactions were used to estimate future annual interaction rates using a Bayesian state-space model using uninformative gamma priors. The posterior distributions of interaction rate consisted of 20,000 samples from three chains with a burn-in of 50,000 iterations. From this, we estimated the mean number of giant manta rays captured by the United States WCPO purse seine fishery each year, if all giant manta rays were correctly identified and 36.85% of unidentified mobulids were giant manta ray, as 165 (95th percentile: 199; note the 95th percentile is from a nonparametric bootstrap analysis) based on 3,100 sets per year with 1,581 of them being FAD sets with a maximum 5-year running average of 187.4 and up to 1,847 captures over 10 years.

Earlier in this opinion, we expressed concern that the FFA observer program and the United States WCPO purse seine fishery likely have a problem with their data on giant manta rays. FFA provided NMFS with 15 photographs, and based on the date and time stamp IFD believes the 15 photos represent 6 or 7 individual rays. Although we do not know exactly what FFA observers may have cataloged the identification of these species as, the collective consensus of several experts (IFD, PRD, Manta Trust, and others; pers. comm. E. Crigler to A. Garrett, Oct 2020) is that they are not *M. birostris* and are instead, *M. tarapacana* (the Chilean devil ray). Although pictures of seven individuals is not much given the large number of manta rays intercepted by this fishery, we are aware that the IATTC had a manta species identification problem where large numbers of giant manta rays were reportedly captured. We are also aware that there is a substantial reef manta population near Kiribati, where this fishery is thought to interact with a large number of *M. birostris* (G. Stevens, Manta Trust, pers. comm. to A. Garrett, Feb. 2020). Consequently, we have reason to believe that this may be a substantial issue for the United States WCPO purse seine fishery too. After the IATTC improved identification techniques and protocols for rays, reef mantas were found to be caught with higher frequency than giant mantas (J. Stewart pers. comm. to J. Rudolph, 11 May 2020).

As described in the *Effects of the Action* section, in the United States WCPO purse seine fishery, 76% (95% confidence interval= 66 to 86%) of individuals identified to be giant manta ray have weights estimated to be less than the birth weight of giant manta ray, suggesting these individuals are more likely to be smaller ray species. In a separate analysis of the United States WCPO purse seine fishery observer data, Nowlis (2021) suggests that, based on reported weights and the weight distribution of all mobulid species that may interact with the fishery, no more than 25% are likely to be giant manta rays. However, there are caveats to these analyses of observer-estimated weights, as presented in *Effects of the Action* section, specifically that these weights were estimated by observers based on sighting only and not actual measurements. Despite our concerns that the weight data may lack precision, absent other information like a sizeable photo catalogue or tissue samples from a large number of *Mobula* captured by this fishery that we could use to develop estimates of the proportion of giant manta ray to other *Mobula* species, our analysis of observed weights and the work done by Nowlis (2021) represent the best information available to infer the magnitude of any misidentification problem in this fishery. We also know that even after improving identification techniques in the IATTC for rays, observers in this RFMO still misidentify about 25% of giant manta rays (Stewart pers. comm. to J. Rudolph, 11 May 2020).

With the understanding that the fishery likely has a significant number of rays misidentified as giant manta ray, we revised our Bayesian state-space model with the assumption that 75% of giant manta rays were misidentified (i.e. that only 25% of those individuals identified as giant manta ray in Table 52 were identified correctly) and that 75% fewer unidentified mobulids should be assigned to the giant manta ray category. This assumption relies on the distribution of observed and estimated weights, which suggest that a number of rays are smaller than the expected size of giant manta ray at birth (approximately 75% of captured individuals are likely too small to be a giant manta ray). We adjusted the observer data to account for this assumed error in identification, and reran our Bayesian state-space model. This provided us with a new more plausible estimate that the mean number of giant manta rays likely to be captured by the United States WCPO purse seine fishery each year is 45 (95th percentile: 50) based on 3,100 sets per year with 1,581 of those sets being FAD sets with a maximum 5-year running average of 47.0 and up to 472 captures over 10 years. In the *Effects of the Action* section, we explain in more detail why we consider this level of interaction (45 [95th percentile: 50]) is a more reliable estimate than a mean estimate of 165 interactions (95th percentile: 199).

Although we do not know at vessel mortality for giant manta rays, our best professional judgement is that at vessel mortality for giant manta rays is 96% because of the current handling methods implemented in the fishery. Currently, bridles are passed through an animal's gills or body to lift them and slings are not used, additionally animals are left on deck until catch sorting is completed. We acknowledge that the recent measures (CMM 2019-05) may improve post-release survival of giant manta ray, however until there are data available to ascertain the effectiveness of these measures at reducing mortality, we will continue to assume high mortality rates for this species. Using this estimate, then on average 158 (95th percentile: 191; not adjusted for misidentification) of the observed giant manta rays captured are likely to die in the United States WCPO purse seine fishery annually, or 43 (95th percentile: 48) assuming 75% are misidentified. Up to 8 giant manta rays are expected to be released with varying degrees of injuries if identifications were considered correct. However, only two giant manta rays are expected to be released with varying degrees of injury a year when considering 75% misidentification. If individual giant manta rays do not immediately die from their interaction with the fishery, they may suffer impaired swimming or foraging abilities, altered migratory behavior, altered breeding or reproductive patterns, and latent mortality from their interactions. 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 conducted a giant manta ray hotspot analysis (Figure 58) to examine the spatial distribution of the interactions between giant manta rays and the United States WCPO purse seine fishery. Our hotspot analysis reveals that most interactions occur in the EEZs of the Republic of Nauru and Kiribati and north of the Solomon Islands (Figure 58) primarily within the Papua New Guinea EEZ (PNG East), with 52% of all observed captures occurring in the primary hotspot. Our hotspot analysis illustrates where interactions have been occurring and where subpopulations may be present, but the connectivity or discreteness between subpopulations is unknown without results from genetic testing, which is not currently available. Due to the spatial distance from the Nauru/Kiribati hotspot to the PNG East hotspot, we can assume these represent two discrete subpopulations, at a minimum.

Of all United States WCPO purse seine sets interacting with giant manta rays, 3% of the sets captured 10 or more individuals on a single set, representing 30% of all observed giant manta ray captures. Because giant manta rays likely exist in discrete subpopulations with limited movement between subpopulations (see CITES 2013 and Table 6), these sets, if giant manta ray were correctly identified, may have already had substantial negative impacts on the subpopulations. In other areas, giant manta ray subpopulations that have been extirpated by fisheries have not recolonized (Lewis et al. 2015; Moazzam 2018; D. Djalal and A. Doali pers. comm. cited in Beale et al. 2019), suggesting that migration does not recover local reductions in subpopulations.

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, 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 United States WCPO purse seine fishery. This allowed us to identify 19 putative subpopulations (Table 56) using a reasoned approach that incorporated the best information we have about the species distribution, movements, and demographics.

Next, we estimated the number of interactions (and mortalities) anticipated to affect each subpopulation annually. In other words, we distributed our predictions for future interactions across our putative subpopulations. Before taking into consideration any misidentification issue, this analysis determined that the fishery is not likely to impact the French Polynesia and Johnston Atoll subpopulations and we eliminated those subpopulations from further analyses, and focused the remainder of our analysis on the remaining 17 subpopulations noted in Table 52.

We then developed several scenarios to assess the likelihood that fishery captures would cause subpopulation declines. This step is important for several reasons, noted earlier in this opinion, particularly within the *Approach to the Assessment* section, because reductions in the viability of one or more of the populations (or subpopulations) that comprise the species are often a prelude to a species' extinction. The ability of population to remain stable or increasing with fishery captures depends on the initial population size and the maximum intrinsic population growth (r_{max}), which incorporates estimates of natural mortality, age at maturity, longevity and fecundity.

Because our subpopulation structure is estimated, and we do not know the size of these subpopulations, we examined the effect of the action on the viability of each subpopulation. To do this our assessment we examined a range of initial subpopulation abundances based on values gathered from the literature (abundances were 60, 100, 200, 400, 600, 800, 1,000, 1,500, 1,875 or 2,464 individuals). As described above and in the *Exposure Analysis* section, we focused most of our attention on population sizes larger than 1,000 individuals as this is likely the minimum size of viable populations. The smaller population sizes, while in the literature, are likely a result of either incomplete sampling of individuals; however, since they may also represent exploited populations that may be at higher risk of extirpation, to provide the benefit of the doubt to the species, each scenario includes subpopulation sizes of less than 1,000 individuals . In all, we created more than 5,000 reasoned simulations to examine the effect of the action on threatened giant manta rays. The first two scenarios examine the effect of the action on 17 subpopulations with ten initial (pre action) abundances, three values for r_{max}, and looked at multiple time intervals (10, 20 and 40 years) for the level of anticipated number of interactions estimated assuming 100% correct species identification and 36.85% of unidentified mobulids were giant manta ray (i.e. 187 maximum 5-year running average per year for Scenario 1 and 95th percentile of 199 per year for Scenario 2).

Because the fishery may have counted a number of captured rays as giant manta rays when they were actually another *Mobulid* species, we were concerned that our estimates of the effects of the action on giant manta ray subpopulations, as modeled with Scenario 1, likely overestimated the effects of the action on giant manta rays. Therefore, we examined the literature and sought assistance from NMFS IFD to explore whether there was information that could be used to ascertain what proportion of observer records were misidentified, or what proportion of interactions might be with another *Mobulid* species. This is what Kulijs (2014) describes as a case where the science does not seem to provide enough insight to figure out just how wrong the problem of species identification is, but does suggest that there is very likely an issue in this fishery. The evidence for misidentification of giant manta rays is specified in the *Exposure Analysis* section and at the end of this section. Briefly, using weights as estimated by observers, we found that between 0 and 48% of individuals identified as giant manta rays had estimated weights lower than the presumed birth weight of giant manta ray each year from 2010 to 2018, suggesting that the smaller individuals were juveniles or adults of different, smaller *Mobulid* species. Overall, we estimated a mean of 76% (95% confidence interval: 66 to 87%) individuals identified as giant manta rays were smaller than the presumed birth weight of giant manta rays, and were likely misidentified. Given that the weights are estimated by observers, we consider it reasonably likely that the misidentification rate is at least a 50% (approximating the highest annual rate) based on the weight analysis we conducted for this consultation, with 75% (approximating the mean rate) misidentification rates as a reasonable upper estimate. That there is a misidentification problem in this fishery seems highly likely, which means the estimates that we develop with this data are going to be wrong, but by how much, we do not know. Our best estimate relies upon the weight analysis and Nowlis (2021), which suggests that about 75% of individual rays recorded by observers as giant manta rays are likely to be another species of *Mobula*.

Therefore, we performed a third scenario, which is similar to Scenarios 1 and 2, but we reduced the maximum 5-year running average annual interaction number by 50 and 75% (i.e. 94 and 47 per year respectively). Scenarios 3a and 3b provide for an interesting comparison with the results of Scenario 1; even at half the predicted interactions, the Nauru and Kiribati subpopulations still experience serious declines if r_{max}- = 0.019 if population sizes are at least 1,000. If r_{max}- 0.042 and initial population size is at least 1,000 individuals, all subpopulations had increasing population sizes (Appendix B). Under Scenario 3a, the highest number of interactions experienced by any one population is 32, and the smallest number predicted to occur is one.

In our fourth scenario, we considered the role that immigration and emigration might play in maintaining these clusters of subpopulations, and how that may affect their resilience in the face of harvest and other threats. However, we suspect that Scenario 4 is not realistic. Not only does the available literature suggest that there is likely very little movement between subpopulations, the results of Scenario 4 indicate that it is unlikely that smaller subpopulations would be sustained by immigration and unlikely that larger subpopulations can withstand both the loss of individuals through emigration and fishery mortalities. Therefore, we do not consider this scenario further.

We essentially replaced Scenario 3 with Scenarios 5 and 6. Scenarios 5 and 6 were conducted in the same manner as Scenarios 1 and 2, but used the maximum 5-year running average and 95th percentile anticipated interactions under the assumption of 75% misidentification rate for giant manta ray (47 [95th percentile: 50]) as described above. The key difference between Scenario 3b and Scenario 5 is that Scenario 3b was simply a 75% reduction of the annual estimated number of interactions assuming 100% correct observer identification while for Scenario 5 we reduced annual observed interactions by 75% and revised the Bayesian state-space model to calculate new interaction rates for both FAD and free sets, combining them based on the projected 3,100 sets per year with 1,581 of them being FAD sets. Hence, we consider this a more precise prediction than simply reducing the anticipated level of interactions by 75% as we did under Scenario 3.

Our simulation of the effect of the action on the 17 putative subpopulations allowed us to examine a range of reasoned alternatives using the best of what we know about giant manta rays to forecast the effect of the action on the viability of the subpopulations the individual giant manta rays captured by the United States WCPO purse seine fishery likely represent. As noted in the *Effects of the Action* section, we consider Scenario 5 to be representative of the best estimate of the impacts of the fishery on giant manta ray based on our best professional judgment and we focus our analysis and develop our conclusions based on this scenario. However, we will contrast the results of Scenario 5 with Scenario 1 throughout our analysis here to highlight the impact of species identification

Our initial analysis included estimates of the effects of the action considering three different intrinsic rates of population growth from published literature on giant manta rays. However, as noted in the *Exposure Analysis* section, 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; Table 57), NMFS has concluded that the r_{max} values published by Rambahiniarison et al. (2018) likely represent the best r_{max} value available for this consultation (r_{max} = 0.019). We concluded that the r_{max} value published by Ward-Paige et al. (2013; r_{max} = 0.042) is also reasonable, but left us with some concerns over the methodology because the density dependent assumptions used by Ward-Paige et al. (2013) result in somewhat illogical r_{max}

values, whereby the lowest adult mortality rate (0.02; Table 57) and highest fecundity (0.36; Table 57) would combine to give a low r_{max} value of 0.012. Lastly, NMFS is concerned that the r_{max} estimated by Dulvy et al. (2014) is likely an overestimate because the Euler-Lotka equation that they used is not consistent with other derivations in the literature (i.e. it is missing a term for the probability of survival to maturity; Myers et al. 1997; Smith et al. 1998; Cortes 2016) and may be overestimating r_{max}, which may underestimate fishery impacts to giant manta ray subpopulations. Therefore, we focus on the modeling results using r_{max} values of 0.019 (Rambahiniarison et al. 2018) and 0.042 (Ward-Paige et al. 2013).

As noted above, we considered the anticipated number of interactions based on 75% misidentification of giant manta ray and the 5-year running average as our best estimate of the effect of this fishery and focus our analysis on this level of interaction (Scenario 5); however, we considered a range of scenarios, initial population sizes, and r_{max} values and recognize that Scenario 1 represents the anticipated interaction levels based on the records of individuals identified as giant manta rays by observers. Scenarios 2 (no consideration of misidentification) and 6 (75% misidentification) used annual capture numbers equivalent to the 95th percentile, and while this number of annual captures may occur in a given year, it is not anticipated to occur over multiple years, therefore we do not consider these scenarios further.

Through all scenarios, there is a similar pattern — as subpopulation size decreases, the risk that the proposed action would extirpate the subpopulation increases. For Scenario 1 (no consideration of misidentification), if initial population abundance is at least 1,000, no subpopulation would experience extirpation and four (r_{max} = 0.019) or two (r_{max} = 0.042) of the 17 subpopulations would experience serious declines over 10 years (greater than 5%; Appendix B). Over 20 years, one subpopulation would experience extirpation and three would experience substantial declines if r_{max} = 0.019; if r_{max} = 0.042, no subpopulation would experience extirpation and two would experience substantial declines. Over 40 years, three subpopulations would experience extirpation and one would experience substantial declines if r_{max} = 0.019; if r_{max} = 0.042 two subpopulations would experience extirpation and none would experience substantial declines. The subpopulations identified around Nauru and the Gilbert Islands (Kiribati) are the two most impacted subpopulations and both experience serious declines in Scenario 1 within 10 years and extirpation within 40 years. These represent the primary hotspot identified in Figure 66.

By contrast, under our best estimate of the impacts of the fishery, Scenario 5 (considering a 75% misidentification rate), if initial population abundance is at least 1,000, no subpopulation experiences declines under any timeframe (10 to 40 years) under either value of r_{max} (0.019 or 0.042; Appendix B). Overall, in Scenario 5, we found that the combination of 20 or fewer annual interactions and population sizes of 1,000 or greater resulted stable or increasing population trajectories over 10 years if r_{max}=0.019, and as a result this level would not likely result in an appreciable reductions of the numbers, distribution or reproduction of giant manta ray subpopulations in the *Action Area*. Our analysis (see Appendix B, Tables 18 and 19) shows that some putative subpopulations may start to experience declines or extirpation based on the initial size of the population under 1,000 individuals given these anticipated take levels (see numbers in red). Specifically, if initial population sizes are 100 or fewer individuals, 9 of the subpopulations would experience extirpation or serious declines over all timeframes (10 to 40 years) when r_{max}=0.019. For initial abundances of 200 to 800, no subpopulation experiences extirpation within 10 years but at 40 years, three and one subpopulation/s are extirpated and three and two subpopulations experience substantial declines with initial abundances of 200 and 400 respectively, while two and one subpopulation/s experience substantial declines with initial abundances of 600 and 800 respectively.

However, given that localized extirpations of giant manta rays have occurred in other areas, it would be reasonable to expect that the continuing harvest of tens to a hundred animals from a single subpopulation in a given year for multiple years would lead to localized extirpation if they were in fact, all giant manta rays. That is to say, we would expect the historical number of interactions by this fishery would have already extirpated subpopulations in these areas based on our current knowledge of subpopulation abundances and when considering the level of interactions that have occurred over the years. Therefore, because giant manta rays continue to be harvested in large numbers each year by this fishery in areas like Nauru and Kiribati and continue to do so, we expect that the fishery is likely intercepting other ray species from this area in addition to giant manta rays. That some of these subpopulations, like Nauru and Kiribati, are not already extirpated due to historic harvest further supports the idea that the WCPO observer data likely contains an implausible number of individuals identified as giant manta ray.

In total, we looked at over 5,000 combinations of fishery interaction rates, initial population abundance, emigration, immigration, 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 outlined in this biological opinion several lines of evidence that suggest not all animals identified as giant manta ray were correctly identified in this fishery. At this time, the best scientific and commercial data available to identify the scale of any misidentification issues for giant manta rays is not ideal; however, it is all that we had at the time of this consultation and obtaining better data will take time, likely years.

While we have compelling evidence that giant manta ray are likely misidentified by observers in this fishery, the observer data remains the best available scientific and commercial data available for this fishery. We do not find it reasonable to consider the database ‘flawed’ even if the fishery likely has a problem with misidentified ray species. Instead, as noted earlier in this biological opinion, the data provide a plausible count of *Mobula* species and combined with other categories of *Mobula*, the fishery likely has a credible accounting of how many individuals of the genus *Mobula* it interacts with. Although we find the observer data on giant manta rays provides an implausible accounting of the true number of interactions that this fishery likely has with giant manta ray, we felt it prudent to examine the data as it was presented and recorded by observers, as well as using a best estimate of the misidentification problem to adjust the data for this analysis. As such, we examined capture rates for giant manta ray in the exact same way we analyzed capture rates for all species analyzed in the opinion, including partitioning unidentified individuals to analyzed species. Given the data as contained in the observer database, we estimated the number of interactions with giant manta rays using the aggregate data resulting in a mean of 165 (95th percentile: 199) giant manta rays would be captured by this fishery each year. Our scenarios (Scenarios 1 and 2) clearly show that if this level of interaction were true, it would result in appreciable reductions in numbers of giant manta ray and with those reductions in numbers, we would expect changes in reproduction and likely changes in distribution of at least some subpopulations. Under these scenarios, 4 of the 17 subpopulations would be expected to experience extirpation or a serious decline (defined as greater than 5%) over 10 years under

Scenario 1 if initial abundance is greater than 1,000 individuals. These predictions worsen as we extend our analytical time to 20 and 40 years, which is appropriate for this fishery consultation. These estimates highlight just how important it is for this fishery to prioritize and improve their identification of all *Mobula* species, and particularly threatened giant manta ray to verify the actual number of interactions this fishery is having with giant manta rays, and to ascertain estimates of post interaction mortality.

However, as noted in the *Effects of the Action* section, we consider Scenario 5 to be the most plausible of the scenarios we examined. As described earlier, this scenario is identical to Scenario 1, and is still based upon the best scientific and commercial data available, which is the observer data. However, under this scenario that data is adjusted to account for other sources of information that suggest a straightforward reading of the FFA observer data on giant manta rays does not likely provide an accurate accounting of the number of individual giant manta rays exposed to this fishery. Moreover, our analysis of scenario 1 leaves us perplexed as to why some of our putative subpopulations are not already extirpated when considering the level of interactions that have occurred in these areas in the past. So, we revised the Bayesian analysis for interaction rates and predictions of future anticipated number of interactions under the assumption that 75% of giant manta ray were misidentified and that 75% fewer unidentified mobulids should be assigned to giant manta ray than were in the initial analysis. In this analysis, we assumed that this proportion was overestimated by 75%, or that 9.21% ([36.85 x 25]/100) of unidentified mobulids were giant manta ray.

We recognize that the best available information suggests that giant manta rays occur in regionally distinct subpopulations that are likely composed of at least 1,000 individuals but may be as large as 25,000, but we do not know the number of subpopulations, which may be present in the *Action Area*, nor do we know the total abundance of giant manta rays in the *Action Area*. Therefore, our analysis incorporated the best information we have on giant manta ray population sizes and behavior from the literature to infer the population structure of giant manta rays in the *Action Area*. Assuming giant manta rays were incorrectly identified 75% of the time, we estimate that this fishery will interact with 45 (95th percentile: 50) giant manta rays per year into the future, with a maximum 5-year running average of 47 and up to 472 over 10 years, and about 43 (95th percentile: 48), or up to 455 over 10 years, will die as a result of their interaction. Assuming this fishery continues for at least another 25 years, based on the mean estimated number of interactions we would expect that the United States WCPO purse seine fishery would interact with about 1,175 (95th percentile: 1,250) more giant manta rays and about 1,128 (95th percentile: 1,200) of these animals would die from their interactions over a 25-year period.

NMFS expects that these interactions would be distributed across at least a few demographically independent subpopulations of giant manta ray in the *Action Area*, and using our best estimate is that these interactions would be distributed across about 17 putative subpopulations. Generally, we would expect that the greater number of populations that comprise a species, the greater the species' resilience. Based on what we know about metapopulations, such an intricate network of subpopulations can create a resiliency as subpopulations may rescue or recolonize areas where the species has declined or disappeared. Ultimately, it is reasonable to expect for species with such a complex population structure, the viability and importance of the various subpopulations would differ and some may be more important than others in ensuring the species' chances of survival and recovery. However, until we know more about the population structure of giant manta ray in the *Action Area*, we consider it prudent to ensure that all subpopulations "retain the potential to achieve viable status. This would ensure that all parts of the system are maintained (McElhany et al. 2000)" until we develop better information and can identify a different management strategy for the recovery of giant manta rays.

We know, based on our simulations that the ability of giant manta ray subpopulations to withstand fishing pressure is dependent on the size of the subpopulation, the extent of the annual mortality incurred by that subpopulation, and the maximum intrinsic population growth rates, $r_{\max}$. Until we know more about the species, and the underlying demographic processes that lead to and maintain numerous small, regionally distinct subpopulations, the only reasonable course of action is to assume that each subpopulation is important to the survival and recovery of the species.

We know that a common path that leads to the extinction of species begins with declines in the density of local or regional subpopulations of the species followed by gaps that represent the extinction of individual subpopulations, which is followed by the gaps in the global or regional distributions expanding as other subpopulations decline and become extinct (Udvardy 1969; Towns and Daugherty 1994; Lomolino and Channel 1995, 1998; Gaston et al. 2000; O'Grady et al. 2004; Reed 2004; Collen et al. 2011). The giant manta ray fits this pattern: their populations are declining globally except for a small number of populations that are not exposed to fishing and which may be stable (Miller and Klimovich 2017). Local and regional giant manta ray subpopulations have already been extirpated, creating gaps in the species' regional and global distribution.

Our analysis demonstrates that if there is no misidentification issue (Scenario 1, which given the evidence previously presented we do not consider this plausible), the authorization of the United States WCPO purse seine fishery as currently managed is likely to continue to contribute to the appreciable declines in *numbers*, *reproduction and distribution* of giant manta rays into the future and those reductions would be spread across a substantial number of putative subpopulations. However, as noted previously, there is a substantial amount of information that indicates there is an issue with misidentification in the observer data, and that our transformed data incorporating a 75% misidentification rate is the best available scientific and commercial information we have on which to base our analysis (Scenario 5).

Numbers, the second component of the jeopardy standard is the primary focus of this analysis. We distributed numbers, in terms of anticipated captures, across subpopulations using a reasoned approach, and under Scenario 5, we found that providing initial population sizes are larger than 1,000 individuals, the action does not cause appreciable declines in numbers of individuals under any timeframe (10 to 40 years). Further, in 10 years when a subpopulation experiences of 20 or fewer annual interactions and has a population size of 1,000 or greater then we expect stable or increasing population trajectories when $r_{\max}=0.019$.

We know that this species combination of long-lives, late-maturation, and low reproductive output (that is, single pup per pregnancy) makes this species highly vulnerable to overharvest, particularly harvests of adults (Dulvy et al. 2014; Croll et al. 2016; Miller and Klimovich 2017). The combination of small subpopulation size, small proportion of pregnant females, and low reproductive output (females produce a single pup and may only reproduce every 2 or 3 years), means these subpopulations have limited ability to recover from reductions in numbers. 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 as many as 4 to almost 9 decades to double in size has limited ability to recover from exploitation and disturbance, particularly when the exploitation or disturbance reduces their abundance every year. Hence, while we acknowledge that reproduction can be an issue with this species, given that, under Scenario 5, since we do not anticipate appreciable reductions in numbers, we similarly do not anticipate an appreciable reduction in *reproduction, which is the first component of the jeopardy standard*.

Finally, NMFS does not expect that the proposed action would extirpate any of the putative subpopulations of giant manta ray in the *Action Area* under the assumption that 75% of the species are misidentified. Therefore, NMFS does not anticipate the United States WCPO fishery would result in a substantial reduction in the species' *distribution*, which is the third component of the jeopardy standard.

Our analysis is based on a number of assumptions, which we have attempted to ensure are clearly explained in this opinion and the consultation record. As stated by Kuljis (2014) "there will likely always be some available science that contradicts the agency's ultimate conclusions." While it is not concerning necessarily that information may be available to contradict the agency, our concern is largely focused on: (1) did we identify and consider available information; and (2) did we consider it in a way that is reasoned, given the available evidence to help the reader understand what data or information is best, and why. We believe that we have done this. However, it seems important to return to the issue of the source data. The raw data provided to us by the fishery represents what is the "best available" and therefore formed the foundation for our analysis and our conclusion. However, even the "best available" data gives us reason to pause. Two important aspects of information or data error that scientists frequently are concerned with are precision versus accuracy. In this case, we suspect that the data provided for this consultation (numbers of interactions with giant manta rays) is more precise than it is accurate. Precision here refers to the fact that the data is collected relatively consistently from day to day, and vessel to vessel; importantly, it is possible to collect and deliver precise data without being accurate. Accuracy means that the values collected correctly reflect the measured or observed item. The "best" information would be both precise and accurate. For this action, we suspect the observer and interaction data for rays is more precise than it is accurate. In other words, we have reason to believe that the United States WCPO purse seine fishery may have a problem with species identification when it comes to rays, and that numbers of giant manta rays suspected of being captured in this fishery are less than have been reported. We reiterate our reasons for this conclusion here:

1. There is substantial range overlap among ray species, and we know that there are other ray species like the *M. tarapacana* and *M. alfredi* in the *Action Area* for this consultation, which suggests that mixed ray catches would be more likely than singular species catches particularly for this large *Action Area*.
2. Morphological similarities make it difficult to distinguish species (Stevens et al. 2018). Morphological variation occurs not only among species, but within populations and between populations of the same species occur. For instance, *M. birostris* has a coloration that ranges from melanistic (black) to leucistic (pale) with the most common color variation occurring in between the two (Stevens et al. 2018).
3. Given the large numbers of rays that this fishery has interacted with in the past, it is NMFS best professional judgement that it would be reasonable that some of the hardest

affected subpopulations would likely already be extirpated or at least have experienced substantial declines. Trends in catch data for the United States WCPO purse seine fishery indicates declines in CPUE, however these declines are not statistically significant (Figure 55).

4. We asked IFD for photos of giant manta ray in this fishery, and the only photos of rays (albeit they were few) were identified as *M. tarapacana*.
5. About 80% of the giant manta rays observed in this fishery and recorded with an estimated weight fall below the benchmark live-at-birth weight value of 0.0685 mt for giant manta rays, which suggests the estimates of captured individuals are poor, or the fishery is interacting with a different, smaller-bodied mobulid species (however, see the caveats to this analysis presented on page 253).
6. The limited number of other mobulid species besides giant manta ray identified in the observer database meant that 36.9% of unidentified mobulids were assigned to the giant manta ray category, which is likely to be an overestimate.

Therefore, based on the best available data and the analyses conducted as part of this biological opinion using the best available information; NMFS expects that the authorization of the United States WCPO purse seine fishery as currently managed is not likely to appreciably reduce the likelihood of either the survival and recovery of giant manta rays in the wild by reducing their *reproduction*, *numbers,* and *distribution* within the *Action Area*. While Scenario 5 provides us with the best estimate of the effect of the United States WCPO purse seine fishery on giant manta ray, we pine for better data on this species. We have made the most appropriate use of the best available data for this consultation, but in this situation, better data is required to check our assumptions and analysis. Thus, it is imperative that NMFS take immediate action to evaluate the severity of the misidentification problem as it relates to rays captured in this fishery, and more importantly to improve data collection for threatened giant manta rays.

7 CONCLUSION

After reviewing the *Status of Listed Resources*, the *Environmental Baseline* for the *Action Area*, the effects of the proposed action and the cumulative effects, it is NMFS' biological opinion that the operation of the United States WCPO purse seine fishery as currently managed is not likely to jeopardize the continued existence of the following species:

Endangered sei whales, endangered fin whales, endangered sperm whales, endangered leatherback sea turtles, endangered South Pacific loggerhead sea turtles, threatened Eastern Pacific green sea turtles, threatened East Indian-West Pacific green sea turtles, threatened Southwest Pacific green sea turtles, endangered Central West Pacific green sea turtles, endangered Central South Pacific green sea turtles, threatened olive ridley sea turtles and olive ridley sea turtles from the endangered Mexico breeding population, threatened oceanic whitetip sharks, threatened Indo-West Pacific scalloped hammerhead sharks, and threatened giant manta ray.

8 INCIDENTAL TAKE STATEMENT

Section 9 of the ESA and protective regulations pursuant to Section 4(d) of the ESA prohibit the take of endangered and threatened species without a special exemption. “Incidental take” is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity (50 CFR 402.02). Under the terms of Section 7(b)(4) and Section 7(o)(2), taking that is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the ESA provided that such taking is in compliance with the reasonable and prudent measures and terms and conditions of the Incidental Take Statement (ITS).

The measures described below are non-discretionary, 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 ITS. 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 ITS (50 CFR 402.14(I)(3)).

The proposed action results in the incidental take of endangered fin whales, endangered sei whales, endangered sperm whales, endangered leatherback sea turtles, endangered South Pacific loggerhead sea turtles, Eastern Pacific green sea turtles, East Indian-West Pacific green sea turtles, Central West Pacific green sea turtles, Southwest Pacific green sea turtles, Central South Pacific green sea turtles, endangered Mexico breeding population of olive ridley sea turtles, and threatened (other) populations of olive ridley sea turtles, threatened oceanic whitetip sharks, threatened Indo-West Pacific scalloped hammerhead shark, and threatened giant manta rays. Currently there are no take prohibition for oceanic whitetip sharks, Indo-West Pacific scalloped hammerhead shark, or giant manta ray; thus an ITS is not required to provide an exemption to the prohibition of take under Section 9 of the ESA for these three species. However, consistent with the decision in *Center for Biological Diversity v. Salazar*, 695 F.3d 893 (9th Cir. 2012), we have included an ITS to serve as a check on the no-jeopardy conclusion by providing a reinitiation trigger so the action does not jeopardize the species if the level of take analyzed in the biological opinion is exceeded.

8.1 MMPA Authorization

A marine mammal species or population stock that is listed as threatened or endangered under the ESA is, by definition, also considered depleted under the MMPA. The ESA allows takings of threatened and endangered marine mammals only if authorized by Section 101(a)(5) of the MMPA. When an action will result in incidental take of ESA-listed marine mammals, ESA Section 7(b)(4) requires that such taking be authorized under the MMPA Section 101(a)(5) before the Secretary can issue an ITS for ESA-listed marine mammals and that an ITS specify those measures that are necessary to comply with Section 101(a)(5) of the MMPA. Section 7(b)(4) and Section 7(o)(2) provide that taking that is incidental to an otherwise lawful agency action is not considered to be prohibited taking under the ESA if that action is performed in compliance with the terms and conditions of this ITS, including those specified as necessary to comply with the MMPA, Section 101(a)(5).

Section 101(a)(5)(E) of the Marine Mammal Protection Act (MMPA), 16 USC. 1361 *et seq.*, has provisions for NOAA’s National Marine Fisheries Service (NMFS), as delegated by the Secretary of Commerce, to issue permits for the taking of marine mammals designated as depleted because of their listing under the Endangered Species Act (ESA), 16 USC. 1531 *et seq.,* by United States vessels and those vessels which have valid fishing permits issued by the Secretary in accordance with Section 204(b) of the Magnuson-Stevens Fishery Conservation and Management Act, 16 USC. 1824(b), for a period of up to three years. NMFS may issue the authorization to take ESA-listed marine mammals incidental to these commercial fisheries only after the agency has determined, after notice and opportunity for public comment, that:

> (1) the incidental mortality and serious injury from commercial fisheries will have a negligible impact on the affected species or stock;
>
> (2) a recovery plan has been developed or is being developed for such species or stock under the ESA; and
>
> (3) where required under Section 118 of the MMPA, a monitoring program has been established, vessels engaged in such fisheries are registered in accordance with Section 118 of the MMPA, and a take reduction plan has been developed or is being developed for such species or stock.

MMPA 101(a)(5)(E) authorizations are required for commercial fisheries with frequent or occasional incidental M&SI of ESA-listed marine mammals (i.e., fisheries classified as Category I or Category II fisheries on the MMPA List of Fisheries (LOF). Authorizations are not required for commercial fisheries involving a remote likelihood of or no known incidental taking of marine mammals (i.e., fisheries classified as Category III fisheries on the LOF). Category III fisheries are those commercial fisheries that have a remote likelihood of or no known incidental M&SI of marine mammals (MMPA Section 118(c)(1)(A)(iii)). For the purposes of Section 101(a)(5)(E) of the MMPA, a commercial fishery classified as Category I or II on the LOF solely because of incidental M&SI of non-ESA listed marine mammals, would be considered a Category III fishery for negligible impact determination purposes, because the fishery has a remote likelihood or no known incidental M&SI of ESA-listed marine mammals (NMFS Procedural Directive 02-204-02). Additionally, in 60 FR 45399, NMFS announced that, consistent with the provisions of MMPA Section 101(a)(5)(E)(ii) and codified at 50 CFR 229.20(d), permits are not required for commercial fisheries involving a remote likelihood of or no known incidental taking of marine mammals.

Because the proposed action involves fisheries which have no documented incidental M&SI of ESA-listed marine mammals, as documented on the 2021 LOF (86 FR 3028) a 101(a)(5)(E) authorization under the MMPA is not required at this time. Vessels engaged in these fisheries shall not be subject to the penalties of the MMPA under Section 118, so long as the owner or master of such vessel reports any incidental mortality or injury of such marine mammals to the Secretary in accordance with Section 118.

8.2 Amount or Extent of Take

The following levels of incidental take may be expected to result from the proposed action. The reasonable and prudent measures, with their implementing terms and conditions, are designed to minimize the impact of incidental take that might otherwise result from the proposed action. NMFS uses causal inference to determine if individual threatened and endangered species, or their designated critical habitat, would likely be taken by harassing, harming, pursuing, hunting, shooting, wounding, killing, trapping, capturing, or collecting or attempting to engage in any such conduct. If take is anticipated to occur then the Services must describe the amount or extent of such anticipated take and the reasonable and prudent measures, and terms and conditions necessary to minimize the impacts of incidental take (FWS and NMFS 1998). If, during the course of the action, this level of incidental take is exceeded for any of the species as listed, NMFS IFD must immediately reinitiate formal consultation with NMFS PRD pursuant to the Section 7 regulations (50 CFR 402.16).

The mean annual numbers of interactions and mortalities predicted to result from the implementation of the proposed action, with the 95th percentile, maximum 5-year running average and running sums are shown in Table 83 below. This consultation establishes a maximum 5-year running sum and 5-year running average to monitor the action's incidental take of threatened and endangered species. We expect that annual interaction rates that should fall near to the predicted annual mean. Over any five consecutive years the number of interactions are not to exceed the defined 5-year running sum, which is five times the 5-year running average, rounded up to the nearest integer. The 95th percentile is designed to be used by NMFS as a check on the fishery's performance against the estimated take, and would act as a signal that the fishery may be on a path to exceed exempted take as denoted by the 5-year running sum (and average) of captures. If the 95th percentile is exceeded in two years, within a five-year period, it is unlikely the fishery will meet the maximum 5-year running average or running sum, and reinitiation will be warranted. Year to year variation in capture numbers is expected, and by managing the incidental take by the 5-year running sum this accounts for this annual variation, allowing for years with higher than average captures and years with lower than average captures. NMFS PRD anticipates that the following species could be taken as a result of the proposed action by capturing, harming, wounding, and killing:

Table 83. The average number of whales, sea turtles, oceanic whitetip shark, Indo-West Pacific scalloped hammerhead shark, and giant manta ray interactions expected from the proposed action during one calendar year, with the 95th percentile, maximum 5-year running average and running sums. The table also includes total mortalities (males and females, adults and juveniles) expected to result from this number of interactions.

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**		
Sei whale	3	7	0	5.98	**30**
Sperm whale	0	1	0	0.67	**4**
Fin whale	0	1	0	0.75	**4**
Hawksbill sea turtle	4	6	1 to 2	4.53	**23**
Leatherback sea turtle	1	1	0	0.47	**3**

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**		
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**
**Oceanic whitetip shark	88	160	135	102.7	**514**
**Indo-West Pacific scalloped hammerhead shark	2	2	2	1.99	**10**
**Giant manta ray	45	50	48	47***	**236**

*The total number of interactions for the species and populations can be any combination from the listed populations for olive ridley sea turtles or green sea turtle species considered herein.

**An ITS is not required to provide an exemption on the prohibition of take for the giant manta ray, Indo-West Pacific scalloped hammerhead shark, and oceanic whitetip shark because there are no take prohibitions under ESA Section 4(d) for these species. Consistent with the decision in *Center for Biological Diversity v. Salazar*, 695 F.3d 893 (9th Cir. 2012) however, this ITS is included to provide a reinitiation trigger if the level of take analyzed in the biological opinion is exceeded.

8.3 Reasonable and Prudent Measures

Reasonable and prudent measures are those actions that are necessary or appropriate to minimize the impacts, i.e., amount or extent, of incidental take that is anticipated to result from the proposed action (50 CFR 402.02). The associated terms and conditions set out the specific methods by which the reasonable and prudent measures are to be accomplished. Reasonable and prudent measures along with the terms and conditions that implement them cannot alter the basic design, location, scope, duration, or timing of the action, and may involve only minor changes.

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 United States WCPO purse seine 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 and must be undertaken by NMFS IFD for the exemption in ESA Section 7(o)(2) to apply.

1. NMFS shall ensure that data are collected for the United States WCPO purse seine fishery on the capture, injury, life history, and mortality of all threatened and endangered sea turtles, marine mammals, and elasmobranchs.

2. NMFS shall take actions to minimize the effects of FADs on ESA-listed species for the United States WCPO purse seine fishery.

3. NMFS shall develop and implement minimization measures to reduce the incidental capture and mortality of marine mammals in the United States WCPO purse seine fishery.

8.4 Terms and Conditions

NMFS shall undertake and comply with the following terms and conditions to implement the reasonable and prudent measures identified in Section 8.4 above. These terms and conditions are non-discretionary, and if NMFS 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. In order to be exempt from the prohibitions of Section 9 of the ESA, the Federal action agency must comply (or must ensure that any applicant complies) with the following terms and conditions.

1. The following terms and conditions implement reasonable and prudent measure No. 1

    a. NMFS shall work with the relevant regional observer programs, such as the WCPFC and FFA, and technical experts such as the SPC, to improve the collection of standardized information regarding the incidental capture, injury, and mortality of threatened and endangered sea turtles, marine mammals and elasmobranchs (all protected species) particularly with respect to species identification, fate, condition, handling, and set information for interactions with protected species.

    b. NMFS shall work with the relevant regional observer programs, such as the WCPFC and FFA, to explore carcass/sample retention of ESA-listed species for scientific purposes to illustrate the life history characteristics of animals encountered in this fishery before returning them to the ocean. This shall include exploring methods to train observers in the collection of genetic samples from sea turtles, oceanic whitetip sharks, and scalloped hammerhead sharks, and other protected species in order to determine which population(s) are affected by the fishery.

c. NMFS shall, on an annual basis, acquire and analyze all available protected species data from the United States WCPO purse seine fishery, to monitor actual take of threatened and endangered species against the take exemptions provided in this biological opinion, including monitoring the actual take of those threatened species with no 4(d) prohibitions to serve as a check on the agency's decision that the incidental take of these species is not likely to jeopardize their continued existence. Unidentified animals should be prorated appropriately to ESA-listed species. This report of the previous year's actual and estimated take should be provided to NMFS PRD annually, by the end of each calendar year.

2. The following terms and conditions implement Reasonable and Prudent Measure No. 2:

   a. NMFS shall develop regulations to implement the provisions of CMM 2018-01, regarding non-entangling FAD design requirements for purse seine fishing vessels.

   b. NMFS shall explore, including through the WCPFC, data collection options to help determine the effects of FADs on ESA-listed species, including methods to ascertain the frequency with which FADs are checked to determine if ESA-listed species are entangled, the number of ESA-listed species entangled in FADs on an annual basis, and the number of FADs that are lost by each vessel on an annual basis.

   c. NMFS shall work with industry to develop best management practices for FADs to reduce the loss or abandonment of FADs, including techniques for removing all threatened and endangered sea turtles, marine mammals, and elasmobranchs based on the species' biology, and minimizing adverse effects from FADs in order to promote protection and conservation for threatened and endangered species. The best management practices may include practices for:
      - Removing/retiring FADs that are no longer in use by the vessel that deployed them.
      - Tracking and retrieving missing or lost FADs to reduce the chances of FADs becoming marine debris.
      - Making every reasonable effort to remove and release any entangled threatened and endangered sea turtles, elasmobranchs, and marine mammals with as little harm as possible when inspecting or removing a FAD.

   d. NMFS shall work in cooperation with other fishing States in the region, such as through the WCPFC, to explore and implement improved FAD designs and materials, and tracking and retrieval systems, to lessen the likelihood of their becoming marine debris and/or to lessen adverse impacts of such marine debris.

   e. NMFS IFD shall provide an annual update to NMFS PRD on the FAD related data collection described in paragraph b (above).

3. The following terms and conditions implement Reasonable and Prudent Measure No. 3:

a. NMFS shall develop regulations to implement the provisions of WCPFC CMM 2011-03, and continue to work in the WCPFC to explore and agree on measures to further reduce the number and severity of interactions with cetaceans, including consideration of measures that would: (a) prohibit vessels from setting on a school of tuna with a cetacean, by requiring vessel owners and operators to attempt to identify whales before setting the net, monitoring during the set; (b) ensure that all reasonable steps are taken to ensure the safe release of cetaceans including stopping the net roll and not recommencing fishing operation until the animal has been released and is no longer at risk of recapture; (c) include methods to mitigate whale interactions, such as requiring the area be monitored for a certain period of time before the net is set to determine their presence, establishing an electronic monitoring program to monitor setting and hauling, etc.; (d) prohibit vessel owners and operators from chasing and corralling marine mammals out of the net in an aggressive manner when incidentally captured to limit a whale's stress level(s). Whales may be escorted to the corkline in a controlled manner; and (e) require incident reporting including details of the species (if known) and number of individuals, location and date of such encirclement, steps taken to ensure safe release, and an assessment of the life status of the animal on release (including, if possible, whether the animal was released alive but subsequently died), and to facilitate the determination of mortality and serious injury events, including data on the presence of other marine mammals and the duration of the incident (from start of capture to release).

b. NMFS shall work with the relevant regional observer programs, such as the WCPFC and FFA, and relevant species experts, to improve the collection of data on marine mammal interactions.

8.5 Conservation Recommendations

1. NMFS should develop and implement minimization measures to reduce the incidental capture and mortality of ESA-listed sharks in the United States WCPO purse seine fishery.

   a. NMFS should work with industry to develop best handling practices to remove sharks from the purse before they are brailed, to the maximum extent practicable and in a way that is safe for the crew and protective of the physiology of sharks. NMFS IFD should provide a report of best handling practices to NMFS PRD with an action plan for working with the fleet to reduce shark interactions and associated mortality.

   b. NMFS should develop regulations to implement the provisions of WCPFC CMM 2019-04,which includes provisions to (a) allow for an exemption for purse seine vessels to retain sharks in cases where oceanic whitetip shark are not seen during fishing operations and are delivered to the vessel hold, are retained for scientific sampling or sufficient tissue is retained for scientific analyses, and (b) require, when an observer or monitoring camera is on board that sharks be hauled alongside the vessel before being cut free to facilitate species identification. NMFS should assess whether increases in the survival of oceanic whitetip sharks or Indo-West Pacific scalloped hammerhead sharks that interact with the United States WCPO purse seine fishery could be made through regulatory requirements, and work with PRD to craft any appropriate revisions.

2. NMFS should develop and implement minimization measures to reduce the incidental capture and mortality of giant manta rays in the United States WCPO purse seine fishery.

    a. NMFS should determine the extent to which giant manta rays are misidentified in the United States WCPO purse seine fishery, and work with experts to improve species identification. NMFS IFD should provide NMFS PRD with a plan for assessing the magnitude of the species identification problem not later than 6 months from the signing of this biological opinion. NMFS IFD should develop and use best practices for estimating interaction and mortality rates for giant manta rays, following their plan and should provide a description of the method used to calculate interaction data on giant manta rays to PRD within two years of the signing of this biological opinion.

    b. NMFS should work with the relevant regional observer programs, such as the WCPFC and FFA, and relevant species experts, to improve species identification of giant manta ray in the WCPO.

    c. NMFS should develop regulations to implement the provisions of WCPFC CMM 2019-05, which includes provisions that require vessels to promptly release mobulid rays, alive and unharmed, to the extent practical, as soon as possible and in a manner that will result in the least possible harm to the individuals captured; provide an exemption in cases where a mobulid ray not seen during fishing is delivered into the vessel hold, or for the collection of biological samples from any such animals; and require that vessels allow for observers to collect biological samples of mobulid rays that are dead at haul-back.

    d. NMFS should use the population centers, as noted in Table 56 and Table 68 of this biological opinion, to monitor and, to the maximum extent practicable, limit annual capture of giant manta rays to no more than 20 captures from any single putative subpopulation while keeping within the aggregate 5-year running average of 47. NMFS should record captures in the United States purse seine fishery that occur within a 500 km radius of each putative subpopulation centerpoints identified in this biological opinion and attribute those captures to that subpopulation. In addition, 15% of captures that occur between 500 and 1500 km from the centerpoint will also be attributed to that subpopulation. NMFS IFD should place particular emphasis on threatened giant manta rays in the geographic areas identified in this consultation as the Nauru/Kiribati hotspot (from a center point of 0°32'21.715"S 169°3'22.948"E in Kiribati, to extend a radius of 700 km in all directions) and north of

    Bougainville, Solomon Islands in the east side of PNG’s EEZ (PNG East; from a center point of 3°41'47.808"S 156°51'8.868"E in Papua New Guinea to extend a radius of 275 km in all directions).

  e. NMFS IFD should analyze fishery-wide data for giant manta ray interactions to see if any additional catch or geographic trends may be present within the data,

3. NMFS IFD should work with the SPC to evaluate reports of discarded marine debris from United States vessels and work with the industry to reduce the amount of pollution from the United States WCPO purse seine fishery.

8.6 Reinitiation Notice

This concludes formal consultation on the operation of the United States WCPO purse seine 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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10 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 (corals). Next we reviewed whether we had data (observations) on the species exposure, or reasoned information that exposure could occur (potential) to one or more of the action's stressors: fishery interactions (e.g., vessel noise, vessel collision, or entanglement in gear); vessel waste, discharge, and emissions. We ended up with four general exposure scenarios to which a species was assigned based on its primary geographic area or occupancy: (1) observed pelagic exposure; (2) observed coastal exposure; (3) potential pelagic exposure; 4) potential coastal exposure. Each exposure profile that results for each species, 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.

Therefore, the species listed in the potential coastal exposure would not be exposed to entanglement/entrapment from fishing gear; however, they would likely be exposed to vessel transit and vessel noise. The unoccupied observed coastal exposure profile reflects the number of documented interactions (vessel collisions) with coastal species. There were no accounts in the observer data 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 entrapment or entanglement. These data shaped the observed pelagic exposure profile. Although species within this exposure profile may collide with a vessel, the primary stressor from the proposed action is capture in gear (entrapment or entanglement). Lastly, the potential pelagic exposure is comprised of species that 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 United States WCPO purse seine fishery.

This subsection also summarizes the results of our evaluations of the probability of a response given exposure, which considers susceptibility. Here we briefly describe the species and stressors that are not likely to adversely affect listed species, and our reasoning for this conclusion.

In addition, vessel noise, wastes and discharges, and vessel emissions would have discountable[3] effects on leatherback sea turtles, loggerhead sea turtles, all green sea turtles under consideration, olive ridley sea turtles, sei whales, sperm whales, fin whales, giant manta rays, oceanic whitetip sharks, and Indo-West Pacific scalloped hammerhead sharks. However, because interactions

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[3] The Services' 1998 ESA Section 7 Consultation Handbook contains the definition, 'Discountable effects are those exteremly unlikely to occur." When the terms "discountable" or "discountable effects" appear in this document, they refer to potential effects that are found to support a "not likely to adversely affect" conclusion because they are extremely likely to occur. The use of these terms should not be interpreted as having any meaning inconsistent with our regulatory definition of "effects of the action".

with purse seine gear are observed for all of these species (that is, these species fall into observed pelagic exposures category) we discuss these species in greater detail later in this biological opinion (see section 6, *Effects of the Action*).

Table 84. 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**
Sei whale	North Pacific Loggerhead sea turtle
Sperm whale	Southeast Indo-Pacific Loggerhead sea turtle
South Pacific Loggerhead sea turtle*	Central North Pacific green sea turtle
Leatherback sea turtle*	Eastern Pacific scalloped hammerhead shark
Green sea turtle*[4]	Guadalupe fur seal
Olive Ridley sea turtle*	Mexico Humpback whale
Hawksbill sea turtle*	Central America Humpback whale
Oceanic white-tip shark	
Giant manta ray*	
Indo-West Pacific scalloped hammerhead shark	
Fin whale	
Observed Coastal Exposure	**Potential Coastal Exposure**
none	*Acropora globiceps*
	Acropora jacquelineae
	Acropora lokani
	Acropora pharaonis
	Acropora retusa
	Acropora rudis
	Acropora speciosa
	Acropora tenella
	Anacropora spinosa
	Euphyllia paradivisa
	Isopora crateriformis
	Montipora australiensis
	Pavona diffluens
	Porites napopora
	Seriatopora aculeata
	Hawksbill sea turtle*
	Olive Ridley sea turtle*

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[4] Central South Pacific, Southwest Pacific, East Pacific, Central West Pacific, and East Indian-West Pacific green sea turtles.

Observed Coastal Exposure	Potential Coastal Exposure
	Loggerhead sea turtle (both North and South Pacific)*
	Leatherback sea turtle*
	Green sea turtle (all species)*
	Giant manta ray*
	Chambered nautilus
	Guadalupe fur seal

10.1 Potential Coastal Exposure

By far the largest category, 36 species fall into the category of potential coastal exposures. This includes: *Acropora globiceps; Acropora jacquelineae; Acropora lokani; Acropora pharaonis; Acropora retusa; Acropora rudis; Acropora speciosa; Acropora tenella; Anacropora spinosa; Euphyllia paradivisa; Isopora crateriformis; Montipora australiensis; Pavona diffluens; Porites napopora; Seriatopora aculeate*; chambered nautilus; hawksbill sea turtle, olive ridley sea turtle, loggerhead sea turtle (both North and South Pacific), leatherback sea turtle, green sea turtle (all species), Guadalupe fur seal, and giant manta ray. As mentioned above, fishing occurs offshore in pelagic waters. As such, these species are exposed to all or any of the stressors emanating from vessel transits (e.g., vessel collision, vessel noise, vessel waste, discharge and emissions).

10.2 Observed Coastal Exposure

Purse seine fishing does not occur 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.

10.3 Potential Pelagic Exposure

Five species fall into the potential pelagic exposure category for fishery exposures: Western North Pacific humpback whales, North Pacific loggerhead sea turtle, Southeast Indo-Pacific loggerhead sea turtle, and Central North Pacific green sea turtles. 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).

An additional four species fall into the potential pelagic exposure category for the vessel transit routes to ports only (i.e. they are not anticipated to occur in the fishery portion of the *Action Area*): Mexico humpback whale, Central America humpback whale, Guadalupe fur seal, and East Pacific scalloped hammerhead shark. These species would only be exposed to stressors emanating from vessel transits (e.g., vessel collision, vessel noise, vessel waste, discharge and emissions). These stressors may result in immediate exposures, or exposures later in time (e.g., interaction with derelict gear or vessel waste).

10.4 Observed Pelagic Exposure

Fifteen species fall into the observed pelagic exposure category: fin whale, sei whales, sperm whales, South Pacific loggerhead sea turtles, leatherback sea turtles, East Pacific green sea turtles, Central South Pacific green sea turtles, Central West Pacific green sea turtles, Southwest Pacific green sea turtles, East Indian-West Pacific green sea turtles, olive ridley sea turtles, hawksbill sea turtles, oceanic white-tip sharks, Indo-West Pacific scalloped hammerhead shark, 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, entanglement in gear); and vessel waste, discharge, or emissions. These stressors may result in immediate exposures, or exposures later in time (e.g., interaction with derelict gear or vessel waste). As previously described, there are documented observations of interactions with fishing gear resulting in entrapment or entanglement with the species listed in the observed pelagic exposure profile and are described in greater detail of this biological opinion in the *Effects of the Action* section.

10.5 Stressors Not Likely to Adversely Affect Listed Resources

10.5.1 Vessel Noise

Man-made sounds can affect animals exposed to them in several ways such as: non-auditory damage to gas-filled organs, hearing loss expressed in permanent threshold shift (PTS) or temporary threshold shift (TTS) hearing loss, and behavioral responses. They may also experience reduced hearing by masking (i.e. the presence of one sound affecting the perception of another sound). Masking and behavioral avoidance are the most likely responses of animals in the vicinity of United States WCPO purse seine fishing vessels. However, NMFS expects that vessel noises would have an insignificant effect on listed species because they would not be expected to result in measurable responses (should never reach the scale where take occurs).

Given the size of the United States WCPO purse seine fishery (the small number of vessels in the fishery and the wide area they cover), the fact that the sound field produced by the vessels in the fishery is relatively small and would move with the vessel, the animals would be moving as well, vessel speeds would be slow, vessel transit vectors would be predictable, sudden or loud noises would be unlikely or infrequent, and generally the sound field would be in motion, 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, NMFS expects this stressor would have insignificant effects on the ESA-listed resources in Table 3 and Table 84.

10.5.1 Collision with vessels

The proposed action would expose all ESA-listed marine species under NMFS' jurisdiction found in both the coastal and pelagic exposure categories (both potential and observed) to the risk of collision with vessels. The vessels in the United States WCPO fishery range in size from 53-79 m (WCPFC 2021b). They are steel-or fiberglass-hull vessels that travel at speeds less than 10 kt (NMFS 2018b). Since 2009, the number of active vessels has remained relatively consistent at 39 vessels (NMFS 2017a) until a marked decline was noted in 2019. As previously discussed, only 20 vessels are currently authorized to fish in this fishery and we do not expect any additional entrants into the fishery at this time.

Even in high-density turtle areas, such as those around Hawaii, collisions with vessels are relatively rare events. NMFS conservatively estimated 37.5 sea turtle vessel strikes and mortalities per year from an estimated 577,872 vessel trips per year in Hawaii. This includes fishing and non-fishing vessels (NMFS 2008b). This calculates to a 0.006% probability of a vessel strike with sea turtles for all vessels and trips, and many of these vessels are not likely reducing speeds or employing lookouts for listed species. We expect the abundance of oceanic whitetip sharks, giant manta rays, and chambered nautilus to be less than that of turtles in these regions. Additionally, turtles are required to surface to breathe whereas elasmobranchs and cephalopods are not, reducing their potential exposure to this stressor.

The majority of tuna from this fishery is landed in Pago Pago, American Samoa; Majuro, Republic of the Marshall Islands, and in Pohnpei, Federated States of Micronesia (Table 1). Tuna is also landed in several other countries at much lower numbers (Table 1). All of these islands have a considerably smaller density of sea turtles in their surrounding waters compared to the density of green sea turtles around the Hawaiian Islands. In high-density areas like Hawaii, collisions between turtles and vessels are relatively rare events. We expect that the chances of a United States WCPO purse seine fishery vessel strike is even less due to the lower density of turtles around the islands where tuna is landed compared to the density around Hawaii. As a result, the risk of collision with a United States WCPO purse seine fishery vessel is considered highly unlikely for sea turtles.

Similarly, collisions between whales and vessels are relatively rare events in the Pacific. Lammers et al. (2013) analyzed 37 years of data and determined that a total of 68 collisions with humpback whales occurred over that time in Hawaiian waters. Vessel traffic around Hawaii is likely much higher than in either the fishery portion of the *Action Area* or the common ports used to offload tuna. As a result, the risk of collision with a United States WCPO purse seine fishery vessel is considered highly unlikely for whales.

Additionally, chambered nautiluses are closely associated with steeply-sloped forereefs and muddy bottoms and are found in depths typically between 200 and 500 m and are not known to swim in the open water column nor found in shallow water depths except for rare occasions when the water is cold enough (Miller 2018). Open ocean environments and specific temperature gradients are considered geographic barriers to movement as the species does not swim through the mid-water (Miller 2018). Therefore, it is highly unlikely a chambered nautilus would be exposed to vessels at the surface within this fishery. Additionally, for this reason, we only considered the species in the coastal exposure group and not the pelagic exposure group even though the estimated species range in Miller (2018) (see page 6) may contain areas of strictly pelagic habitat where depths would exclude potential interactions with the fishery.

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, the expectation that ESA-listed marine species would be widely scattered throughout the proposed *Action Area*, the potential for an incidental vessel strike is extremely unlikely to occur. Thus, NMFS expects this stressor would have discountable effects on the ESA-listed resources in Table 3 and Table 84.

10.5.2 Introduction of Vessel Wastes and Discharges, Gear Loss, and Vessel Emissions

The diffuse stressors associated with the purse seine fisheries: vessel waste discharge, gear loss, and carbon emissions and greenhouse gasses, can affect both pelagic and coastal areas. ESA-listed resources could be exposed to discharges, and run-off from vessels that contain chemicals such as fuel oils, gasoline, lubricants, hydraulic fluids and other toxicants. Richardson et al. (2017) assessed SPC/FFA data specifically looking at the GEN-6 reporting form and found marine pollution attributed to purse seine fleets in the WCPO were mostly composed of plastic waste and discarded fishing gear. Between 2003 and 2015, the distant water fleets were responsible for 71% of the total reported incidents with the United States WCPO fleet accounting for upwards of 15% of these events (Richardson et al. 2017). In December of 2017, the WCPFC adopted CMM 2017-04, which was implemented on January 1, 2019, which engage in various pollution prevention techniques to reduce discharges and wastes into marine waters.

United States WCPO purse seine fishery vessels also burn fuel and emit carbon into the atmosphere during fishing operations and transiting. Parker et al. (2018), estimates that in 2011, the world’s fishing fleets burned 40 billion liters of fuel and emitted 179 million tons of carbon dioxide greenhouse gasses into the atmosphere. Between 1990 and 2011, emissions grew by 28% primarily due to increased harvests of crustaceans, a fuel intensive fishery (Parker et al. 2018). While we don’t have an accurate estimate of the carbon footprint of the United States WCPO purse seine 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, gear loss and vessel emissions would occur as a result of the United States WCPO purse seine 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, NMFS expects that this stressor would have discountable effects on the ESA-listed resources in Table 3 and Table 84.

10.6 Species Not Likely to be Adversely Affected

Based on the preceding discussion the following species are not likely to be adversely affected by the proposed action because they occur in coastal areas only, where they are not subject to fishing but are only subject to potential stressors related to vessel transiting or FAD groundings. As previously discussed, it has been shown that FADs with GPS transmitters have grounded in nearshore environments and can cause damage (State of Hawaii DLNR 2014; Balderson and Martin 2015; Maufroy et al. 2016, 2018; Davies et al. 2017). While it is possible a FAD could damage a sessile marine invertebrate (i.e. ESA-listed corals) we consider these species are present over a vast geographic expanse, occur at varying depths, are sporadic in distribution, and that a FAD is highly unlikely to contact a colony of ESA-listed coral. Furthermore, each FAD is unique, may be disintegrated to some degree, or may have long trailing parts (i.e. variable size and remnant parts), and ocean conditions are extremely diverse over this geographic scale.

We are aware of two events from United States flagged vessels where FADs grounded. One was set by a purse seine vessel fishing in the EPO which grounded in Hawaii. The FAD had a long trailing sausage net which contacted and was stuck in a coral colony. No damage was noted to the colony, the netting was simply caught. However this was from a separate fishery, and outside of the *Action Area* where fishing and deployment of FADs from this fishery occurs. The second FAD we are aware of grounded on Palmyra Atoll, however we are unsure where the FAD was deployed or from what fishery it belonged too.

At this time, we do not have sufficient information to describe effects given the lack of data regarding FADs in this fishery or the likelihood that they would contact a single colony in an open ocean nearshore environment. Overall, while plausible, we believe their exposure is considered highly unlikely and therefore discountable. These are: 15 species of corals (*Acropora globiceps, A. jacquelineae, A. lokani, A. pharaonis, A. retusa, A. rudis, A. speciosa, A. tenella, Anacropora spinose, Euphyllia paradivisa, Isopora crateriformis, Montopora australiensis, Pavona diffluens, Porites napopora, Seriatopora aculeate),* and chambered nautilus.

As noted in the *Description of the Proposed Action* section, vessels that fish in the WCPO may transit through and fish in the East Pacific on the same trip, offloading catch from the WCPO in ports in Ecuador, Mexico, El Salvador, and Peru. The fishing that occurs in the East Pacific is managed under the West Coast Highly Migratory Species Management Plan and its implementing regulations (see 50 CFR 660 Subpart K), as well NMFS regulations implementing decisions of the IATTC under the Tuna Conventions Act (16 U.S.C. 951 et seq.) (See 50 CFR 300 Subpart C) and NMFS regulations implementing the Agreement of the International Dolphin Conservation Act under the Marine Mammal Protection Act (16 U.S.C. 1361 et seq.) (See 50 CFR 200 Subpart C). NMFS completed ESA Section 7 consultation for the U.S. EPO purse seine fishery in 2004 (NMFS 2004). Hence, the following species are not likely to be adversely affected by the proposed action because they only occur in pelagic habitats in the East Pacific where they are not subject to potential stressors related to United States WCPO purse seine fishery sets, but are only subject to potential stressor related to vessel transiting. Therefore, their exposure is considered highly unlikely and therefore discountable. These are: Mexico humpback whales, Central America humpback whales, Eastern Pacific scalloped hammerhead sharks, and Guadalupe fur seals.

Our analysis demonstrates that several species may be exposed to fishing, yet none of the species listed in the potential pelagic exposure category have been caught in United States WCPO purse seine fishery. We have concluded, based on the information that follows, that the effect of the action on the following species is discountable. As such, the following species are also not likely to be adversely affected by the proposed action: blue whales, Central North Pacific green sea turtles, North Pacific loggerhead sea turtles, Southeast Indo-Pacific loggerhead sea turtles, Western North Pacific humpback whales.

10.6.1 Blue Whales

Blue whales are found in the *Action Area* and there have been three recorded observations of blue whale sightings by United States WCPO purse seine fishery observers (the animals were observed in the distance and not in association with a fishery set). In addition, there has been one recorded interaction with a blue whale, however, the interaction was defined as occurring on a 'live whale' suggesting the vessel essentially used the whale as a FAD to locate its target species and made the set. As described in the *Effects of the Action,* such a set is unlawful and therefore the take cannot be exempted. Blue whales are the largest animal to have ever lived on the planet, reaching lengths of 30 meters (98 feet; NMFS 2018f) and the recorded observations of the species at distances greater than 100 meters from the vessel confirm that their presence is readily observed. It is unlikely that any blue whales in the area would be unnoticed by the fishing master, therefore any direct interaction of a blue whale with purse seine fishing gear would be the result of an unlawful deliberate set. Given the size of blue whales, we consider the possibility of an incidental interaction with the United States WCPO purse seine fishery to be discountable and therefore the action is not likely to adversely affect blue whales.

10.6.2 Central North Pacific Green Sea Turtles

The majority of nesting for the Central North Pacific green sea turtle occurs in the Northwestern Hawaiian Islands (NMFS and FWS 1998a; Balazs et al. 2015), with nearshore juvenile and adult forage habitat concentrated around the Hawaiian Islands and Johnston Atoll, both of which are outside of the range of the United States WCPO purse seine fishery. The ASLL fishery overlaps some with the United States WCPO purse seine fishery and interacts with green sea turtles which are sampled for genetics (*n* = 31); however, none have been from the Central North Pacific (NMFS unpublished data). Hence, it is unlikely that Central North Pacific green sea turtles interact with the United States WCPO purse seine fishery. Therefore, we conclude that the Central North Pacific green sea turtle is not likely to be adversely affected by the proposed action.

10.6.3 North Pacific Loggerhead Sea Turtle

Based on several studies that have analyzed tracks from more than 250 satellite-tagged North Pacific loggerhead sea turtles within the Pacific, we do not expect the North Pacific loggerhead sea turtle to occur within the *Action Area* (Kobayashi et al. 2008; Howell et al. 2010; Abecassis et al. 2013; Briscoe et al. 2016a). The satellite-tagged turtles represent both captive-reared and wild sea turtles, the captive-reared sea turtles were released into the ocean offshore of Japan and the wild sea turtles were captured by Hawaii-based longline vessels (Abecassis et al. 2013). No overlap between the *Action Area* and satellite location data for North Pacific loggerhead sea turtles was detected.

Briscoe et al. (2016a) compared satellite tracks of 44 juvenile North Pacific loggerhead sea turtles released offshore of Japan with the results of an oceanic particle dispersal model for passive transport. While the northern-most extent of the *Action Area* is 11°N, their results for the sea turtle satellite tracks were consistent with previous studies showing no tracks south of 20°N in the western Pacific, and most tracks north of 30°N. In contrast, the particle dispersal model indicated substantial particle dispersal between 20°and 30°N, extending down to 10°N in the westernmost North Pacific. These results underscore that North Pacific loggerhead sea turtles preferentially use habitats north of at least 20°N. Hence, it is unlikely that North Pacific loggerhead sea turtles would be exposed to the United States WCPO purse seine fishery. Therefore, we conclude that the United States WCPO purse seine fishery is not likely to adversely affect the North Pacific loggerhead sea turtles.

As North Pacific loggerhead sea turtles use foraging habitat off of Baja California, it is possible that they would be exposed to United States WCPO purse seine vessels as the are transiting to ports in Mexico and South America, however, as described above, stressors associated with vessels (eg. noise, collision and exposure to waste) are anticipated to have insignificant or discountable effects on listed species and therefore are not likely to adversely affect North Pacific loggerhead sea turtles.

10.6.4 Southeast Indo-Pacific Loggerhead Sea Turtle

Nesting for the Southeast Indo-Pacific loggerhead sea turtle is confined to the west coast of Australia in the Southeast Indian Ocean (Baldwin et al. 2003). Hamann et al. (2013) estimated ecological range for this species using data from molecular studies, migration behavior, tag recoveries, and expert opinion. They determined that the eastern-most extent of foraging and migration for this species is the Western Torres Strait at 141°E.

Southeast Indo-Pacific loggerhead sea turtles also use forage habitat around Indonesia and the Arafura Sea south of Papua New Guinea (Hamann et al. 2013). There is no geographic overlap between these foraging areas and the fishing area used by the United States WCPO purse seine fishery (Figure 4). Hence, it is unlikely that Southeast Indo-Pacific loggerhead sea turtles will interact with the United States WCPO purse seine fishery. Therefore, we conclude that the United States WCPO purse seine fishery is not likely to adversely affect the Southeast Indo-Pacific loggerhead sea turtle.

10.6.5 Western North Pacific, Mexico, and Central America Humpback Whales

NMFS recognizes 14 species of humpback whales, and of those three have the potential to interact with the United States WCPO purse seine fishery sets: The Western North Pacific, the East Australia, and the Oceania humpback whale (81 FR 62259). The East Australia and Oceania humpback whales are not listed under the ESA, the Western North Pacific humpback whale is listed as endangered. Two additional species of humpback whales (Mexico, listed as threatened, and Central America, listed as endangered) use winter habitats off of Baja California and Central America, and it is possible that they would be exposed to United States WCPO purse seine vessels as they are transiting to ports in Mexico and South America. However, as described above, stressors associated with vessels (eg. noise, collision and exposure to waste) are anticipated to have insignificant or discountable effects on listed species and therefore are not likely to adversely affect Mexico or Central America humpback whales.

The estimated breeding distribution of the East Australia humpback whale extends east to approximately 165°E and is between approximately 16 and 21°S, but members of this species are seen as far north as 10°S (Bettridge et al. 2015). The Oceania humpback whale breeds near the South Pacific Islands and feeds in an unknown location in the Antarctic region (Bettridge et al. 2015). The Western North Pacific humpback whale uses foraging grounds in the North Pacific, primarily off the Russian coast but potentially in the Aleutian Islands as well (Bettridge et al. 2015). The breeding area for this species extends from approximately 10°N to 30°N. The Oceania and East Australia humpback whales would be expected to occur only below the equator, while the Western North Pacific humpback whale would be expected to occur only above the equator (B. Brownell, NOAA Fisheries, pers. comm.). Observations and interactions with humpback whales with the United States WCPO purse seine fishery occurred both above and below the equator and East and West of 165°E, suggesting that the sightings and interactions could have been from any or all of the three species. Although the sightings and interactions were north of 21°S, they were within the range in which individuals of the East Australia humpback whale have been observed.

These sightings and interactions are much further from Okinawa/Philippines, which represents the core breeding area of the Western North Pacific humpback whale than from breeding areas of the Oceania and East Australia humpbacks. Only Western North Pacific humpback whales are expected to occur above the equator, but the sightings and interactions with humpback whales in the tuna purse seine fishery do not indicate an obvious gap in distribution at the equator to signify a distinction between humpback whales that might approach the equator from the Oceania, East Australia, and Western North Pacific humpbacks. The northernmost sighting and interaction for the United States WCPO purse seine fishery between 2008 and 2018 are at only approximately 5°N. The main effort of the fishery does not extend further than about 9°N, so it is difficult to assess whether a gap in distribution may occur between the equatorial area observed by the fishery and the main breeding range of the Western North Pacific humpbacks farther north. It would be expected that Western North Pacific humpback whales would be present near the equator mainly during northern winter (austral summer), and East Australia and Oceania humpback whales would be present near the equator mainly during austral winter (northern summer).

An examination of humpback whale sightings and interactions in the United States WCPO purse seine fishery indicates that humpback whales were present in the *Action Area* most of the year, though no sightings or interactions occurred in February, October, or November. Sightings and/or interactions above the equator occurred four times, once in May and three times in August, which is not the time of year Western North Pacific humpback whales would be expected to be in the region. These observations suggest that humpback whales from at least one of the two southern hemisphere species under consideration extend their range north of the equator, and/or the Western North Pacific humpback may extend its range South of the equator.

Alternatively, all humpback whales observed may be from the southern hemisphere species, which are not ESA-listed, given the Western North Pacific humpback whale is known to breed mainly in the Philippines, Okinawa, and Ogasawara (Bettridge et al. 2015), which are located at approximately 12-27°N. The relatively continuous observation of humpback whales through spring months does not indicate a clear transition between the expected migration timing of humpback whales in the northern and southern hemispheres, assuming humpback whales in the northern hemisphere would move north away from the equator and humpback whales in the southern hemisphere would move north toward the equator during northern spring/summer. Although there is a gap in October and November with no sightings of humpback whales during those months, all sightings and interactions in northern winter (*n* = 7) were below the equator.

Rasmussen et al. (2007) found that southern hemisphere humpback whales wintering off Central America were observed as far North as 11°N, resulting in what they referred to as “unique spatial overlap” with a northern hemisphere species. They attributed this to water temperature, and suggested that this can influence distribution, with humpback whales occurring in warm waters 21.1-28.3°C irrespective of latitude. Rasmussen et al. (2007) describe a cold tongue of water during austral winter that causes the temperature below the equator along South America to stay below 24°C, likely resulting in the migration of southern hemisphere humpback whales farther northward than the equator for warmer calving grounds. They note a similar pattern off the western coast of Africa. Rasmussen et al. (2007) did not report this phenomenon for the area in which the United States WCPO tuna purse seine fishery occurs, and Bettridge et al. (2015) does not define these species as extending beyond the equator. Calambokidis et al. (2008) reported that occasional re-sightings of humpback whales occurred between the Philippines and the Hawaiian Islands, and Ogasawara and the Hawaiian Islands. These movements are most likely indirect given distances and the need to travel to feeding grounds to maintain energy budgets.

Humpback whales can occur year-round in an area where breeding occurs. For example this phenomenon has been documented in the Gulf of California, with observations of humpback whales feeding during both summer and winter months (Urban and Aguayo 1987). Humpback whales in the Arabian Sea are year-round residents in tropical waters, likely because productivity is unusually high in the region (Mikhalev 1997). United States WCPO purse seine fishery is mainly concentrated in areas of high productivity, where convergence zones between warm, low-salinity water meets cold saline water from equatorial upwelling (Lehodey et al. 1998). Thus, East Australia and Oceania humpback whales could take advantage of high productivity, staying year-round in equatorial upwelling areas, which Lehodey et al. (1998) reported occur over several thousands of kilometers and shift zonally in correlation with the El Nino Southern Oscillation.

Humpback whales observed north of the equator in the United States WCPO purse seine fishery could be Oceania humpbacks extending a little further north, possibly related to water temperature, productivity, or other cues. Alternatively, the unknown breeding area associated with the Western North Pacific humpback could be south of Ogasawara, with individuals transiting through Ogasawara in such a way that it is possible to have the observed gap in distribution from about 5-9°N, though this scenario seems unlikely. Fishing effort is also not random, so there are limitations to using humpback sightings and interactions to determine distributions.

Overall, the relatively continuous distribution of sightings and interactions with humpback whales, the few sightings above the equator and none during northern winter, relatively high productivity that can draw humpback whales as well as tuna, and at least a small gap in sightings between 5°N and 9°N (the northern limit of the fishery) suggest these whales may all be from southern hemisphere species, most likely mainly the Oceania humpback whale because of the proximity of its breeding area to the fishing grounds. It is unlikely that the interactions recorded from 2008-2018 were from the Western North Pacific humpback whales, because of the reasons cited above and because the two winter interactions were below the equator and the one interaction above the equator occurred in summer. Hence, it is extremely unlikely that Western North Pacific, Mexico, or Central America humpback whales interact with the United States WCPO purse seine fishery. Therefore, we conclude that the United States WCPO purse seine fishery is not likely to adversely affect Western North Pacific humpback whales, Mexico humpback whales, or Central America humpback whales.

10.6.6 Eastern Pacific Scalloped Hammerhead Shark

The western-most boundary of the geographic distribution of the Eastern pacific scalloped hammerhead shark is 150°W longitude (79 FR 38213), which does not overlap with the *Action Area*. While the boundaries for the geographic distributions of the three species of scalloped hammerhead sharks in the Pacific are somewhat subjective, scalloped hammerhead sharks use coastal seas and therefore the Eastern Pacific scalloped hammerhead shark is primarily located along the Mexico, Central and South American Pacific coastlines. The eastern-most observed scalloped hammerhead shark interaction with the United States WCPO occurred at approximately 170°W, which is considerably west of the anticipated distribution of the Eastern Pacific scalloped hammerhead shark.

Eastern Pacific scalloped hammerhead sharks may be exposed to United States WCPO purse seine vessels as they transit to ports in Mexico and South America, however, as described above, stressors associated with vessels (e.g., noise, collision, and exposure to waste) are anticipated to have insignificant or discountable effects on ESA-listed species and therefore are not likely to adversely affect Eastern Pacific scalloped hammerhead sharks.

10.6.7 Guadalupe fur seal

Guadalupe fur seals are non-migratory and their breeding grounds are almost entirely on Guadalupe Island, Mexico. Small populations also exist off of Baja California on San Benito Island and off of Southern California at San Miguel Island (Carretta 2017a). However, in 2015, a sub-adult male stranded at Cerro Hermoso, Oaxaca, Mexico and is considered the southernmost documentation of the species to date (Villegas-Zurita et al. 2015). This suggests that the species can potentially be exposed to vessels from this fishery that are transiting to Mazanillo and Mazatlan, Mexico. Additionally, this species pelagic distribution is outside of the *Action Area* and the only documentations of their presence in broad ocean areas of the eastern Pacific where vessels from this fishery *may* transit, have been from incidental hookings in the operational range of the Hawaii shallow-set longline fishery (NMFS 2019b). Therefore, fishery interactions are not expected with this species. Furthermore, only three vessel transits have occurred to these ports in the history of the fishery (Table 1) and the exact vessel paths are unknown within a vast geographic expanse of ocean. Therefore, as described above, stressors associated with vessels (eg. noise, collision, and exposure to waste) are anticipated to have insignificant or discountable effects as discussed below and therefore are not likely to adversely affect Guadalupe fur seals.

11 APPENDIX B: FULL RESULTS FOR THE GIANT MANTA RAY ANALYSIS

Table 1. Results of Scenarios 1 and 2 for 10-year population projections and r = 0.019. Numbers represent the percent difference between initial abundance and abundance in year 10; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00. Annual anticipated take levels (ATL) for each subpopulation are given in parentheses.

r=0.019	**Initial Abundance**									
Population (AMTL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Mean Future Annual Interactions; Scenario 1; 10 years										
Nauru (64)	1.00	1.00	1.00	1.00	0.90	0.62	0.46	0.23	0.15	0.06
Kiribati Gilbert Is (57)	1.00	1.00	1.00	1.00	0.79	0.54	0.39	0.19	0.11	0.03
PNG East (33)	1.00	1.00	1.00	0.66	0.37	0.23	0.14	0.02	0.02	0.07
Solomon (21)	1.00	1.00	0.88	0.34	0.15	0.06	0.01	0.06	0.09	0.12
Tuvalu (39)	1.00	1.00	1.00	0.80	0.46	0.30	0.19	0.06	0.01	0.05
PNG West (12)	1.00	1.00	0.45	0.12	0.01	0.05	0.08	0.12	0.14	0.16
Pohnpei FSM (11)	1.00	0.99	0.39	0.09	0.01	0.06	0.09	0.13	0.15	0.16
Kiribati Phoenix Is East (17)	1.00	1.00	0.66	0.23	0.08	0.01	0.03	0.09	0.12	0.14
Kiribati Phoenix Is West (19)	1.00	1.00	0.77	0.28	0.12	0.04	0.01	0.08	0.10	0.13
PNG (4)	0.52	0.23	0.01	0.10	0.14	0.15	0.17	0.18	0.19	0.19
Fiji (6)	0.88	0.45	0.12	0.05	0.10	0.13	0.14	0.17	0.17	0.18
Kiribati Line Is (6)	0.70	0.34	0.06	0.07	0.12	0.14	0.15	0.17	0.18	0.19
Cook Islands (3)	0.34	0.12	0.05	0.13	0.15	0.17	0.18	0.19	0.19	0.20
New Caledonia (1)	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20	0.20
Yap FSM (1)	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20	0.20
American Samoa (4)	0.52	0.23	0.01	0.10	0.14	0.15	0.17	0.18	0.19	0.19
Palmyra (3)	0.34	0.12	0.05	0.13	0.15	0.17	0.18	0.19	0.19	0.20
Upper 95% CI Future Annual Interactions; Scenario 2; 10 years										
Nauru (69)	1.00	1.00	1.00	1.00	0.99	0.69	0.51	0.27	0.17	0.08
Kiribati Gilbert Is (61)	1.00	1.00	1.00	1.00	0.86	0.60	0.43	0.22	0.13	0.05
PNG East (35)	1.00	1.00	1.00	0.72	0.41	0.25	0.16	0.04	0.01	0.06
Solomon Islands (23)	1.00	1.00	0.99	0.39	0.19	0.09	0.03	0.05	0.08	0.11
Tuvalu (40)	1.00	1.00	1.00	0.83	0.48	0.31	0.21	0.07	0.01	0.04
PNG West (12)	1.00	1.00	0.45	0.12	0.01	0.05	0.08	0.12	0.14	0.16
Pohnpei FSM (13)	1.00	1.00	0.45	0.12	0.01	0.05	0.08	0.12	0.14	0.16

r=0.019	**Initial Abundance**									
Population (AMTL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Kiribati Phoenix Is East (18)	1.00	1.00	0.72	0.25	0.10	0.02	0.02	0.09	0.11	0.13
Kiribati Phoenix Is West (21)	1.00	1.00	0.88	0.34	0.15	0.06	0.01	0.06	0.09	0.12
PNG (4)	0.52	0.23	0.01	0.10	0.14	0.15	0.17	0.18	0.19	0.19
Fiji (6)	0.88	0.45	0.12	0.05	0.10	0.13	0.14	0.17	0.17	0.18
Kiribati Line Is (6)	0.88	0.45	0.12	0.05	0.10	0.13	0.14	0.17	0.17	0.18
Cook Islands (3)	0.34	0.12	0.05	0.13	0.15	0.17	0.18	0.19	0.19	0.20
New Caledonia (1)	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20	0.20
Yap FSM (1)	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20	0.20
American Samoa (4)	0.52	0.23	0.01	0.10	0.14	0.15	0.17	0.18	0.19	0.19
Palmyra (3)	0.34	0.12	0.05	0.13	0.15	0.17	0.18	0.19	0.19	0.20

Table 2. Results of Scenarios 1 and 2 for 20-year population projections and r = 0.019. Numbers represent the percent difference between initial abundance and abundance in year 20; red numbers represent population and black numbers represent population increases. Total extirpation is represented by 1.00. Annual anticipated take levels (ATL) for each subpopulation are given in parentheses.

r=0.019	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Mean Future Annual Interactions; Scenario 1; 20 years										
Nauru (64)	1.00	1.00	1.00	1.00	1.00	1.00	1.00	0.52	0.32	0.13
Kiribati Gilbert Is (57)	1.00	1.00	1.00	1.00	1.00	1.00	0.86	0.42	0.24	0.08
PNG East (33)	1.00	1.00	1.00	1.00	0.82	0.50	0.31	0.05	0.05	0.15
Solomon (21)	1.00	1.00	1.00	0.74	0.34	0.14	0.02	0.14	0.21	0.27
Tuvalu (39)	1.00	1.00	1.00	1.00	1.00	0.65	0.43	0.13	0.01	0.10
PNG West (12)	1.00	1.00	0.98	0.26	0.02	0.10	0.17	0.27	0.31	0.34
Pohnpei FSM (11)	1.00	1.00	0.86	0.20	0.02	0.13	0.20	0.29	0.32	0.35
Kiribati Phoenix Is East (17)	1.00	1.00	1.00	0.50	0.18	0.02	0.08	0.21	0.26	0.31
Kiribati Phoenix Is West (19)	1.00	1.00	1.00	0.62	0.26	0.08	0.03	0.17	0.23	0.29
PNG (4)	1.00	0.50	0.02	0.22	0.30	0.34	0.37	0.40	0.41	0.42
Fiji (6)	1.00	0.98	0.26	0.10	0.22	0.28	0.32	0.37	0.39	0.40
Kiribati Line Is (6)	1.00	0.74	0.14	0.16	0.26	0.31	0.34	0.38	0.40	0.41
Cook Islands (3)	0.74	0.26	0.10	0.28	0.34	0.37	0.39	0.41	0.42	0.43
New Caledonia (1)	0.06	0.22	0.34	0.40	0.42	0.43	0.44	0.45	0.45	0.45
Yap FSM (1)	0.06	0.22	0.34	0.40	0.42	0.43	0.44	0.45	0.45	0.45
American Samoa (4)	1.00	0.50	0.02	0.22	0.30	0.34	0.37	0.40	0.41	0.42
Palmyra (3)	0.74	0.26	0.10	0.28	0.34	0.37	0.39	0.41	0.42	0.43
Upper 95% CI Future Annual Interactions; Scenario 2; 20 years										
Nauru (69)	1.00	1.00	1.00	1.00	1.00	1.00	1.00	0.60	0.39	0.18
Kiribati Gilbert Is (61)	1.00	1.00	1.00	1.00	1.00	1.00	0.96	0.49	0.30	0.11
PNG East (35)	1.00	1.00	1.00	1.00	0.90	0.56	0.36	0.08	0.03	0.13
Solomon Islands (23)	1.00	1.00	1.00	0.86	0.42	0.20	0.07	0.11	0.18	0.25
Tuvalu (40)	1.00	1.00	1.00	1.00	1.00	0.68	0.45	0.15	0.03	0.09
PNG West (12)	1.00	1.00	0.98	0.26	0.02	0.10	0.17	0.27	0.31	0.34
Pohnpei FSM (13)	1.00	1.00	0.98	0.26	0.02	0.10	0.17	0.27	0.31	0.34
Kiribati Phoenix Is East (18)	1.00	1.00	1.00	0.56	0.22	0.05	0.05	0.19	0.24	0.30

r=0.019	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Kiribati Phoenix Is West (21)	1.00	1.00	1.00	0.74	0.34	0.14	0.02	0.14	0.21	0.27
PNG (4)	1.00	0.50	0.02	0.22	0.30	0.34	0.37	0.40	0.41	0.42
Fiji (6)	1.00	0.98	0.26	0.10	0.22	0.28	0.32	0.37	0.39	0.40
Kiribati Line Is (6)	1.00	0.98	0.26	0.10	0.22	0.28	0.32	0.37	0.39	0.40
Cook Islands (3)	0.74	0.26	0.10	0.28	0.34	0.37	0.39	0.41	0.42	0.43
New Caledonia (1)	0.06	0.22	0.34	0.40	0.42	0.43	0.44	0.45	0.45	0.45
Yap FSM (1)	0.06	0.22	0.34	0.40	0.42	0.43	0.44	0.45	0.45	0.45
American Samoa (4)	1.00	0.50	0.02	0.22	0.30	0.34	0.37	0.40	0.41	0.42
Palmyra (3)	0.74	0.26	0.10	0.28	0.34	0.37	0.39	0.41	0.42	0.43

Table 3. Results of Scenarios 1 and 2 for 40-year population projections and r = 0.019. Numbers represent the percent difference between initial abundance and abundance in year 40; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00. Annual anticipated take levels (ATL) for each subpopulation are given in parentheses.

r=0.019	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Mean Future Annual Interactions; Scenario 1; 40 years										
Nauru (64)	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	0.79	0.33
Kiribati Gilbert Is (57)	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	0.60	0.19
PNG East (33)	1.00	1.00	1.00	1.00	1.00	1.00	0.76	0.13	0.13	0.37
Solomon (21)	1.00	1.00	1.00	1.00	0.84	0.35	0.05	0.35	0.51	0.66
Tuvalu (39)	1.00	1.00	1.00	1.00	1.00	1.00	1.00	0.33	0.03	0.25
PNG West (12)	1.00	1.00	1.00	0.64	0.05	0.25	0.43	0.66	0.76	0.85
Pohnpei FSM (11)	1.00	1.00	1.00	0.49	0.05	0.32	0.49	0.70	0.79	0.87
Kiribati Phoenix Is East (17)	1.00	1.00	1.00	1.00	0.44	0.05	0.19	0.51	0.63	0.75
Kiribati Phoenix Is West (19)	1.00	1.00	1.00	1.00	0.64	0.20	0.07	0.43	0.57	0.70
PNG (4)	1.00	1.00	0.05	0.54	0.74	0.84	0.90	0.98	1.01	1.04
Fiji (6)	1.00	1.00	0.64	0.25	0.54	0.69	0.78	0.90	0.95	0.99
Kiribati Line Is (6)	1.00	1.00	0.35	0.40	0.64	0.77	0.84	0.94	0.98	1.02
Cook Islands (3)	1.00	0.64	0.25	0.69	0.84	0.92	0.96	1.02	1.04	1.07
New Caledonia (1)	0.15	0.54	0.84	0.99	1.04	1.06	1.08	1.10	1.11	1.11
Yap FSM (1)	0.15	0.54	0.84	0.99	1.04	1.06	1.08	1.10	1.11	1.11
American Samoa (4)	1.00	1.00	0.05	0.54	0.74	0.84	0.90	0.98	1.01	1.04
Palmyra (3)	1.00	0.64	0.25	0.69	0.84	0.92	0.96	1.02	1.04	1.07
Upper 95% CI Future Annual Interactions; Scenario 2; 40 years										
Nauru (69)	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	0.95	0.45
Kiribati Gilbert Is (61)	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	0.73	0.28
PNG East (35)	1.00	1.00	1.00	1.00	1.00	1.00	0.88	0.21	0.06	0.32
Solomon Islands (23)	1.00	1.00	1.00	1.00	1.00	0.49	0.17	0.27	0.44	0.61
Tuvalu (40)	1.00	1.00	1.00	1.00	1.00	1.00	1.00	0.37	0.06	0.22
PNG West (12)	1.00	1.00	1.00	0.64	0.05	0.25	0.43	0.66	0.76	0.85
Pohnpei FSM (13)	1.00	1.00	1.00	0.64	0.05	0.25	0.43	0.66	0.76	0.85
Kiribati Phoenix Is East (18)	1.00	1.00	1.00	1.00	0.54	0.12	0.13	0.47	0.60	0.73

r=0.019	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Kiribati Phoenix Is West (21)	1.00	1.00	1.00	1.00	0.84	0.35	0.05	0.35	0.51	0.66
PNG (4)	1.00	1.00	0.05	0.54	0.74	0.84	0.90	0.98	1.01	1.04
Fiji (6)	1.00	1.00	0.64	0.25	0.54	0.69	0.78	0.90	0.95	0.99
Kiribati Line Is (6)	1.00	1.00	0.64	0.25	0.54	0.69	0.78	0.90	0.95	0.99
Cook Islands (3)	1.00	0.64	0.25	0.69	0.84	0.92	0.96	1.02	1.04	1.07
New Caledonia (1)	0.15	0.54	0.84	0.99	1.04	1.06	1.08	1.10	1.11	1.11
Yap FSM (1)	0.15	0.54	0.84	0.99	1.04	1.06	1.08	1.10	1.11	1.11
American Samoa (4)	1.00	1.00	0.05	0.54	0.74	0.84	0.90	0.98	1.01	1.04
Palmyra (3)	1.00	0.64	0.25	0.69	0.84	0.92	0.96	1.02	1.04	1.07

Table 4. Results of Scenarios 1 and 2 for 10-year population projections and r = 0.042. Numbers represent the percent difference between initial abundance and abundance in year 10; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00. Annual anticipated take levels (ATL) for each subpopulation are given in parentheses.

r=0.042	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Mean Future Annual Interactions; Scenario 1; 10 years										
Nauru (64)	1.00	1.00	1.00	1.00	0.72	0.41	0.22	0.03	0.13	0.22
Kiribati Gilbert Is (57)	1.00	1.00	1.00	1.00	0.59	0.31	0.15	0.08	0.17	0.25
PNG East (33)	1.00	1.00	1.00	0.45	0.13	0.04	0.13	0.26	0.31	0.36
Solomon (21)	1.00	1.00	0.69	0.09	0.12	0.22	0.28	0.36	0.39	0.42
Tuvalu (39)	1.00	1.00	1.00	0.60	0.23	0.04	0.07	0.22	0.28	0.34
PNG West (12)	1.00	0.94	0.21	0.16	0.28	0.34	0.38	0.42	0.44	0.46
Pohnpei FSM (11)	1.00	0.82	0.15	0.19	0.30	0.35	0.39	0.43	0.45	0.47
Kiribati Phoenix Is East (17)	1.00	1.00	0.45	0.04	0.20	0.28	0.33	0.39	0.42	0.44
Kiribati Phoenix Is West (19)	1.00	1.00	0.57	0.03	0.16	0.25	0.30	0.38	0.41	0.43
PNG (4)	0.29	0.04	0.28	0.40	0.44	0.46	0.47	0.49	0.50	0.50
Fiji (6)	0.69	0.21	0.16	0.34	0.40	0.43	0.45	0.47	0.48	0.49
Kiribati Line Is (6)	0.49	0.09	0.22	0.37	0.42	0.45	0.46	0.48	0.49	0.50
Cook Islands (3)	0.09	0.16	0.34	0.43	0.46	0.48	0.49	0.50	0.50	0.51
New Caledonia (1)	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52	0.52
Yap FSM (1)	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52	0.52
American Samoa (4)	0.29	0.04	0.28	0.40	0.44	0.46	0.47	0.49	0.50	0.50
Palmyra (3)	0.09	0.16	0.34	0.43	0.46	0.48	0.49	0.50	0.50	0.51
Upper 95% CI Future Annual Interactions; Scenario 2; 10 years										
Nauru (69)	1.00	1.00	1.00	1.00	0.82	0.48	0.28	0.01	0.09	0.20
Kiribati Gilbert Is (61)	1.00	1.00	1.00	1.00	0.67	0.38	0.20	0.04	0.14	0.23
PNG East (35)	1.00	1.00	1.00	0.51	0.17	0.00	0.11	0.25	0.30	0.35
Solomon Islands (23)	1.00	1.00	0.82	0.15	0.08	0.19	0.25	0.34	0.38	0.41
Tuvalu (40)	1.00	1.00	1.00	0.63	0.25	0.06	0.06	0.21	0.28	0.33
PNG West (12)	1.00	0.94	0.21	0.16	0.28	0.34	0.38	0.42	0.44	0.46
Pohnpei FSM (13)	1.00	0.94	0.21	0.16	0.28	0.34	0.38	0.42	0.44	0.46
Kiribati Phoenix Is East (18)	1.00	1.00	0.51	0.00	0.18	0.26	0.32	0.38	0.41	0.44

r=0.042	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Kiribati Phoenix Is West (21)	1.00	1.00	0.69	0.09	0.12	0.22	0.28	0.36	0.39	0.42
PNG (4)	0.29	0.04	0.28	0.40	0.44	0.46	0.47	0.49	0.50	0.50
Fiji (6)	0.69	0.21	0.16	0.34	0.40	0.43	0.45	0.47	0.48	0.49
Kiribati Line Is (6)	0.69	0.21	0.16	0.34	0.40	0.43	0.45	0.47	0.48	0.49
Cook Islands (3)	0.09	0.16	0.34	0.43	0.46	0.48	0.49	0.50	0.50	0.51
New Caledonia (1)	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52	0.52
Yap FSM (1)	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52	0.52
American Samoa (4)	0.29	0.04	0.28	0.40	0.44	0.46	0.47	0.49	0.50	0.50
Palmyra (3)	0.09	0.16	0.34	0.43	0.46	0.48	0.49	0.50	0.50	0.51

Table 5. Results of Scenarios 1 and 2 for 20-year population projections and r = 0.042. Numbers represent the percent difference between initial abundance and abundance in year 20; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00. Annual anticipated take levels (ATL) for each subpopulation are given in parentheses.

r=0.042	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Mean Future Annual Interactions; Scenario 1; 20 years										
Nauru (64)	1.00	1.00	1.00	1.00	1.00	1.00	0.56	0.07	0.32	0.56
Kiribati Gilbert Is (57)	1.00	1.00	1.00	1.00	1.00	0.79	0.37	0.19	0.42	0.63
PNG East (33)	1.00	1.00	1.00	1.00	0.32	0.09	0.33	0.66	0.79	0.92
Solomon (21)	1.00	1.00	1.00	0.22	0.29	0.55	0.70	0.91	0.99	1.07
Tuvalu (39)	1.00	1.00	1.00	1.00	0.58	0.10	0.18	0.56	0.71	0.86
PNG West (12)	1.00	1.00	0.52	0.40	0.70	0.86	0.95	1.07	1.12	1.17
Pohnpei FSM (11)	1.00	1.00	0.37	0.47	0.75	0.89	0.98	1.09	1.14	1.18
Kiribati Phoenix Is East (17)	1.00	1.00	1.00	0.09	0.50	0.70	0.83	0.99	1.05	1.12
Kiribati Phoenix Is West (19)	1.00	1.00	1.00	0.06	0.40	0.63	0.76	0.95	1.02	1.09
PNG (4)	0.73	0.09	0.70	1.01	1.11	1.16	1.19	1.23	1.25	1.27
Fiji (6)	1.00	0.52	0.40	0.86	1.01	1.09	1.13	1.19	1.22	1.24
Kiribati Line Is (6)	1.00	0.22	0.55	0.93	1.06	1.12	1.16	1.21	1.23	1.25
Cook Islands (3)	0.22	0.40	0.86	1.09	1.16	1.20	1.22	1.25	1.27	1.28
New Caledonia (1)	0.80	1.01	1.16	1.24	1.27	1.28	1.29	1.30	1.30	1.30
Yap FSM (1)	0.80	1.01	1.16	1.24	1.27	1.28	1.29	1.30	1.30	1.30
American Samoa (4)	0.73	0.09	0.70	1.01	1.11	1.16	1.19	1.23	1.25	1.27
Palmyra (3)	0.22	0.40	0.86	1.09	1.16	1.20	1.22	1.25	1.27	1.28
Upper 95% CI Future Annual Interactions; Scenario 2; 20 years										
Nauru (69)	1.00	1.00	1.00	1.00	1.00	1.00	0.71	0.03	0.24	0.49
Kiribati Gilbert Is (61)	1.00	1.00	1.00	1.00	1.00	0.95	0.49	0.11	0.35	0.58
PNG East (35)	1.00	1.00	1.00	1.00	0.42	0.01	0.27	0.62	0.76	0.89
Solomon Islands (23)	1.00	1.00	1.00	0.37	0.19	0.47	0.64	0.87	0.96	1.04
Tuvalu (40)	1.00	1.00	1.00	1.00	0.63	0.14	0.15	0.54	0.69	0.84
PNG West (12)	1.00	1.00	0.52	0.40	0.70	0.86	0.95	1.07	1.12	1.17
Pohnpei FSM (13)	1.00	1.00	0.52	0.40	0.70	0.86	0.95	1.07	1.12	1.17
Kiribati Phoenix Is East (18)	1.00	1.00	1.00	0.01	0.45	0.66	0.79	0.97	1.04	1.10

r=0.042	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Kiribati Phoenix Is West (21)	1.00	1.00	1.00	0.22	0.29	0.55	0.70	0.91	0.99	1.07
PNG (4)	0.73	0.09	0.70	1.01	1.11	1.16	1.19	1.23	1.25	1.27
Fiji (6)	1.00	0.52	0.40	0.86	1.01	1.09	1.13	1.19	1.22	1.24
Kiribati Line Is (6)	1.00	0.52	0.40	0.86	1.01	1.09	1.13	1.19	1.22	1.24
Cook Islands (3)	0.22	0.40	0.86	1.09	1.16	1.20	1.22	1.25	1.27	1.28
New Caledonia (1)	0.80	1.01	1.16	1.24	1.27	1.28	1.29	1.30	1.30	1.30
Yap FSM (1)	0.80	1.01	1.16	1.24	1.27	1.28	1.29	1.30	1.30	1.30
American Samoa (4)	0.73	0.09	0.70	1.01	1.11	1.16	1.19	1.23	1.25	1.27
Palmyra (3)	0.22	0.40	0.86	1.09	1.16	1.20	1.22	1.25	1.27	1.28

Table 6. Results of Scenarios 1 and 2 for 40-year population projections and r = 0.042. Numbers represent the percent difference between initial abundance and abundance in year 40; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00. Annual anticipated take levels (ATL) for each subpopulation are given in parentheses.

r=0.042	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Mean Future Annual Interactions; Scenario 1; 40 years										
Nauru (64)	1.00	1.00	1.00	1.00	1.00	1.00	1.00	0.23	1.05	1.85
Kiribati Gilbert Is (57)	1.00	1.00	1.00	1.00	1.00	1.00	1.00	0.63	1.38	2.09
PNG East (33)	1.00	1.00	1.00	1.00	1.00	0.29	1.11	2.19	2.63	3.04
Solomon (21)	1.00	1.00	1.00	0.72	0.97	1.82	2.33	3.01	3.28	3.54
Tuvalu (39)	1.00	1.00	1.00	1.00	1.00	0.34	0.60	1.86	2.36	2.84
PNG West (12)	1.00	1.00	1.00	1.31	2.33	2.84	3.14	3.55	3.71	3.87
Pohnpei FSM (11)	1.00	1.00	1.00	1.57	2.50	2.97	3.25	3.62	3.77	3.91
Kiribati Phoenix Is East (17)	1.00	1.00	1.00	0.29	1.65	2.33	2.74	3.28	3.50	3.70
Kiribati Phoenix Is West (19)	1.00	1.00	1.00	0.21	1.31	2.08	2.53	3.14	3.39	3.62
PNG (4)	1.00	0.29	2.33	3.35	3.69	3.86	3.96	4.09	4.15	4.20
Fiji (6)	1.00	1.00	1.31	2.84	3.35	3.60	3.75	3.96	4.04	4.12
Kiribati Line Is (6)	1.00	0.72	1.82	3.09	3.52	3.73	3.86	4.03	4.09	4.16
Cook Islands (3)	0.72	1.31	2.84	3.60	3.86	3.98	4.06	4.16	4.20	4.24
New Caledonia (1)	2.67	3.35	3.86	4.11	4.20	4.24	4.26	4.30	4.31	4.32
Yap FSM (1)	2.67	3.35	3.86	4.11	4.20	4.24	4.26	4.30	4.31	4.32
American Samoa (4)	1.00	0.29	2.33	3.35	3.69	3.86	3.96	4.09	4.15	4.20
Palmyra (3)	0.72	1.31	2.84	3.60	3.86	3.98	4.06	4.16	4.20	4.24
Upper 95% CI Future Annual Interactions; Scenario 2; 40 years										
Nauru (69)	1.00	1.00	1.00	1.00	1.00	1.00	1.00	0.11	0.78	1.64
Kiribati Gilbert Is (61)	1.00	1.00	1.00	1.00	1.00	1.00	1.00	0.36	1.16	1.93
PNG East (35)	1.00	1.00	1.00	1.00	1.00	0.04	0.91	2.06	2.52	2.96
Solomon Islands (23)	1.00	1.00	1.00	1.00	0.63	1.57	2.13	2.87	3.17	3.46
Tuvalu (40)	1.00	1.00	1.00	1.00	1.00	0.47	0.50	1.79	2.30	2.80
PNG West (12)	1.00	1.00	1.00	1.31	2.33	2.84	3.14	3.55	3.71	3.87
Pohnpei FSM (13)	1.00	1.00	1.00	1.31	2.33	2.84	3.14	3.55	3.71	3.87
Kiribati Phoenix Is East (18)	1.00	1.00	1.00	0.04	1.48	2.20	2.64	3.21	3.44	3.66

r=0.042	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Kiribati Phoenix Is West (21)	1.00	1.00	1.00	0.72	0.97	1.82	2.33	3.01	3.28	3.54
PNG (4)	1.00	0.29	2.33	3.35	3.69	3.86	3.96	4.09	4.15	4.20
Fiji (6)	1.00	1.00	1.31	2.84	3.35	3.60	3.75	3.96	4.04	4.12
Kiribati Line Is (6)	1.00	1.00	1.31	2.84	3.35	3.60	3.75	3.96	4.04	4.12
Cook Islands (3)	0.72	1.31	2.84	3.60	3.86	3.98	4.06	4.16	4.20	4.24
New Caledonia (1)	2.67	3.35	3.86	4.11	4.20	4.24	4.26	4.30	4.31	4.32
Yap FSM (1)	2.67	3.35	3.86	4.11	4.20	4.24	4.26	4.30	4.31	4.32
American Samoa (4)	1.00	0.29	2.33	3.35	3.69	3.86	3.96	4.09	4.15	4.20
Palmyra (3)	0.72	1.31	2.84	3.60	3.86	3.98	4.06	4.16	4.20	4.24

Table 7. Results of Scenarios 1 and 2 for 10-year population projections and r = 0.116. Numbers represent the percent difference between initial abundance and abundance in year 10; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00. Annual anticipated take levels (ATL) for each subpopulation are given in parentheses.

r=0.116	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Mean Future Annual Interactions; Scenario 1; 10 years										
Nauru (64)	1.00	1.00	1.00	0.53	0.38	0.83	1.10	1.47	1.61	1.75
Kiribati Gilbert Is (57)	1.00	1.00	1.00	0.26	0.56	0.97	1.21	1.54	1.67	1.79
PNG East (33)	1.00	1.00	0.66	0.77	1.24	1.48	1.62	1.81	1.89	1.96
Solomon (21)	1.00	1.00	0.41	1.30	1.60	1.74	1.83	1.95	2.00	2.05
Tuvalu (39)	1.00	1.00	1.00	0.54	1.09	1.37	1.53	1.75	1.84	1.92
PNG West (12)	1.00	0.05	1.12	1.66	1.83	1.92	1.98	2.05	2.08	2.10
Pohnpei FSM (11)	1.00	0.23	1.21	1.70	1.86	1.95	1.99	2.06	2.09	2.11
Kiribati Phoenix Is East (17)	1.00	0.66	0.77	1.48	1.72	1.83	1.91	2.00	2.04	2.07
Kiribati Phoenix Is West (19)	1.00	1.00	0.59	1.39	1.66	1.79	1.87	1.98	2.02	2.06
PNG (4)	1.00	1.48	1.83	2.01	2.07	2.10	2.12	2.14	2.15	2.16
Fiji (6)	0.41	1.12	1.66	1.92	2.01	2.06	2.08	2.12	2.13	2.15
Kiribati Line Is (6)	0.71	1.30	1.74	1.97	2.04	2.08	2.10	2.13	2.14	2.15
Cook Islands (3)	1.30	1.66	1.92	2.06	2.10	2.12	2.14	2.15	2.16	2.17
New Caledonia (1)	1.89	2.01	2.10	2.15	2.16	2.17	2.17	2.18	2.18	2.18
Yap FSM (1)	1.89	2.01	2.10	2.15	2.16	2.17	2.17	2.18	2.18	2.18
American Samoa (4)	1.00	1.48	1.83	2.01	2.07	2.10	2.12	2.14	2.15	2.16
Palmyra (3)	1.30	1.66	1.92	2.06	2.10	2.12	2.14	2.15	2.16	2.17
Upper 95% CI Future Annual Interactions; Scenario 2; 10 years										
Nauru (69)	1.00	1.00	1.00	0.75	0.23	0.72	1.01	1.41	1.56	1.71
Kiribati Gilbert Is (61)	1.00	1.00	1.00	0.44	0.44	0.88	1.14	1.49	1.63	1.76
PNG East (35)	1.00	1.00	0.84	0.68	1.18	1.43	1.58	1.79	1.87	1.94
Solomon Islands (23)	1.00	1.00	0.23	1.21	1.54	1.70	1.80	1.93	1.98	2.03
Tuvalu (40)	1.00	1.00	1.00	0.50	1.06	1.34	1.51	1.74	1.83	1.92
PNG West (12)	1.00	0.05	1.12	1.66	1.83	1.92	1.98	2.05	2.08	2.10
Pohnpei FSM (13)	1.00	0.05	1.12	1.66	1.83	1.92	1.98	2.05	2.08	2.10
Kiribati Phoenix Is East (18)	1.00	0.84	0.68	1.43	1.69	1.81	1.89	1.99	2.03	2.07

r=0.116	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Kiribati Phoenix Is West (21)	1.00	1.00	0.41	1.30	1.60	1.74	1.83	1.95	2.00	2.05
PNG (4)	1.00	1.48	1.83	2.01	2.07	2.10	2.12	2.14	2.15	2.16
Fiji (6)	0.41	1.12	1.66	1.92	2.01	2.06	2.08	2.12	2.13	2.15
Kiribati Line Is (6)	0.41	1.12	1.66	1.92	2.01	2.06	2.08	2.12	2.13	2.15
Cook Islands (3)	1.30	1.66	1.92	2.06	2.10	2.12	2.14	2.15	2.16	2.17
New Caledonia (1)	1.89	2.01	2.10	2.15	2.16	2.17	2.17	2.18	2.18	2.18
Yap FSM (1)	1.89	2.01	2.10	2.15	2.16	2.17	2.17	2.18	2.18	2.18
American Samoa (4)	1.00	1.48	1.83	2.01	2.07	2.10	2.12	2.14	2.15	2.16
Palmyra (3)	1.30	1.66	1.92	2.06	2.10	2.12	2.14	2.15	2.16	2.17

Table 8. Results of Scenarios 1 and 2 for 20-year population projections and r = 0.116. Numbers represent the percent difference between initial abundance and abundance in year 20; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00. Annual anticipated take levels (ATL) for each subpopulation are given in parentheses.

r=0.116	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Mean Future Annual Interactions; Scenario 1; 20 years										
Nauru (64)	1.00	1.00	1.00	1.00	1.59	3.49	4.62	6.14	6.75	7.33
Kiribati Gilbert Is (57)	1.00	1.00	1.00	1.00	2.34	4.05	5.07	6.44	6.99	7.51
PNG East (33)	1.00	1.00	1.00	3.21	5.20	6.19	6.79	7.58	7.90	8.21
Solomon (21)	1.00	1.00	1.72	5.45	6.69	7.31	7.68	8.18	8.38	8.57
Tuvalu (39)	1.00	1.00	1.00	2.28	4.58	5.73	6.42	7.34	7.70	8.06
PNG West (12)	1.00	0.22	4.70	6.94	7.68	8.06	8.28	8.58	8.70	8.81
Pohnpei FSM (11)	1.00	0.97	5.07	7.12	7.81	8.15	8.36	8.63	8.74	8.84
Kiribati Phoenix Is East (17)	1.00	1.00	3.21	6.19	7.19	7.68	7.98	8.38	8.54	8.69
Kiribati Phoenix Is West (19)	1.00	1.00	2.46	5.82	6.94	7.50	7.83	8.28	8.46	8.63
PNG (4)	4.20	6.19	7.68	8.43	8.68	8.80	8.88	8.98	9.02	9.05
Fiji (6)	1.72	4.70	6.94	8.06	8.43	8.62	8.73	8.88	8.94	8.99
Kiribati Line Is (6)	2.96	5.45	7.31	8.24	8.55	8.71	8.80	8.93	8.98	9.02
Cook Islands (3)	5.45	6.94	8.06	8.62	8.80	8.90	8.95	9.03	9.06	9.08
New Caledonia (1)	7.93	8.43	8.80	8.99	9.05	9.08	9.10	9.13	9.14	9.15
Yap FSM (1)	7.93	8.43	8.80	8.99	9.05	9.08	9.10	9.13	9.14	9.15
American Samoa (4)	4.20	6.19	7.68	8.43	8.68	8.80	8.88	8.98	9.02	9.05
Palmyra (3)	5.45	6.94	8.06	8.62	8.80	8.90	8.95	9.03	9.06	9.08
Upper 95% CI Future Annual Interactions; Scenario 2; 20 years										
Nauru (69)	1.00	1.00	1.00	1.00	0.97	3.02	4.25	5.89	6.55	7.18
Kiribati Gilbert Is (61)	1.00	1.00	1.00	1.00	1.84	3.67	4.77	6.24	6.83	7.39
PNG East (35)	1.00	1.00	1.00	2.83	4.95	6.01	6.64	7.48	7.82	8.15
Solomon Islands (23)	1.00	1.00	0.97	5.07	6.44	7.12	7.53	8.08	8.30	8.51
Tuvalu (40)	1.00	1.00	1.00	2.09	4.45	5.63	6.34	7.29	7.66	8.03
PNG West (12)	1.00	0.22	4.70	6.94	7.68	8.06	8.28	8.58	8.70	8.81
Pohnpei FSM (13)	1.00	0.22	4.70	6.94	7.68	8.06	8.28	8.58	8.70	8.81
Kiribati Phoenix Is East (18)	1.00	1.00	2.83	6.01	7.06	7.59	7.91	8.33	8.50	8.66

r=0.116	Initial Abundance									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Kiribati Phoenix Is West (21)	1.00	1.00	1.72	5.45	6.69	7.31	7.68	8.18	8.38	8.57
PNG (4)	4.20	6.19	7.68	8.43	8.68	8.80	8.88	8.98	9.02	9.05
Fiji (6)	1.72	4.70	6.94	8.06	8.43	8.62	8.73	8.88	8.94	8.99
Kiribati Line Is (6)	1.72	4.70	6.94	8.06	8.43	8.62	8.73	8.88	8.94	8.99
Cook Islands (3)	5.45	6.94	8.06	8.62	8.80	8.90	8.95	9.03	9.06	9.08
New Caledonia (1)	7.93	8.43	8.80	8.99	9.05	9.08	9.10	9.13	9.14	9.15
Yap FSM (1)	7.93	8.43	8.80	8.99	9.05	9.08	9.10	9.13	9.14	9.15
American Samoa (4)	4.20	6.19	7.68	8.43	8.68	8.80	8.88	8.98	9.02	9.05
Palmyra (3)	5.45	6.94	8.06	8.62	8.80	8.90	8.95	9.03	9.06	9.08

Table 9. Results of Scenarios 1 and 2 for 40-year population projections and r = 0.116. Numbers represent the percent difference between initial abundance and abundance in year 40; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00. Annual anticipated take levels (ATL) for each subpopulation are given in parentheses.

r=0.116	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Mean Future Annual Interactions; Scenario 1; 40 years										
Nauru (64)	1.00	1.00	1.00	1.00	17.78	38.97	51.69	68.64	75.42	81.90
Kiribati Gilbert Is (57)	1.00	1.00	1.00	1.00	26.12	45.23	56.69	71.97	78.09	83.93
PNG East (33)	1.00	1.00	1.00	35.85	58.08	69.20	75.87	84.76	88.32	91.72
Solomon (21)	1.00	1.00	19.17	60.86	74.75	81.70	85.87	91.43	93.65	95.78
Tuvalu (39)	1.00	1.00	1.00	25.43	51.13	63.98	71.70	81.98	86.09	90.02
PNG West (12)	1.00	2.50	52.52	77.53	85.87	90.04	92.54	95.87	97.21	98.48
Pohnpei FSM (11)	1.00	10.83	56.69	79.62	87.26	91.08	93.37	96.43	97.65	98.82
Kiribati Phoenix Is East (17)	1.00	1.00	35.85	69.20	80.31	85.87	89.20	93.65	95.43	97.13
Kiribati Phoenix Is West (19)	1.00	1.00	27.51	65.03	77.53	83.79	87.54	92.54	94.54	96.45
PNG (4)	46.96	69.20	85.87	94.21	96.99	98.38	99.21	100.32	100.77	101.19
Fiji (6)	19.17	52.52	77.53	90.04	94.21	96.29	97.54	99.21	99.88	100.51
Kiribati Line Is (6)	33.07	60.86	81.70	92.12	95.60	97.33	98.38	99.77	100.32	100.85
Cook Islands (3)	60.86	77.53	90.04	96.29	98.38	99.42	100.04	100.88	101.21	101.53
New Caledonia (1)	88.65	94.21	98.38	100.46	101.15	101.50	101.71	101.99	102.10	102.21
Yap FSM (1)	88.65	94.21	98.38	100.46	101.15	101.50	101.71	101.99	102.10	102.21
American Samoa (4)	1.00	0.29	2.33	3.35	3.69	3.86	3.96	4.09	4.15	4.20
Palmyra (3)	0.72	1.31	2.84	3.60	3.86	3.98	4.06	4.16	4.20	4.24
Upper 95% CI Future Annual Interactions; Scenario 2; 40 years										
Nauru (69)	1.00	1.00	1.00	1.00	10.83	33.76	47.52	65.86	73.20	80.21
Kiribati Gilbert Is (61)	1.00	1.00	1.00	1.00	20.56	41.06	53.35	69.75	76.31	82.58
PNG East (35)	1.00	1.00	1.00	31.68	55.30	67.11	74.20	83.65	87.43	91.04
Solomon Islands (23)	1.00	1.00	10.83	56.69	71.97	79.62	84.20	90.32	92.76	95.10
Tuvalu (40)	1.00	1.00	1.00	23.34	49.74	62.94	70.86	81.42	85.65	89.69

r=0.116	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
PNG West (12)	1.00	2.50	52.52	77.53	85.87	90.04	92.54	95.87	97.21	98.48
Pohnpei FSM (13)	1.00	2.50	52.52	77.53	85.87	90.04	92.54	95.87	97.21	98.48
Kiribati Phoenix Is East (18)	1.00	1.00	31.68	67.11	78.92	84.83	88.37	93.10	94.99	96.79
Kiribati Phoenix Is West (21)	1.00	1.00	19.17	60.86	74.75	81.70	85.87	91.43	93.65	95.78
PNG (4)	46.96	69.20	85.87	94.21	96.99	98.38	99.21	100.32	100.77	101.19
Fiji (6)	19.17	52.52	77.53	90.04	94.21	96.29	97.54	99.21	99.88	100.51
Kiribati Line Is (6)	19.17	52.52	77.53	90.04	94.21	96.29	97.54	99.21	99.88	100.51
Cook Islands (3)	60.86	77.53	90.04	96.29	98.38	99.42	100.04	100.88	101.21	101.53
New Caledonia (1)	88.65	94.21	98.38	100.46	101.15	101.50	101.71	101.99	102.10	102.21
Yap FSM (1)	88.65	94.21	98.38	100.46	101.15	101.50	101.71	101.99	102.10	102.21
American Samoa (4)	1.00	0.29	2.33	3.35	3.69	3.86	3.96	4.09	4.15	4.20
Palmyra (3)	0.72	1.31	2.84	3.60	3.86	3.98	4.06	4.16	4.20	4.24

Table 10. Results of Scenarios 3a and 3b for 10-year population projections and r = 0.019 assuming 50% or 75% of giant manta ray were misidentified. Numbers represent the percent difference between initial abundance and abundance in year 10; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00. Annual anticipated take levels (ATL) for each subpopulation are given in parentheses.

r=0.019	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Mean Future Annual Interactions; 50% Reduction; Scenario 3a; 10 years										
Nauru (64)	1.00	1.00	1.00	0.64	0.35	0.21	0.13	0.02	0.03	0.07
Kiribati Gilbert Is (57)	1.00	1.00	1.00	0.55	0.30	0.17	0.10	0.01	0.05	0.09
PNG East (33)	1.00	1.00	0.66	0.23	0.08	0.01	0.03	0.09	0.12	0.14
Solomon (21)	1.00	0.99	0.39	0.09	0.01	0.06	0.09	0.13	0.15	0.16
Tuvalu (39)	1.00	1.00	0.83	0.31	0.14	0.05	0.00	0.07	0.10	0.13
PNG West (12)	0.88	0.45	0.12	0.05	0.10	0.13	0.14	0.17	0.17	0.18
Pohnpei FSM (11)	0.88	0.45	0.12	0.05	0.10	0.13	0.14	0.17	0.17	0.18
Kiribati Phoenix Is East (17)	1.00	0.77	0.28	0.04	0.05	0.09	0.11	0.14	0.16	0.17
Kiribati Phoenix Is West (19)	1.00	0.88	0.34	0.06	0.03	0.07	0.10	0.14	0.15	0.16
PNG (4)	0.15	0.01	0.10	0.15	0.17	0.18	0.19	0.19	0.20	0.20
Fiji (6)	0.34	0.12	0.05	0.13	0.15	0.17	0.18	0.19	0.19	0.20
Kiribati Line Is (6)	0.34	0.12	0.05	0.13	0.15	0.17	0.18	0.19	0.19	0.20
Cook Islands (3)	0.15	0.01	0.10	0.15	0.17	0.18	0.19	0.19	0.20	0.20
New Caledonia (1)	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20	0.20
Yap FSM (1)	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20	0.20
American Samoa (4)	0.15	0.01	0.10	0.15	0.17	0.18	0.19	0.19	0.20	0.20
Palmyra (3)	0.15	0.01	0.10	0.15	0.17	0.18	0.19	0.19	0.20	0.20
Mean Future Annual Interactions; 75% Reduction; Scenario 3b; 10 years										
Nauru (69)	1.00	1.00	0.61	0.20	0.06	0.00	0.05	0.10	0.12	0.14
Kiribati Gilbert Is (61)	1.00	1.00	0.50	0.15	0.03	0.03	0.07	0.11	0.13	0.15
PNG East (35)	1.00	0.66	0.23	0.01	0.06	0.10	0.12	0.15	0.16	0.17
Solomon Islands (23)	0.70	0.34	0.06	0.07	0.12	0.14	0.15	0.17	0.18	0.19
Tuvalu (40)	1.00	0.88	0.34	0.06	0.03	0.07	0.10	0.14	0.15	0.16
PNG West (12)	0.34	0.12	0.05	0.13	0.15	0.17	0.18	0.19	0.19	0.20
Pohnpei FSM (13)	0.34	0.12	0.05	0.13	0.15	0.17	0.18	0.19	0.19	0.20
Kiribati Phoenix Is East (18)	0.52	0.23	0.01	0.10	0.14	0.15	0.17	0.18	0.19	0.19

r=0.019	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Kiribati Phoenix Is West (21)	0.70	0.34	0.06	0.07	0.12	0.14	0.15	0.17	0.18	0.19
PNG (4)	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20	0.20
Fiji (6)	0.15	0.01	0.10	0.15	0.17	0.18	0.19	0.19	0.20	0.20
Kiribati Line Is (6)	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20	0.20
Cook Islands (3)	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20	0.20
New Caledonia (1)	0.21	0.21	0.21	0.21	0.21	0.21	0.21	0.21	0.21	0.21
Yap FSM (1)	0.21	0.21	0.21	0.21	0.21	0.21	0.21	0.21	0.21	0.21
American Samoa (4)	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20	0.20
Palmyra (3)	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20	0.20

Table 11. Results of Scenarios 3a and 3b for 10-year population projections and r = 0.042 assuming 50% and 75% of giant manta ray were misidentified. Numbers represent the percent difference between initial abundance and abundance in year 10; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00. Annual anticipated take levels (ATL) for each subpopulation are given in parentheses.

r=0.042	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Mean Future Annual Interactions; 50% Reduction; Scenario 3a; 10 years										
Nauru (64)	1.00	1.00	1.00	0.42	0.11	0.05	0.14	0.27	0.32	0.37
Kiribati Gilbert Is (57)	1.00	1.00	1.00	0.33	0.05	0.10	0.18	0.29	0.34	0.38
PNG East (33)	1.00	1.00	0.45	0.04	0.20	0.28	0.33	0.39	0.42	0.44
Solomon (21)	1.00	0.82	0.15	0.19	0.30	0.35	0.39	0.43	0.45	0.47
Tuvalu (39)	1.00	1.00	0.63	0.06	0.14	0.23	0.29	0.37	0.40	0.43
PNG West (12)	0.69	0.21	0.16	0.34	0.40	0.43	0.45	0.47	0.48	0.49
Pohnpei FSM (11)	0.69	0.21	0.16	0.34	0.40	0.43	0.45	0.47	0.48	0.49
Kiribati Phoenix Is East (17)	1.00	0.57	0.03	0.25	0.34	0.39	0.41	0.45	0.46	0.48
Kiribati Phoenix Is West (19)	1.00	0.69	0.09	0.22	0.32	0.37	0.40	0.44	0.46	0.47
PNG (4)	0.12	0.28	0.40	0.46	0.48	0.49	0.50	0.51	0.51	0.51
Fiji (6)	0.09	0.16	0.34	0.43	0.46	0.48	0.49	0.50	0.50	0.51
Kiribati Line Is (6)	0.09	0.16	0.34	0.43	0.46	0.48	0.49	0.50	0.50	0.51
Cook Islands (3)	0.12	0.28	0.40	0.46	0.48	0.49	0.50	0.51	0.51	0.51
New Caledonia (1)	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52	0.52
Yap FSM (1)	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52	0.52
American Samoa (4)	0.12	0.28	0.40	0.46	0.48	0.49	0.50	0.51	0.51	0.51
Palmyra (3)	0.12	0.28	0.40	0.46	0.48	0.49	0.50	0.51	0.51	0.51
Mean Future Annual Interactions; 75% Reduction; Scenario 3b; 10 years										
Nauru (69)	1.00	1.00	0.39	0.07	0.22	0.29	0.34	0.40	0.42	0.45
Kiribati Gilbert Is (61)	1.00	1.00	0.27	0.13	0.26	0.32	0.36	0.42	0.44	0.46
PNG East (35)	1.00	0.45	0.04	0.28	0.36	0.40	0.42	0.46	0.47	0.48
Solomon Islands (23)	0.49	0.09	0.22	0.37	0.42	0.45	0.46	0.48	0.49	0.50
Tuvalu (40)	1.00	0.69	0.09	0.22	0.32	0.37	0.40	0.44	0.46	0.47
PNG West (12)	0.09	0.16	0.34	0.43	0.46	0.48	0.49	0.50	0.50	0.51
Pohnpei FSM (13)	0.09	0.16	0.34	0.43	0.46	0.48	0.49	0.50	0.50	0.51
Kiribati Phoenix Is East (18)	0.29	0.04	0.28	0.40	0.44	0.46	0.47	0.49	0.50	0.50

r=0.042	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Kiribati Phoenix Is West (21)	0.49	0.09	0.22	0.37	0.42	0.45	0.46	0.48	0.49	0.50
PNG (4)	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52	0.52
Fiji (6)	0.12	0.28	0.40	0.46	0.48	0.49	0.50	0.51	0.51	0.51
Kiribati Line Is (6)	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52	0.52
Cook Islands (3)	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52	0.52
New Caledonia (1)	0.52	0.52	0.52	0.52	0.52	0.52	0.52	0.52	0.52	0.52
Yap FSM (1)	0.52	0.52	0.52	0.52	0.52	0.52	0.52	0.52	0.52	0.52
American Samoa (4)	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52	0.52
Palmyra (3)	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52	0.52

Table 14. Results of Scenario 4 for 10-year population projections 0% emigration, and r_{max} = 0.019. Numbers represent the percent difference between initial abundance and abundance in year 10; red numbers represent population and black numbers represent population increases. Total extirpation is represented by 1.00.

96% AV	**Initial Abundance**									
Annual Interactions	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Emigration = 0%; Immigration = 5%										
100	1.00	1.00	1.00	1.00	1.00	1.00	0.83	0.46	0.31	0.18
50	1.00	1.00	1.00	1.00	0.64	0.42	0.28	0.10	0.02	0.05
40	1.00	1.00	1.00	0.83	0.46	0.28	0.17	0.02	0.04	0.09
30	1.00	1.00	1.00	0.55	0.28	0.14	0.06	0.05	0.09	0.14
20	1.00	1.00	0.83	0.28	0.10	0.00	0.05	0.12	0.15	0.18
10	1.00	0.83	0.28	0.00	0.09	0.13	0.16	0.20	0.21	0.22
5	0.64	0.28	0.00	0.13	0.18	0.20	0.21	0.23	0.24	0.25
2	0.10	0.05	0.16	0.21	0.23	0.24	0.25	0.25	0.26	0.26
Emigration = 0%; Immigration = 10%										
100	1.00	1.00	1.00	1.00	1.00	1.00	0.81	0.43	0.28	0.14
50	1.00	1.00	1.00	1.00	0.62	0.38	0.24	0.05	0.02	0.10
40	1.00	1.00	1.00	0.81	0.43	0.24	0.13	0.02	0.08	0.14
30	1.00	1.00	1.00	0.53	0.24	0.10	0.01	0.10	0.15	0.19
20	1.00	1.00	0.81	0.24	0.05	0.04	0.10	0.18	0.21	0.24
10	1.00	0.81	0.24	0.04	0.14	0.19	0.21	0.25	0.27	0.28
5	0.62	0.24	0.04	0.19	0.23	0.26	0.27	0.29	0.30	0.30
2	0.05	0.10	0.21	0.27	0.29	0.30	0.31	0.31	0.32	0.32
Emigration = 0%; Immigration = 15%										
100	1.00	1.00	1.00	1.00	1.00	1.00	0.80	0.40	0.25	0.09
50	1.00	1.00	1.00	1.00	0.60	0.35	0.21	0.01	0.07	0.15
40	1.00	1.00	1.00	0.80	0.40	0.21	0.09	0.07	0.13	0.19
30	1.00	1.00	1.00	0.50	0.21	0.06	0.03	0.15	0.20	0.24
20	1.00	1.00	0.80	0.21	0.01	0.09	0.15	0.23	0.26	0.29
10	1.00	0.80	0.21	0.09	0.19	0.24	0.27	0.31	0.32	0.34
5	0.60	0.21	0.09	0.24	0.29	0.31	0.33	0.35	0.36	0.36
2	0.01	0.15	0.27	0.33	0.35	0.36	0.36	0.37	0.37	0.38

Table 15. Results of Scenario 4 for 10-year population projections 0% emigration, and r_{max} = 0.042. Numbers represent the percent difference between initial abundance and abundance in year 10; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00.

96% AV	**Initial Abundance**									
Annual Interactions	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Emigration = 0%; Immigration = 5%										
100	1.00	1.00	1.00	1.00	1.00	0.93	0.62	0.22	0.06	0.10
50	1.00	1.00	1.00	0.93	0.42	0.17	0.01	0.19	0.27	0.35
40	1.00	1.00	1.00	0.62	0.22	0.01	0.11	0.27	0.33	0.40
30	1.00	1.00	1.00	0.32	0.01	0.14	0.23	0.35	0.40	0.45
20	1.00	1.00	0.62	0.01	0.19	0.29	0.35	0.43	0.46	0.50
10	1.00	0.62	0.01	0.29	0.39	0.44	0.47	0.51	0.53	0.55
5	0.42	0.01	0.29	0.44	0.49	0.52	0.53	0.55	0.56	0.57
2	0.19	0.35	0.47	0.53	0.55	0.56	0.57	0.58	0.58	0.59
Emigration = 0%; Immigration = 10%										
100	1.00	1.00	1.00	1.00	1.00	0.92	0.60	0.18	0.01	0.15
50	1.00	1.00	1.00	0.92	0.39	0.13	0.03	0.24	0.33	0.41
40	1.00	1.00	1.00	0.60	0.18	0.03	0.16	0.33	0.40	0.46
30	1.00	1.00	1.00	0.29	0.03	0.19	0.29	0.41	0.46	0.51
20	1.00	1.00	0.60	0.03	0.24	0.35	0.41	0.50	0.53	0.56
10	1.00	0.60	0.03	0.35	0.46	0.51	0.54	0.58	0.60	0.62
5	0.39	0.03	0.35	0.51	0.56	0.59	0.60	0.63	0.63	0.64
2	0.24	0.41	0.54	0.60	0.63	0.64	0.64	0.65	0.65	0.66
Emigration = 0%; Immigration = 15%										
100	1.00	1.00	1.00	1.00	1.00	0.91	0.58	0.14	0.04	0.20
50	1.00	1.00	1.00	0.91	0.36	0.09	0.08	0.30	0.39	0.47
40	1.00	1.00	1.00	0.58	0.14	0.08	0.21	0.39	0.46	0.53
30	1.00	1.00	1.00	0.25	0.08	0.24	0.34	0.48	0.53	0.58
20	1.00	1.00	0.58	0.08	0.30	0.41	0.48	0.56	0.60	0.63
10	1.00	0.58	0.08	0.41	0.52	0.58	0.61	0.65	0.67	0.69
5	0.36	0.08	0.41	0.58	0.63	0.66	0.67	0.70	0.71	0.71

96% AV	**Initial Abundance**									
Annual Interactions	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
2	0.30	0.48	0.61	0.67	0.70	0.71	0.71	0.72	0.73	0.73

Table 16. Results of Scenario 4 for 10-year population projections 0% immigration, and r_{max} = 0.019. Numbers represent the percent difference between initial abundance and abundance in year 10; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00.

96% AV	**Initial Abundance**									
Annual Interactions	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Emigration = 5%; Immigration = 0%										
100	1.00	1.00	1.00	1.00	1.00	1.00	0.85	0.52	0.38	0.26
50	1.00	1.00	1.00	1.00	0.68	0.48	0.35	0.18	0.12	0.05
40	1.00	1.00	1.00	0.85	0.52	0.35	0.25	0.12	0.06	0.01
30	1.00	1.00	1.00	0.60	0.35	0.23	0.15	0.05	0.01	0.03
20	1.00	1.00	0.85	0.35	0.18	0.10	0.05	0.02	0.04	0.07
10	1.00	0.85	0.35	0.10	0.02	0.02	0.05	0.08	0.10	0.11
5	0.68	0.35	0.10	0.02	0.07	0.09	0.10	0.12	0.12	0.13
2	0.18	0.05	0.05	0.10	0.12	0.12	0.13	0.14	0.14	0.14
Emigration = 10%; Immigration = 0%										
200	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	0.93	0.68
150	1.00	1.00	1.00	1.00	1.00	1.00	1.00	0.86	0.67	0.49
100	1.00	1.00	1.00	1.00	1.00	1.00	0.86	0.55	0.42	0.30
50	1.00	1.00	1.00	1.00	0.70	0.51	0.39	0.23	0.16	0.10
40	1.00	1.00	1.00	0.86	0.55	0.39	0.29	0.16	0.11	0.06
30	1.00	1.00	1.00	0.62	0.39	0.27	0.20	0.10	0.06	0.03
20	1.00	1.00	0.86	0.39	0.23	0.15	0.10	0.04	0.01	0.01
10	1.00	0.86	0.39	0.15	0.07	0.03	0.00	0.03	0.04	0.05
5	0.70	0.39	0.15	0.03	0.01	0.03	0.04	0.06	0.07	0.07
2	0.23	0.10	0.00	0.04	0.06	0.07	0.07	0.08	0.08	0.08
Emigration = 15%; Immigration = 0%										
100	1.00	1.00	1.00	1.00	1.00	1.00	0.88	0.57	0.45	0.34
50	1.00	1.00	1.00	1.00	0.72	0.54	0.42	0.27	0.21	0.15
40	1.00	1.00	1.00	0.88	0.57	0.42	0.33	0.21	0.16	0.12
30	1.00	1.00	1.00	0.65	0.42	0.31	0.24	0.15	0.11	0.08
20	1.00	1.00	0.88	0.42	0.27	0.20	0.15	0.09	0.07	0.04
10	1.00	0.88	0.42	0.20	0.12	0.08	0.06	0.03	0.02	0.01

96% AV	**Initial Abundance**									
Annual Interactions	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
5	0.72	0.42	0.20	0.08	0.04	0.03	0.01	0.00	0.01	0.01
2	0.27	0.15	0.06	0.01	0.00	0.01	0.01	0.02	0.02	0.02

Table 17. Results of Scenario 4 for 10-year population projections 0% immigration, and r_{max} = 0.042. Numbers represent the percent difference between initial abundance and abundance in year 10; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00.

96% AV	**Initial Abundance**									
Annual Interactions	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Emigration = 5%; Immigration = 0%										
100	1.00	1.00	1.00	1.00	1.00	0.95	0.67	0.30	0.15	0.00
50	1.00	1.00	1.00	0.95	0.48	0.25	0.11	0.08	0.15	0.22
40	1.00	1.00	1.00	0.67	0.30	0.11	0.00	0.15	0.21	0.27
30	1.00	1.00	1.00	0.39	0.11	0.03	0.11	0.23	0.27	0.31
20	1.00	1.00	0.67	0.11	0.08	0.17	0.23	0.30	0.33	0.36
10	1.00	0.67	0.11	0.17	0.26	0.31	0.34	0.37	0.39	0.40
5	0.48	0.11	0.17	0.31	0.36	0.38	0.39	0.41	0.42	0.43
2	0.08	0.23	0.34	0.39	0.41	0.42	0.43	0.43	0.44	0.44
Emigration = 10%; Immigration = 0%										
100	1.00	1.00	1.00	1.00	1.00	0.96	0.69	0.33	0.19	0.06
50	1.00	1.00	1.00	0.96	0.51	0.29	0.16	0.02	0.09	0.16
40	1.00	1.00	1.00	0.69	0.33	0.16	0.05	0.09	0.15	0.20
30	1.00	1.00	1.00	0.42	0.16	0.02	0.06	0.16	0.21	0.25
20	1.00	1.00	0.69	0.16	0.02	0.11	0.16	0.23	0.26	0.29
10	1.00	0.69	0.16	0.11	0.20	0.24	0.27	0.31	0.32	0.33
5	0.51	0.16	0.11	0.24	0.29	0.31	0.32	0.34	0.35	0.35
2	0.02	0.16	0.27	0.32	0.34	0.35	0.35	0.36	0.36	0.37
Emigration = 15%; Immigration = 0%										
100	1.0000	1.0000	1.0000	1.0000	1.0000	0.9644	0.7109	0.3729	0.2377	0.1085
50	1.0000	1.0000	1.0000	0.9644	0.5419	0.3307	0.2039	0.0349	0.0327	0.0973
40	1.0000	1.0000	1.0000	0.7109	0.3729	0.2039	0.1025	0.0327	0.0867	0.1385
30	1.0000	1.0000	1.0000	0.4574	0.2039	0.0772	0.0011	0.1003	0.1408	0.1796
20	1.0000	1.0000	0.7109	0.2039	0.0349	0.0496	0.1003	0.1679	0.1949	0.2208
10	1.0000	0.7109	0.2039	0.0496	0.1341	0.1763	0.2017	0.2355	0.2490	0.2619
5	0.5419	0.2039	0.0496	0.1763	0.2186	0.2397	0.2524	0.2693	0.2760	0.2825
2	0.0349	0.1003	0.2017	0.2524	0.2693	0.2777	0.2828	0.2895	0.2922	0.2948

Table 18. Results of Scenarios 5 and 6 for 10-year population projections and r = 0.019. Numbers represent the percent difference between initial abundance and abundance in year 10; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00. Annual anticipated take levels (ATL) for each subpopulation are given in parentheses.

r=0.019	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Mean Future Annual Interactions; Scenario 5; 10 years										
Nauru (19)	1.00	1.00	0.77	0.28	0.12	0.04	0.01	0.08	0.10	0.13
Kiribati Gilbert Is (14)	1.00	1.00	0.50	0.15	0.03	0.03	0.07	0.11	0.13	0.15
PNG West (9)	1.00	0.77	0.28	0.04	0.05	0.09	0.11	0.14	0.16	0.17
Solomon (7)	1.00	0.55	0.17	0.02	0.08	0.11	0.13	0.16	0.17	0.18
Tuvalu (7)	1.00	0.55	0.17	0.02	0.08	0.11	0.13	0.16	0.17	0.18
PNG East (3)	0.34	0.12	0.05	0.13	0.15	0.17	0.18	0.19	0.19	0.20
Pohnpei FSM (3)	0.34	0.12	0.05	0.13	0.15	0.17	0.18	0.19	0.19	0.20
Kiribati Phoenix Is East (4)	0.52	0.23	0.01	0.10	0.14	0.15	0.17	0.18	0.19	0.19
Kiribati Phoenix Is West (6)	0.88	0.45	0.12	0.05	0.10	0.13	0.14	0.17	0.17	0.18
PNG (1)	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20	0.20
Fiji (1)	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20	0.20
Kiribati Line Is (1)	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20	0.20
Cook Islands (1)	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20	0.20
New Caledonia (0)	0.21	0.21	0.21	0.21	0.21	0.21	0.21	0.21	0.21	0.21
Yap FSM (0)	0.21	0.21	0.21	0.21	0.21	0.21	0.21	0.21	0.21	0.21
American Samoa (1)	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20	0.20
Palmyra (1)	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20	0.20
Upper 95% CI Future Annual Interactions; Scenario 6; 10 years										
Nauru (19)	1.00	1.00	0.77	0.28	0.12	0.04	0.01	0.08	0.10	0.13
Kiribati Gilbert Is (16)	1.00	1.00	0.61	0.20	0.06	0.00	0.05	0.10	0.12	0.14
PNG West (9)	1.00	0.77	0.28	0.04	0.05	0.09	0.11	0.14	0.16	0.17
Solomon (8)	1.00	0.66	0.23	0.01	0.06	0.10	0.12	0.15	0.16	0.17
Tuvalu (8)	1.00	0.66	0.23	0.01	0.06	0.10	0.12	0.15	0.16	0.17
PNG East (3)	0.34	0.12	0.05	0.13	0.15	0.17	0.18	0.19	0.19	0.20
Pohnpei FSM (3)	0.34	0.12	0.05	0.13	0.15	0.17	0.18	0.19	0.19	0.20
Kiribati Phoenix Is East (4)	0.52	0.23	0.01	0.10	0.14	0.15	0.17	0.18	0.19	0.19

r=0.019	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Kiribati Phoenix Is West (6)	0.88	0.45	0.12	0.05	0.10	0.13	0.14	0.17	0.17	0.18
PNG (1)	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20	0.20
Fiji (1)	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20	0.20
Kiribati Line Is (1)	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20	0.20
Cook Islands (1)	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20	0.20
New Caledonia (0)	0.21	0.21	0.21	0.21	0.21	0.21	0.21	0.21	0.21	0.21
Yap FSM (0)	0.21	0.21	0.21	0.21	0.21	0.21	0.21	0.21	0.21	0.21
American Samoa (1)	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20	0.20
Palmyra (1)	0.03	0.10	0.15	0.18	0.19	0.20	0.20	0.20	0.20	0.20

Table 19. Results of Scenarios 5 and 6 for 10-year population projections and r = 0.042. Numbers represent the percent difference between initial abundance and abundance in year 10; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00. Annual anticipated take levels (ATL) for each subpopulation are given in parentheses.

r=0.042	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Mean Future Annual Interactions; Scenario 5; 10 years										
Nauru (19)	1.00	1.00	0.57	0.03	0.16	0.25	0.30	0.38	0.41	0.43
Kiribati Gilbert Is (14)	1.00	1.00	0.27	0.13	0.26	0.32	0.36	0.42	0.44	0.46
PNG West (9)	1.00	0.57	0.03	0.25	0.34	0.39	0.41	0.45	0.46	0.48
Solomon (7)	0.90	0.33	0.10	0.31	0.38	0.42	0.44	0.47	0.48	0.49
Tuvalu (7)	0.90	0.33	0.10	0.31	0.38	0.42	0.44	0.47	0.48	0.49
PNG East (3)	0.09	0.16	0.34	0.43	0.46	0.48	0.49	0.50	0.50	0.51
Pohnpei FSM (3)	0.09	0.16	0.34	0.43	0.46	0.48	0.49	0.50	0.50	0.51
Kiribati Phoenix Is East (4)	0.29	0.04	0.28	0.40	0.44	0.46	0.47	0.49	0.50	0.50
Kiribati Phoenix Is West (6)	0.69	0.21	0.16	0.34	0.40	0.43	0.45	0.47	0.48	0.49
PNG (1)	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52	0.52
Fiji (1)	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52	0.52
Kiribati Line Is (1)	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52	0.52
Cook Islands (1)	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52	0.52
New Caledonia (0)	0.52	0.52	0.52	0.52	0.52	0.52	0.52	0.52	0.52	0.52
Yap FSM (0)	0.52	0.52	0.52	0.52	0.52	0.52	0.52	0.52	0.52	0.52
American Samoa (1)	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52	0.52
Palmyra (1)	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52	0.52
Upper 95% CI Future Annual Interactions; Scenario 6; 10 years										
Nauru (19)	1.00	1.00	0.57	0.03	0.16	0.25	0.30	0.38	0.41	0.43
Kiribati Gilbert Is (16)	1.00	1.00	0.39	0.07	0.22	0.29	0.34	0.40	0.42	0.45
PNG West (9)	1.00	0.57	0.03	0.25	0.34	0.39	0.41	0.45	0.46	0.48
Solomon (8)	1.00	0.45	0.04	0.28	0.36	0.40	0.42	0.46	0.47	0.48
Tuvalu (8)	1.00	0.45	0.04	0.28	0.36	0.40	0.42	0.46	0.47	0.48
PNG East (3)	0.09	0.16	0.34	0.43	0.46	0.48	0.49	0.50	0.50	0.51
Pohnpei FSM (3)	0.09	0.16	0.34	0.43	0.46	0.48	0.49	0.50	0.50	0.51
Kiribati Phoenix Is East (4)	0.29	0.04	0.28	0.40	0.44	0.46	0.47	0.49	0.50	0.50

r=0.042	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Kiribati Phoenix Is West (6)	0.69	0.21	0.16	0.34	0.40	0.43	0.45	0.47	0.48	0.49
PNG (1)	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52	0.52
Fiji (1)	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52	0.52
Kiribati Line Is (1)	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52	0.52
Cook Islands (1)	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52	0.52
New Caledonia (0)	0.52	0.52	0.52	0.52	0.52	0.52	0.52	0.52	0.52	0.52
Yap FSM (0)	0.52	0.52	0.52	0.52	0.52	0.52	0.52	0.52	0.52	0.52
American Samoa (1)	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52	0.52
Palmyra (1)	0.32	0.40	0.46	0.49	0.50	0.51	0.51	0.51	0.52	0.52

Table 20. Results of Scenarios 5 and 6 for 20-year population projections and r = 0.019. Numbers represent the percent difference between initial abundance and abundance in year 20; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00. Annual anticipated take levels (ATL) for each subpopulation are given in parentheses.

r=0.019	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Mean Future Annual Interactions; Scenario 5; 20 years										
Nauru (19)	1.00	1.00	1.00	0.62	0.26	0.08	0.03	0.17	0.23	0.29
Kiribati Gilbert Is (14)	1.00	1.00	1.00	0.32	0.06	0.07	0.15	0.25	0.30	0.34
PNG West (9)	1.00	1.00	0.62	0.08	0.10	0.19	0.25	0.32	0.35	0.37
Solomon (7)	1.00	1.00	0.38	0.04	0.18	0.25	0.29	0.35	0.37	0.39
Tuvalu (7)	1.00	1.00	0.38	0.04	0.18	0.25	0.29	0.35	0.37	0.39
PNG East (3)	0.74	0.26	0.10	0.28	0.34	0.37	0.39	0.41	0.42	0.43
Pohnpei FSM (3)	0.74	0.26	0.10	0.28	0.34	0.37	0.39	0.41	0.42	0.43
Kiribati Phoenix Is East (4)	1.00	0.50	0.02	0.22	0.30	0.34	0.37	0.40	0.41	0.42
Kiribati Phoenix Is West (6)	1.00	0.98	0.26	0.10	0.22	0.28	0.32	0.37	0.39	0.40
PNG (1)	0.06	0.22	0.34	0.40	0.42	0.43	0.44	0.45	0.45	0.45
Fiji (1)	0.06	0.22	0.34	0.40	0.42	0.43	0.44	0.45	0.45	0.45
Kiribati Line Is (1)	0.06	0.22	0.34	0.40	0.42	0.43	0.44	0.45	0.45	0.45
Cook Islands (1)	0.06	0.22	0.34	0.40	0.42	0.43	0.44	0.45	0.45	0.45
New Caledonia (0)	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46
Yap FSM (0)	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46
American Samoa (1)	0.06	0.22	0.34	0.40	0.42	0.43	0.44	0.45	0.45	0.45
Palmyra (1)	0.06	0.22	0.34	0.40	0.42	0.43	0.44	0.45	0.45	0.45
Upper 95% CI Future Annual Interactions; Scenario 6; 20 years										
Nauru (19)	1.00	1.00	1.00	0.62	0.26	0.08	0.03	0.17	0.23	0.29
Kiribati Gilbert Is (16)	1.00	1.00	1.00	0.44	0.14	0.01	0.10	0.22	0.27	0.32
PNG West (9)	1.00	1.00	0.62	0.08	0.10	0.19	0.25	0.32	0.35	0.37
Solomon (8)	1.00	1.00	0.50	0.02	0.14	0.22	0.27	0.33	0.36	0.38
Tuvalu (8)	1.00	1.00	0.50	0.02	0.14	0.22	0.27	0.33	0.36	0.38
PNG East (3)	0.74	0.26	0.10	0.28	0.34	0.37	0.39	0.41	0.42	0.43
Pohnpei FSM (3)	0.74	0.26	0.10	0.28	0.34	0.37	0.39	0.41	0.42	0.43
Kiribati Phoenix Is East (4)	1.00	0.50	0.02	0.22	0.30	0.34	0.37	0.40	0.41	0.42

r=0.019	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Kiribati Phoenix Is West (6)	1.00	0.98	0.26	0.10	0.22	0.28	0.32	0.37	0.39	0.40
PNG (1)	0.06	0.22	0.34	0.40	0.42	0.43	0.44	0.45	0.45	0.45
Fiji (1)	0.06	0.22	0.34	0.40	0.42	0.43	0.44	0.45	0.45	0.45
Kiribati Line Is (1)	0.06	0.22	0.34	0.40	0.42	0.43	0.44	0.45	0.45	0.45
Cook Islands (1)	0.06	0.22	0.34	0.40	0.42	0.43	0.44	0.45	0.45	0.45
New Caledonia (0)	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46
Yap FSM (0)	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46
American Samoa (1)	0.06	0.22	0.34	0.40	0.42	0.43	0.44	0.45	0.45	0.45
Palmyra (1)	0.06	0.22	0.34	0.40	0.42	0.43	0.44	0.45	0.45	0.45

Table 21. Results of Scenarios 5 and 6 for 20-year population projections and r = 0.042. Numbers represent the percent difference between initial abundance and abundance in year 20; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00. Annual anticipated take levels (ATL) for each subpopulation are given in parentheses.

r=0.042	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Mean Future Annual Interactions; Scenario 5; 20 years										
Nauru (19)	1.00	1.00	1.00	0.06	0.40	0.63	0.76	0.95	1.02	1.09
Kiribati Gilbert Is (14)	1.00	1.00	0.68	0.32	0.65	0.82	0.92	1.05	1.10	1.15
PNG West (9)	1.00	1.00	0.06	0.63	0.86	0.97	1.04	1.13	1.17	1.20
Solomon (7)	1.00	0.83	0.24	0.78	0.96	1.05	1.10	1.17	1.20	1.23
Tuvalu (7)	1.00	0.83	0.24	0.78	0.96	1.05	1.10	1.17	1.20	1.23
PNG East (3)	0.22	0.40	0.86	1.09	1.16	1.20	1.22	1.25	1.27	1.28
Pohnpei FSM (3)	0.22	0.40	0.86	1.09	1.16	1.20	1.22	1.25	1.27	1.28
Kiribati Phoenix Is East (4)	0.73	0.09	0.70	1.01	1.11	1.16	1.19	1.23	1.25	1.27
Kiribati Phoenix Is West (6)	1.00	0.52	0.40	0.86	1.01	1.09	1.13	1.19	1.22	1.24
PNG (1)	0.80	1.01	1.16	1.24	1.27	1.28	1.29	1.30	1.30	1.30
Fiji (1)	0.80	1.01	1.16	1.24	1.27	1.28	1.29	1.30	1.30	1.30
Kiribati Line Is (1)	0.80	1.01	1.16	1.24	1.27	1.28	1.29	1.30	1.30	1.30
Cook Islands (1)	0.80	1.01	1.16	1.24	1.27	1.28	1.29	1.30	1.30	1.30
New Caledonia (0)	1.32	1.32	1.32	1.32	1.32	1.32	1.32	1.32	1.32	1.32
Yap FSM (0)	1.32	1.32	1.32	1.32	1.32	1.32	1.32	1.32	1.32	1.32
American Samoa (1)	0.80	1.01	1.16	1.24	1.27	1.28	1.29	1.30	1.30	1.30
Palmyra (1)	0.80	1.01	1.16	1.24	1.27	1.28	1.29	1.30	1.30	1.30
Upper 95% CI Future Annual Interactions; Scenario 6; 20 years										
Nauru (19)	1.00	1.00	1.00	0.06	0.40	0.63	0.76	0.95	1.02	1.09
Kiribati Gilbert Is (16)	1.00	1.00	0.99	0.17	0.55	0.74	0.86	1.01	1.07	1.13
PNG West (9)	1.00	1.00	0.06	0.63	0.86	0.97	1.04	1.13	1.17	1.20
Solomon (8)	1.00	1.00	0.09	0.70	0.91	1.01	1.07	1.15	1.19	1.22
Tuvalu (8)	1.00	1.00	0.09	0.70	0.91	1.01	1.07	1.15	1.19	1.22
PNG East (3)	0.22	0.40	0.86	1.09	1.16	1.20	1.22	1.25	1.27	1.28
Pohnpei FSM (3)	0.22	0.40	0.86	1.09	1.16	1.20	1.22	1.25	1.27	1.28
Kiribati Phoenix Is East (4)	0.73	0.09	0.70	1.01	1.11	1.16	1.19	1.23	1.25	1.27

r=0.042	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Kiribati Phoenix Is West (6)	1.00	0.52	0.40	0.86	1.01	1.09	1.13	1.19	1.22	1.24
PNG (1)	0.80	1.01	1.16	1.24	1.27	1.28	1.29	1.30	1.30	1.30
Fiji (1)	0.80	1.01	1.16	1.24	1.27	1.28	1.29	1.30	1.30	1.30
Kiribati Line Is (1)	0.80	1.01	1.16	1.24	1.27	1.28	1.29	1.30	1.30	1.30
Cook Islands (1)	0.80	1.01	1.16	1.24	1.27	1.28	1.29	1.30	1.30	1.30
New Caledonia (0)	1.32	1.32	1.32	1.32	1.32	1.32	1.32	1.32	1.32	1.32
Yap FSM (0)	1.32	1.32	1.32	1.32	1.32	1.32	1.32	1.32	1.32	1.32
American Samoa (1)	0.80	1.01	1.16	1.24	1.27	1.28	1.29	1.30	1.30	1.30
Palmyra (1)	0.80	1.01	1.16	1.24	1.27	1.28	1.29	1.30	1.30	1.30

Table 22. Results of Scenarios 5 and 6 for 40-year population projections and r = 0.019. Numbers represent the percent difference between initial abundance and abundance in year 40; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00. Annual anticipated take levels (ATL) for each subpopulation are given in parentheses.

r=0.019	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Mean Future Annual Interactions; Scenario 5; 40 years										
Nauru (19)	1.00	1.00	1.00	1.00	0.64	0.20	0.07	0.43	0.57	0.70
Kiribati Gilbert Is (14)	1.00	1.00	1.00	0.79	0.15	0.17	0.37	0.62	0.73	0.83
PNG West (9)	1.00	1.00	1.00	0.20	0.25	0.47	0.60	0.78	0.85	0.92
Solomon (7)	1.00	1.00	0.94	0.10	0.45	0.62	0.72	0.86	0.92	0.97
Tuvalu (7)	1.00	1.00	0.94	0.10	0.45	0.62	0.72	0.86	0.92	0.97
PNG East (3)	1.00	0.64	0.25	0.69	0.84	0.92	0.96	1.02	1.04	1.07
Pohnpei FSM (3)	1.00	0.64	0.25	0.69	0.84	0.92	0.96	1.02	1.04	1.07
Kiribati Phoenix Is East (4)	1.00	1.00	0.05	0.54	0.74	0.84	0.90	0.98	1.01	1.04
Kiribati Phoenix Is West (6)	1.00	1.00	0.64	0.25	0.54	0.69	0.78	0.90	0.95	0.99
PNG (1)	0.15	0.54	0.84	0.99	1.04	1.06	1.08	1.10	1.11	1.11
Fiji (1)	0.15	0.54	0.84	0.99	1.04	1.06	1.08	1.10	1.11	1.11
Kiribati Line Is (1)	0.15	0.54	0.84	0.99	1.04	1.06	1.08	1.10	1.11	1.11
Cook Islands (1)	0.15	0.54	0.84	0.99	1.04	1.06	1.08	1.10	1.11	1.11
New Caledonia (0)	1.14	1.14	1.14	1.14	1.14	1.14	1.14	1.14	1.14	1.14
Yap FSM (0)	1.14	1.14	1.14	1.14	1.14	1.14	1.14	1.14	1.14	1.14
American Samoa (1)	0.15	0.54	0.84	0.99	1.04	1.06	1.08	1.10	1.11	1.11
Palmyra (1)	0.15	0.54	0.84	0.99	1.04	1.06	1.08	1.10	1.11	1.11
Upper 95% CI Future Annual Interactions; Scenario 6; 40 years										
Nauru (19)	1.00	1.00	1.00	1.00	0.64	0.20	0.07	0.43	0.57	0.70
Kiribati Gilbert Is (16)	1.00	1.00	1.00	1.00	0.35	0.03	0.25	0.54	0.66	0.78
PNG West (9)	1.00	1.00	1.00	0.20	0.25	0.47	0.60	0.78	0.85	0.92
Solomon (8)	1.00	1.00	1.00	0.05	0.35	0.54	0.66	0.82	0.89	0.95
Tuvalu (8)	1.00	1.00	1.00	0.05	0.35	0.54	0.66	0.82	0.89	0.95
PNG East (3)	1.00	0.64	0.25	0.69	0.84	0.92	0.96	1.02	1.04	1.07
Pohnpei FSM (3)	1.00	0.64	0.25	0.69	0.84	0.92	0.96	1.02	1.04	1.07
Kiribati Phoenix Is East (4)	1.00	1.00	0.05	0.54	0.74	0.84	0.90	0.98	1.01	1.04

r=0.019	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Kiribati Phoenix Is West (6)	1.00	1.00	0.64	0.25	0.54	0.69	0.78	0.90	0.95	0.99
PNG (1)	0.15	0.54	0.84	0.99	1.04	1.06	1.08	1.10	1.11	1.11
Fiji (1)	0.15	0.54	0.84	0.99	1.04	1.06	1.08	1.10	1.11	1.11
Kiribati Line Is (1)	0.15	0.54	0.84	0.99	1.04	1.06	1.08	1.10	1.11	1.11
Cook Islands (1)	0.15	0.54	0.84	0.99	1.04	1.06	1.08	1.10	1.11	1.11
New Caledonia (0)	1.14	1.14	1.14	1.14	1.14	1.14	1.14	1.14	1.14	1.14
Yap FSM (0)	1.14	1.14	1.14	1.14	1.14	1.14	1.14	1.14	1.14	1.14
American Samoa (1)	0.15	0.54	0.84	0.99	1.04	1.06	1.08	1.10	1.11	1.11
Palmyra (1)	0.15	0.54	0.84	0.99	1.04	1.06	1.08	1.10	1.11	1.11

Table 23. Results of Scenarios 5 and 6 for 40-year population projections and r = 0.042. Numbers represent the percent difference between initial abundance and abundance in year 40; red numbers represent population declines and black numbers represent population increases. Total extirpation is represented by 1.00. Annual anticipated take levels (ATL) for each subpopulation are given in parentheses.

r=0.042	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Mean Future Annual Interactions; Scenario 5; 40 years										
Nauru (19)	1.00	1.00	1.00	0.21	1.31	2.08	2.53	3.14	3.39	3.62
Kiribati Gilbert Is (14)	1.00	1.00	1.00	1.06	2.16	2.71	3.04	3.48	3.66	3.83
PNG West (9)	1.00	1.00	0.21	2.08	2.84	3.22	3.45	3.75	3.88	3.99
Solomon (7)	1.00	1.00	0.80	2.58	3.18	3.48	3.65	3.89	3.99	4.08
Tuvalu (7)	1.00	1.00	0.80	2.58	3.18	3.48	3.65	3.89	3.99	4.08
PNG East (3)	0.72	1.31	2.84	3.60	3.86	3.98	4.06	4.16	4.20	4.24
Pohnpei FSM (3)	0.72	1.31	2.84	3.60	3.86	3.98	4.06	4.16	4.20	4.24
Kiribati Phoenix Is East (4)	1.00	0.29	2.33	3.35	3.69	3.86	3.96	4.09	4.15	4.20
Kiribati Phoenix Is West (6)	1.00	1.00	1.31	2.84	3.35	3.60	3.75	3.96	4.04	4.12
PNG (1)	2.67	3.35	3.86	4.11	4.20	4.24	4.26	4.30	4.31	4.32
Fiji (1)	2.67	3.35	3.86	4.11	4.20	4.24	4.26	4.30	4.31	4.32
Kiribati Line Is (1)	2.67	3.35	3.86	4.11	4.20	4.24	4.26	4.30	4.31	4.32
Cook Islands (1)	2.67	3.35	3.86	4.11	4.20	4.24	4.26	4.30	4.31	4.32
New Caledonia (0)	4.37	4.37	4.37	4.37	4.37	4.37	4.37	4.37	4.37	4.37
Yap FSM (0)	4.37	4.37	4.37	4.37	4.37	4.37	4.37	4.37	4.37	4.37
American Samoa (1)	2.67	3.35	3.86	4.11	4.20	4.24	4.26	4.30	4.31	4.32
Palmyra (1)	2.67	3.35	3.86	4.11	4.20	4.24	4.26	4.30	4.31	4.32
Upper 95% CI Future Annual Interactions; Scenario 6; 40 years										
Nauru (19)	1.00	1.00	1.00	0.21	1.31	2.08	2.53	3.14	3.39	3.62
Kiribati Gilbert Is (16)	1.00	1.00	1.00	0.55	1.82	2.46	2.84	3.35	3.55	3.75
PNG West (9)	1.00	1.00	0.21	2.08	2.84	3.22	3.45	3.75	3.88	3.99
Solomon (8)	1.00	1.00	0.29	2.33	3.01	3.35	3.55	3.82	3.93	4.04
Tuvalu (8)	1.00	1.00	0.29	2.33	3.01	3.35	3.55	3.82	3.93	4.04
PNG East (3)	0.72	1.31	2.84	3.60	3.86	3.98	4.06	4.16	4.20	4.24
Pohnpei FSM (3)	0.72	1.31	2.84	3.60	3.86	3.98	4.06	4.16	4.20	4.24
Kiribati Phoenix Is East (4)	1.00	0.29	2.33	3.35	3.69	3.86	3.96	4.09	4.15	4.20

r=0.042	**Initial Abundance**									
Population (ATL)	**60**	**100**	**200**	**400**	**600**	**800**	**1000**	**1500**	**1875**	**2464**
Kiribati Phoenix Is West (6)	1.00	1.00	1.31	2.84	3.35	3.60	3.75	3.96	4.04	4.12
PNG (1)	2.67	3.35	3.86	4.11	4.20	4.24	4.26	4.30	4.31	4.32
Fiji (1)	2.67	3.35	3.86	4.11	4.20	4.24	4.26	4.30	4.31	4.32
Kiribati Line Is (1)	2.67	3.35	3.86	4.11	4.20	4.24	4.26	4.30	4.31	4.32
Cook Islands (1)	2.67	3.35	3.86	4.11	4.20	4.24	4.26	4.30	4.31	4.32
New Caledonia (0)	4.37	4.37	4.37	4.37	4.37	4.37	4.37	4.37	4.37	4.37
Yap FSM (0)	4.37	4.37	4.37	4.37	4.37	4.37	4.37	4.37	4.37	4.37
American Samoa (1)	2.67	3.35	3.86	4.11	4.20	4.24	4.26	4.30	4.31	4.32
Palmyra (1)	2.67	3.35	3.86	4.11	4.20	4.24	4.26	4.30	4.31	4.32