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

Action Agency:	National Marine Fisheries Service, Pacific Islands Region, Sustainable Fisheries Division
Federal Action:	Continued authorization of the Hawaii Pelagic Shallow-Set Longline Fishery
Consultation Conducted by:	National Marine Fisheries Service, Pacific Islands Region, Protected Resources Division
NMFS File No. (PCTS):	PIR-
PIRO Reference No.:	
Approved By:	____________________________________ Michael D. Tosatto Regional Administrator, Pacific Islands Region
Date Issued:	____________________________________

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1 INTRODUCTION

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

For the actions described in this document, the action agency is NMFS' Pacific Islands Region (PIRO) Sustainable Fisheries Division (SFD), which proposes to authorize the continued operation of the Hawaii shallow-set longline fishery (HI SSLL), as currently managed under the existing regulatory framework of the Pelagic Fishery Ecosystem Plan (FEP) and other applicable laws. The consulting agency for this proposal is PIRO's Protected Resources Division (PRD). This document represents NMFS' final biological opinion on the effects of the proposed action on endangered and threatened species and critical habitat that has been designated for those species. This biological opinion has been prepared in accordance with the requirements of section 7 of the ESA, the implementing regulations (50 CFR 402), agency policy, and guidance and considers and is based on information contained in SFD's biological evaluation (NMFS 2018), NMFS and FWS recovery plans and status reviews for sea turtles (NMFS and FWS 1998 a,b,c,d; 2007 a,b,c; 2013, 2014; Seminoff et al. 2015), marine mammals (NMFS 2010a, NMFS 2010b, and elasmobranchs (Young et al. 2016, Miller and Klimovich 2016) and the Western Pacific Fishery Management Council (Council) annual stock assessment and fishery evaluation (SAFE reports) for Pacific Island pelagic FEP for 2017 (WPRFMC 2018), and other sources of information as cited herein.

1.1 Consultation History

The proposed federal action addressed by this biological opinion is the continued operation of the HI SSLL fishery. Prior to 2008, NMFS consulted on the HI SSLL fishery as part of the Fisheries Management Plan for the Pelagic Fisheries (pelagic FMP). Consultations on the pelagic FMP were conducted in 1998, 2001 and 2004. Consultation histories for earlier consultations on the FMP and the HI SSLL fishery can be found in the 1998, 2001, 2004, 2008, and 2012 biological opinions.

The HI SSLL fishery has been the subject of several court orders, and has operated under a restricted fishing regime to protect listed sea turtles when NMFS completed a new consultation on January 30, 2012. The 2012 biological opinion concluded that the continued operation of the

HI SSLL fishery was not likely to jeopardize the continued existence of humpback whales[1], loggerhead, leatherback, olive ridley, and green sea turtles.

On November 2, 2012, Plaintiffs Turtle Island Restoration Network and Center for Biological Diversity filed a lawsuit against NMFS under the ESA, MSA, and their implementing regulations, challenging among other decisions the NMFS's final rule approving the continued operation of the shallow-set fishery under sea turtle annual interaction limits of 34 loggerheads and 26 leatherbacks, based on a 2012 no-jeopardy biological opinion. After the parties moved for summary judgment, on August 23, 2013 the district court ruled in the agency's favor on all of Plaintiffs' claims, and Plaintiffs appealed.

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

On March 2, 2015, NMFS determined that the continued authorization of the HI SSLL fishery under the FEP is not likely to adversely affect the Main Hawaiian Islands insular false killer whale (MHI IFKW) distinct population segment (DPS) and the Eastern Pacific scalloped hammerhead shark DPS (NMFS 2015a).

On September 16, 2015, NMFS determined that the continued authorization of the HI SSLL and deep-set longline (DSLL) fisheries is not likely to adversely affect Hawaiian monk seal critical habitat and fin whales (NMFS 2015b).

On December 27, 2017, the US Court of Appeals, Ninth Circuit, vacated the loggerhead decisions and on May 2018, a court-approved settlement agreement was completed that would shut down the HI SSLL fishery for the remainder of the 2018 year, and required implementation of the 2004 hard cap at 17 loggerhead sea turtles starting in January 2019.

On April 20, 2018, NMFS SFD requested reinitiation of formal consultation. Consistent with 50 CFR 402.16, SFD reinitiated consultation because the HI SSLL met three of the four possible reinitiation triggers. The HI SSLL exceeded the amount of incidental take for olive ridley sea turtles; new information revealed effects of the action that may affect listed species or critical habitat in a manner or to an extent not previously considered; and several new species were listed that may be affected by the action. Specifically, the following conditions met the reinitiation trigger:

- In 2015, for the first time the HI SSLL interacted with a Guadalupe fur seal. Three additional interactions occurred between November and December 2017. The Guadalupe fur seal is listed as threatened throughout its range (50 FR 51252, December 16, 1985).
- On April 6, 2016, NMFS and USFWS issued a final rule to list 11 distinct population segments (DPS) of green sea turtle under the ESA (81 FR 20058). This final rule removed the previous range-wide listing and, in its place, listed eight DPSs as threatened and three as endangered. Six green sea turtle DPS occur in the Pacific Ocean and within

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[1] The Hawaii Humpback Whale Distinct Population Segment was delisted from the ESA on September 8, 2016 (81 FR 62259).

range of the HI SSLL fishery: the East Indian-West Pacific, Central West Pacific, Southwest Pacific, Central South Pacific, Central North Pacific, and the East Pacific DPS. While the 2012 biological opinion evaluated the fishery's impact on the globally listed green sea turtle, the listing of 11 new green sea turtle DPS triggered the requirement to reinitiate consultation. The HI SSLL fishery interacted with nine green sea turtles between 2004 and 2017.

- In January 2018, NMFS listed two new species, the giant manta ray and the oceanic whitetip shark, as threatened under the ESA (83 FR 2196 and 83 FR 4153, respectively). The HI SSLL fishery interacts with these two species.
- In July 2018, NMFS designated critical habitat for the MHI insular false killer whale DPS (83 FR 35062). The designated area encompasses waters from the 45-meter depth contour to the 3,200 m depth contour around the main Hawaiian Islands from Niihau east to Hawaii which is part of the action area for the HI SSLL.
- In 2018, NMFS exceeded the authorized take of olive ridley sea turtles as anticipated in the 2012 biological opinion on the HI SSLL fishery. NMFS anticipated and authorized a two-year ITS of four olive ridley sea turtles in the fishery. The ITS was effective on January 30, 2012. In 2017, fisheries observers documented four interactions with olive ridley sea turtles, and one interaction in 2018 for a total of five interactions in a two-year period.

Some other key events that are part of the history on this consultation include:

- On May 16, 2018, PRD requested more information on which species in the California EEZ that the action would not likely adversely affect since they were not specified in the biological Evaluation.
- On June 22, 2018, PRD sent a letter to SFD to inform them that they provided all information required by regulation for initiating interagency consultation, as outlined in 50 CFR 402.14. PRD also confirmed the mutually agreed upon extension date of October 31, 2018 for a complete biological opinion.
- On October 31, 2018, PRD and SFD held a conference call with HLA to appraise them of the status of the consultation and discuss the approach that PRD would take to analyzing the effects of the proposed action.
- On November 30, 2018, PRD provided a copy of the draft status of leatherback and loggerhead sea turtles to SFD for review, and for sharing with their applicant.
- On February 6, 2019, NMFS advised the applicant that more time was necessary to finish the draft biological opinion, in part because of the government shutdown that occurred in December 2018 through January 2019.
- On February 18, 2019, PRD provided a copy of the draft effects section to the Regional Administrator for sharing with the applicant. On that same day, NMFS provided a copy of the draft effects section of this biological opinion to the applicant for review. On March 1, NMFS received comments from the applicant on the draft effects section of this biological opinion.

- On January 19, 2019, the HI SSLL fishery reached its cap of 17 loggerhead sea turtles (a total of 20 loggerhead sea turtles were taken by March 20, 2019) and the fishery was closed for the remainder of the year.

2 DESCRIPTION OF THE PROPOSED ACTION

The proposed action is the continued operation of the HI SSLL fishery, as currently managed under the existing regulatory framework of the Pelagic FEP and other applicable laws.

Longline fishing employs a type of fishing gear consisting of a mainline that exceeds one nautical mile (6,076 ft.) in length suspended horizontally in the water column, from which branch lines with baited hooks are attached (NMFS 2008b). Longline fishing allows a vessel to distribute effort over a large area to harvest fish that are not concentrated in great numbers. Overall catch rates in relation to the number of hooks are generally low (less than 2%). Longline fishing involves setting (deploying) a mainline horizontally at a preferred depth in the water column using floats spaced at regular intervals. Crewmembers usually attach three to five radio buoys at regular intervals along the mainline so the line may be easily located for retrieving (hauling) the gear and retrieving line segments if the mainline breaks during fishing operations.

Crewmembers clip branch lines to the mainline at regular intervals, and each branch line has a single baited hook. Mainline lengths can be 30 to 100 km (18 to 60 nm) long. After deploying the mainline, the gear fishes (soaks) for several hours before being hauled. In longlining, a "set" is the deployment and retrieval of a discrete unbroken section of mainline, floats, and branch lines. Usually, crewmembers make one set per day. Shallow-set fishing trips are usually 4-5 weeks long, with about 17 days spent fishing. By comparison, deep-set trips are historically 3-4 weeks long, and about 13 days fishing (NMFS 2001, NMFS 2005a, WPFMC 2006, Beverly and Chapman 2007, WPFMC 2009a). Fig. 2 illustrates the difference between shallow-set and deep-set gear configuration.

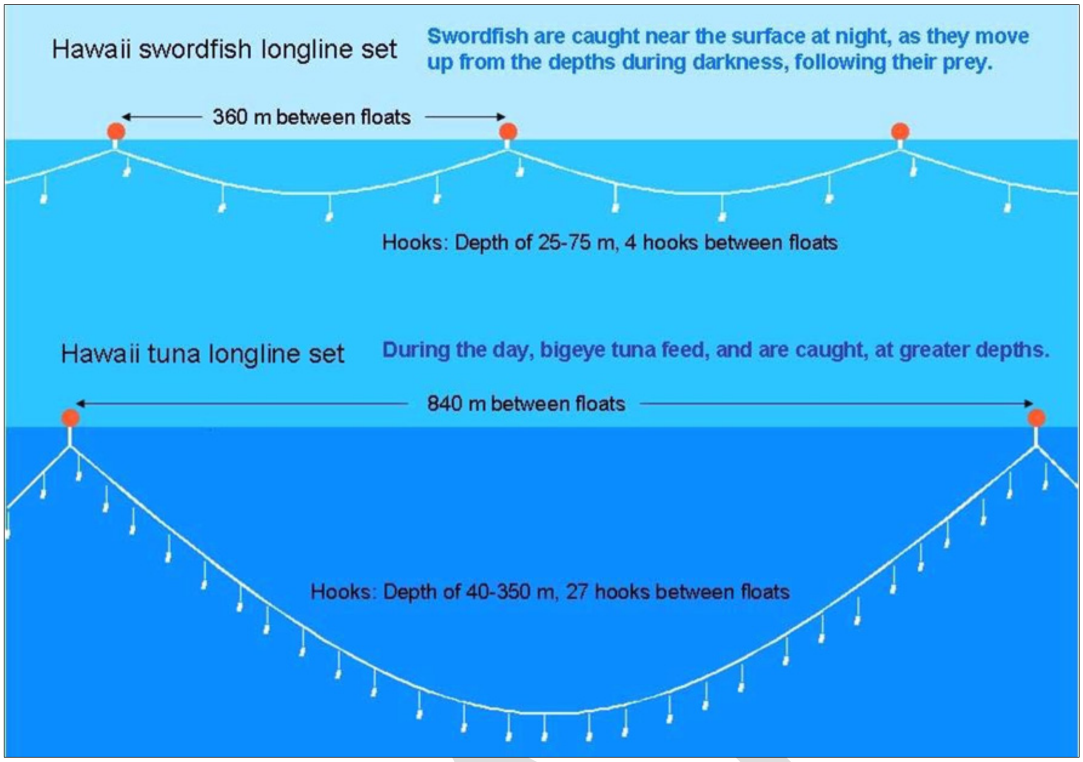


Figure 1. Generalized depiction of shallow-set and deep-set gear configuration

In shallow-set longline fishing, the bait is set at depths of 30 to 90 m. The portion of the mainline with branch lines attached is suspended between floats at about 20 to 75 m deep, and the branch lines hang off the mainline another 10 to 15 m. Fishermen clip four to six branch lines to the mainline between floats, and a typical set for swordfish uses about 800 to 1,000 hooks. Shallow-set longline gear is set at night with luminescent light sticks attached to branch lines and hauled during the day. NMFS requires the fishery to use 18/0 or larger circle hooks and mackerel-type bait to reduce turtle bycatch. The most productive swordfish fishing areas for Hawaii longline fishing are north of Hawaii outside the EEZ on the high seas. Shallow-set longline vessels have displacement hulls and travel at speeds less than 10 kt. Vessel sizes range up to nearly the maximum 100-foot limit, but the average size is 65 to 70 ft.

A summary of many of the current requirements that are part of the proposed action follows:

Fishing Permits and Certificates on board the vessel

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

- State of Hawaii Commercial Marine License.

Reporting, Monitoring, and Gear Identification

- Logbook for recording effort, catch, and other data.
- Transshipping Logbook, if applicable.
- Marine Mammal Authorization Program Mortality/Injury Reporting Form.
- Vessel monitoring system.
- Vessel and fishing gear identification.

Notification Requirement and Observer Placement

- Notify NMFS before departure on a fishing trip to declare the trip type (shallow-set or deep-set).
- Each fishing trip is required to have a fishery observer on board if requested by NMFS; NMFS places observers on every HI SSLL fishing trip, resulting in 100% coverage.
- Fisheries observer guidelines are used.

Prohibited Areas in Hawaii

- Northwestern Hawaiian Islands (NWHI) Longline Protected Species Zone.
- Main Hawaiian Islands Longline Fishing Prohibited Area.
- Papahanaumokuakea Marine National Monument: Commercial fishing is prohibited in the Monument, which has boundaries that align with the NWHI Longline Protected Species Zone.

Protected Species Workshop (PSW)

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

Sea Turtle, Seabird, and Shark Handling and Mitigation Measures

- Vessel owners and operators are required to adhere to regulations for safe handling and release of sea turtles and seabirds.
- Vessel owners and operators must have on board the vessel all required turtle handling/dehooking gear specified in regulations.
- Vessel owners and operators can choose between side setting and stern setting, with additional requirements to reduce seabird interactions.
- When shallow-set longline fishing north of the Equator:
- Use 18/0 or larger circle hooks with no more than 10° offset.

- Use mackerel-type bait.
- Set at night for stern set vessels.
- Vessel owners, operators, and crew are required to release any oceanic whitetip shark or silky shark and take reasonable steps for its safe release.

Marine Mammal Handling and Release

- Vessel owners and operators must follow the marine mammal handling guidelines provided at the PSW.
- Vessel owners or operator must submit the Marine Mammal Authorization Program (MMAP) Mortality/Injury Reporting Form within 48 hours after the end of the fishing trip to NMFS to report injuries or mortalities of marine mammals (50 CFR 229.6).

Unless otherwise noted, most of the above regulations are at 50 CFR Part 665. A summary of regulations for Hawaii longline fisheries (shallow-set and deep-set combined) is provided by the Summary of Hawaii Longline Fishing Regulations (NMFS 2014a).

Beginning in 2004, the fishery resumed operations with suite of mitigation measures designed to reduce interactions with ESA-listed sea turtles. These included requirements to use large circle hooks and mackerel-type bait to reduce and mitigate interactions with sea turtles, a set certificate program that limited the annual number of sets to 2,120, maximum annual interaction limits for loggerhead and leatherback sea turtles and a requirement for owners and operators of longline vessels to attend a protected species education workshop.

In 2010, NMFS implemented Amendment 18, which intended to allow for the attainment of optimum yield in the swordfish fishery while mitigating impacts to listed species. Specifically, Amendment 18 removed the shallow-set effort limit and eliminated the set certificate program (74 FR 65460, January 10, 2010), but retained all other sea turtle mitigation measures. Fishery effort increased slightly in 2010, but has remained below 2,000 sets annually since 2004 (Table 1), and well below the estimated 5,500 sets analyzed in Amendment 18. Accordingly, NMFS believes that past annual effort since 2004 provides the better predictor of future effort, as discussed below.

Although participation and effort has varied and declined in recent years, NMFS expects fishing effort to remain within the approximate range of effort observed from 2004 to 2016 as shown in Table 1. Fishing effort may reasonably reach 2,000 sets annually for the following reasons:

1. 2,000 sets annually is within historical levels (prior to the 2001-2004 closure);
2. Global swordfish demand and demand for fresh swordfish from Hawaii fisheries can fluctuate, resulting in price changes that, in turn, are one driver of fishing effort; and
3. Annual bigeye tuna catch limits applicable to Hawaii longline vessels could change, making the targeting of swordfish a likely alternative in the event of a bigeye closure.

Table 1. Hawaii pelagic longline fishery shallow-set fishery effort, 2004-2018.

Year	Active Vessels	Number of Trips	Number of Sets	Number of Hooks

2004	7	11	135	113,318
2005	33	109	1,645	1,385,457
2006	35	57	850	705,466
2007	28	88	1,570	1,371,949
2008	27	93	1,597	1,496,298
2009	28	112	1,762	1,721,346
2010	28	108	1,833	1,803,432
2011	20	82	1,468	1,489,243
2012	18	81	1,355	1,453,234
2013	15	58	962	1,060,341
2014	20	81	1,338	1,483,809
2015	22	65	1,110	1,235,703
2016	13	40	670	719,385
2017	18	61	949	1,027,013
2018	11	30	420	486,013

Source: NMFS unpublished data and NMFS PIFSC logbook data, https://www.pifsc.noaa.gov/fmb/reports.php

Vessels operating in the HI SSLL operate out of the ports of Honolulu HI and California during certain times of the year. The California Ports include Long Beach, San Pedro, San Diego, and San Francisco. They are prohibited from fishing in the California EEZ. The majority operate out of San Francisco (77%); only one vessel has departed out of San Diego, and the remaining operate out of Los Angeles ports. The number of vessels that operate out of California varies by year and ranges from 1-13 vessels with 1-31 departures from California (Figure 2).

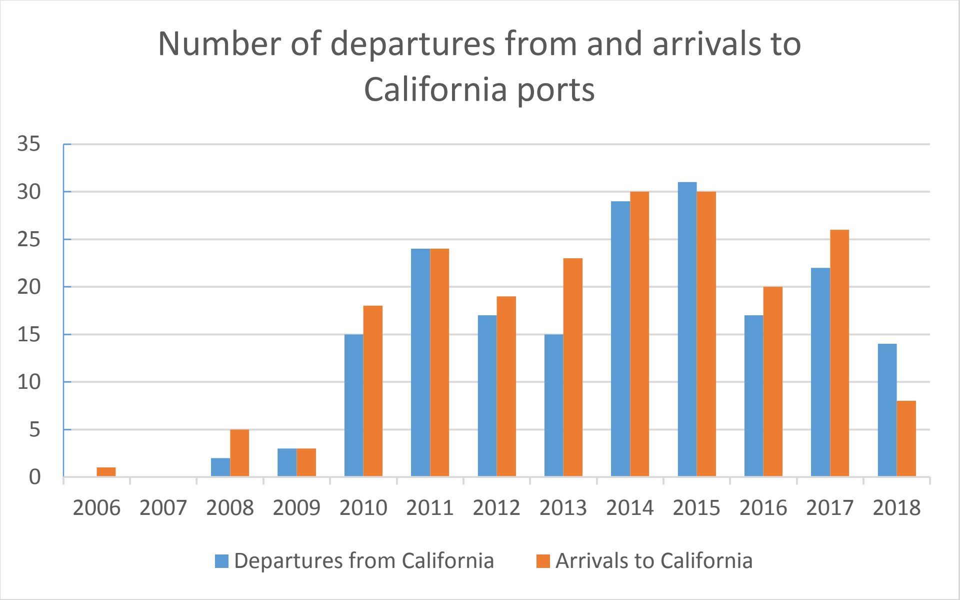


Figure 2. Graph depicting the number of vessel arrivals and departures to and from California ports in each calendar year. Some trips start in Honolulu and end in California or in the reverse order.

3 APPROACH TO THE ASSESSMENT

3.1 Overview of NMFS Assessment Framework

NMFS approaches its section 7 analyses through a series of sequential steps. The first step of this sequence identifies those 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 is the *Action Area* for consultation.

The second step of our analyses identifies the listed species and designated critical habitat (collectively, listed resources) that are likely to co-occur with these potential stressors in space and time. If we conclude that such co-occurrence is likely, we then try to estimate the nature of that co-occurrence (these represent our *exposure analyses*); that is, the intensity of the stressors we expect listed species to be exposed to and the duration and frequency of any exposure. 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.

Our exposure analyses also identify the physical or biological features of critical habitat, including any physical or biological features ("essential features") of critical habitat or areas that require special management consideration or protection such as sites for breeding and rearing, food, water, space for growth and normal behavior, and cover and shelter; and we 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. This information is represented in the *Status of Listed Resources and Critical Habitat*. In this section we review the species' legal status, trends, and the threats that led to this status as well as those that may be impeding the species' chances of recovery. We also assess the effects of past and ongoing human and natural factors leading to the current status of the species, its habitat, and ecosystem. We present this information in the *Environmental Baseline*. The environmental baseline is designed to assess the condition of the habitat and the species within the action area. The information in the *Environmental Baseline* with the *Status of Listed Resources and Critical Habitat* forms the foundation of our analyses and determining the risk a proposed action poses a particular species or their designated critical habitat.

Once we identify the listed resources 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 listed resources are likely to respond given their exposure (these represent our *response analyses*). The final steps of our analyses—establishing the risks those responses pose to listed resources—are different for listed species and designated critical habitat (these represent our *risk analyses*).

3.2 Risk Analyses for Endangered and Threatened Species

Our jeopardy analyses ask whether an action agency has *insured* that its action – the action considered in this consultation is the continued authorization of the HI SSLL fishery – *is not likel*y to jeopardize the continued existence of endangered or threatened species. To assess whether an action agency has complied with this standard, our jeopardy analyses ask if the evidence available allows us to conclude that the agency has *insured* that any reductions in *numbers*, *reproduction*, or the *distribution* of endangered or threatened species that are likely to result from its proposed action are likely to be inconsequential for these species (50 CFR §402.02). This standard specifically focuses on endangered or threatened species as those "species" have been listed, which can include a biological species, a subspecies, or distinct population segments of vertebrate species.

Few federal actions affect every member of endangered and threatened species that occur in marine or coastal ecosystems. Instead, the overwhelming majority of federal actions affect some members of some populations of these listed species. To determine whether an action that affects individuals is likely to affect the listed species those individuals belong to, we rely on the relationship between species, populations, and individuals. The viability of listed species (their probability of extinction or probability of persistence) depends on the viability of the populations that comprise the species while the viability of populations are determined by the fate of the individuals that comprise them: populations grow or decline as the individuals that comprise the population live, die, grow, mature, migrate, and reproduce (or fail to do so).

Our risk analyses reflect these relationships between listed species and the populations that comprise them, and the individuals that comprise those populations. Our risk analyses begin by identifying the probable risks actions pose to listed individuals that are likely to be exposed to an action's effects. Our analyses then integrate those individuals' risks to identify consequences to the populations they represent and next we determine the consequences of population-level effects on the species as listed.

Our risk analyses reflect these relationships between listed species and the populations that comprise them, and the individuals that comprise those populations. Our risk analyses begin by identifying the probable risks actions pose to listed individuals that are likely to be exposed to an action's effects. Our analyses then integrate those individuals' risks to identify consequences to the populations they represent and next we determine the consequences of population-level effects on the species as listed.

We measure risks to listed individuals using the individual's "fitness," which are changes in an individual's growth, survival, annual reproductive success, or lifetime reproductive success as a result of their exposure to a stressor. In particular, we examine the scientific and commercial data available to determine if an individual's probable responses to an action's effect on the environment (which we identify during our response analyses) are likely to have consequences for the individual's fitness. When individual listed plants or animals are expected to experience reductions in fitness, we would expect those reductions to also reduce the abundance, reproduction rates, or growth rates (or increase variance in one or more of these rates) of the populations those individuals represent (see Stearns 1992). A reduction in one or more of these variables (or one of the variables we derive from them) is a *necessary* condition for reductions in a population's viability, which itself is a *necessary* condition for reductions in a species' viability. On the other hand, when listed plants or animals exposed to an action's effects are not expected to experience reductions in fitness, we would not expect the action to have adverse consequences on the viability of the populations those individuals represent or the species those populations comprise (for example, see Anderson 2000; Mills and Beatty 1979; Stearns 1992). If we conclude that listed plants or animals are not likely to experience reductions in their fitness we would conclude our assessment.

If, however, we conclude that listed plants or animals are likely to experience reductions in their fitness, our assessment tries to determine if those reductions are likely to be sufficient to reduce the viability of the populations those individuals represent (measured using changes in the population' abundance, reproduction, spatial structure and connectivity, growth rates, genetic health, or variance in these measures to make inferences about the population's extinction risks). In this step of our analyses, we use the population's base condition (established in the *Environmental Baseline* and *Status of Listed Resources* sections of this biological opinion) as our point of reference.

Finally, our assessment tries to determine if changes in population viability are likely to be sufficient to reduce the viability of the species those populations comprise. That is, our assessment tries to determine if the action agency can insure that changes in the populations are not likely to reduce the viability of the species those populations comprise. In this step of our analyses, we use the species' status (established in the *Status of Listed Resources* and informed by the *Environmental Baseline* sections of this opinion) as our point of reference. The primary advantage of this approach is that it considers the consequences of the response of endangered and threatened species in terms of fitness costs. Individual-level effects can then be translated into changes in demographic parameters of populations, thus allowing for an assessment of the biological significance of particular human disturbances.

Biological opinions, then, distinguish among different kinds of "significance". First, we focus on potential physical, chemical, or biotic stressors that are "significant" or most important (also, salient), and distinct from ambient or background. We then ask if (a) exposing individuals to those potential stressors is likely to represent a "significant" adverse experience in the life of individuals that have been exposed; (b) exposing individuals to those potential stressors is likely to cause the individuals to experience “significant” physical, chemical, or biotic responses; and (c) any “significant” physical, chemical, or biotic responses are likely to have “significant” consequence for the fitness of the individual animal. In the latter two cases, (items (b) and (c)), the term “significant” means “clinically or biotically significant” rather than statistically significant.

For populations (or sub-populations, demes, etc.), we are concerned about whether the number of individuals that experience “significant” reductions in fitness and the nature of any fitness reductions are likely to have a “significant” consequence for the viability (i.e. probability of demographic, ecological, or genetic extinction) of the populations(s) those individuals represent. Here “significant” also means “clinically or biotically significant” rather than statistically significant.

For “species” (the entity listed as threatened or endangered, which may not be the biological species), we are concerned about whether the number of populations that experience “significant” reductions in viability (that is, increases their extinction probabilities) and the nature of any reductions in viability are likely to have “significant” consequences for the viability (the probability of demographic, ecological, or genetic extinction) of the listed species those populations comprise. Here again, “significant” also means “clinically or biotically significant” rather than statistically significant.

In this step, we also ask whether or to what degree the agency has insured that probable reductions in a species’ viability are not likely to have significant consequences for the viability of the listed species those populations comprise. The answer to this question informs our conclusion about whether an agency has insured that any reductions in numbers, reproduction, or the distribution of threatened or endangered species that are likely to result from its proposed action are likely to be inconsequential for these species (50 CFR §402.02).

3.3 Risk Analyses for Critical Habitat

Our destruction or adverse modification determinations must be based on an action’s effects on the value of habitat that has been designated as critical for the conservation of listed species (2016; 81 FR 7214). If an area encompassed in a critical habitat designation is likely to be exposed to the direct or indirect consequences of the proposed action on the natural environment, we ask if essential features included in the designation (or primary or secondary constituent elements, if there are any) or physical, chemical or biotic phenomena that give the designated area value for the conservation are likely to respond to that exposure.

In this step of our assessment, we must identify (a) the spatial distribution of stressors and subsidies produced by an action; (b) the temporal distribution of stressors and subsidies produced by an action; (c) changes in the spatial distribution of the stressors with time; (d) the intensity of stressors in space and time; (e) the spatial distribution of biological and physical features of designated critical habitat; and (f) the temporal distribution of biological and physical features of designated critical habitat.

If the biological and physical features (also called essential features) of designated critical habitat that give the designated area value for the conservation of listed species are likely to respond given exposure to the direct or indirect consequences of the proposed action on the natural environment, we ask if those responses are likely to be sufficient to reduce the quantity, quality or availability of those constituent elements or physical, chemical or biotic phenomena.

In this step of our assessment, we must identify or make assumptions about (a) the habitat's probable conditions before any exposure as our point of reference (that part of the impact of the *Environmental Baseline* on the value of habitat that has been designated as critical for the conservation of listed species); (b) the ecology of the habitat at the time of exposure; (c) where the exposure is likely to occur; and (d) when the exposure is likely to occur; (e) the intensity of exposure; (f) the duration of exposure; and (g) the frequency of exposure.

In this step of our assessment, we recognize that the conservation value of critical habitat, like the base condition of individuals and populations, is a dynamic property that changes over time in response to changes in land use patterns, climate (at several spatial scales), ecological processes, changes in the dynamics of biotic components of the habitat, etc. For these reasons, some areas of critical habitat might respond to an exposure when others do not. We also consider how designated critical habitat is likely to respond to any interactions and synergisms between or cumulative effects of pre-existing stressors and proposed stressors.

If the quantity, quality, or availability of the essential features of the area of designated critical habitat are reduced, we ask if those reductions are likely to be sufficient to reduce the value of the designated critical habitat for the conservation of listed species in the action area. In this step of our assessment, we combine information about the contribution of essential features of critical habitat that give the designated area value for the conservation of listed species, to the critical habitat's value to conservation of the listed species in the action area, given the physical, chemical, biotic, and ecological processes that produce and maintain those essential features in the action area. We use the value of those areas of designated critical habitat that occur in the action area as our point of reference for this comparison. For example, if the critical habitat in the action area has limited current value or potential value for the conservation of listed species, then that limited value is our point of reference for our assessment.

If the value of designated critical habitat in an action area is reduced, the final step in our analyses ask if those reductions are likely to be sufficient to reduce the value of the entire critical habitat designation. In this step of our assessment, we combine information about the essential features of critical habitat that are likely to experience changes in quantity, quality, and availability given exposure to an action with information on the physical, chemical, biotic, and ecological processes that produce and maintain those constituent elements in the action area. We use the conservation value of the entire designated critical habitat as our point of reference for this comparison. For example, if the designated critical habitat has limited current value or potential value for the conservation of the listed species that limited value is our point of reference for our assessment.

3.4 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 SFD's 2018 *Biological Evaluation on the Hawaii Shallow-set Pelagic Longline Fishery*, the Western Pacific Fishery Management Council's 2016 *Stock Assessment and Fishery Evaluation Report*, NMFS marine mammal stock assessment reports, available recovery plans for affected species, the 2016 *Report of the Rare*

Events Bycatch Workshop Series (WPFMC 2016), 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*, *Toxnet*, *Science Direct, BioOne, Conference Papers Index, JSTOR, Google Scholar,* and *Web of Science*.

For our literature searches, we used paired combinations of the keywords: "sea turtles," "Pacific," "*Dermochelys*," "*Caretta*," "*Chelonia," "Eretmochelys," Lepidochelys,"* "life history," "population structure," "population trend," "demography," "vital rates," "bycatch," "longline," "mortality," "fishery impacts," "climate change," "temperature-related sexual determination." 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: "ontogentic shifts," "skeletochronology," "stable isotope," "lost years," "age at first reproduction," "sharks," "mantas," "drop off rates," "longline drop offs," "longline catch loss," "depredation of hooked turtles," "cryptic mortality," "unaccounted mortality," "unobservable mortality," "survivorship," "precatch loss," "slipped catch," "circle hooks," "bait," "hot spots," "shark finning," "marine debris," "micro plastic," "plastic ingestion," "debris entanglement," "climate change," "increased sand temperature and sex determination," "habitat loss," "prey availability," "age to maturity," "causal loop analysis," "tag retention," "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. Finally, we relied our own count data from NMFS' observer program of past interactions between the SSLL fishery and sea turtles and other protected species recorded as bycatch from January 2005 through January 2018. These data represent credible and relevant commercial data.

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

3.5 Application of this Approach in this Consultation

NMFS has identified several aspects of the HI SSLL fishery and fishing vessels that represent potential stressors to threatened or endangered species or critical habitat that has been designated for them. Sources of the stressors are primarily vessels and vessel operations, and gear use. The term *stressor* means any physical, chemical, or biological entity that can induce an adverse response. In this case, our assessment is concerned with identifying any stressors that can induce an adverse response from threatened and endangered species and their critical habitat. The specific stressors addressed in this consultation include:

- Interaction with including capture of non-target species, such as listed species, or their prey,
- derelict gear,
- introduction of oily discharges, cardboard, plastics, and other waste into marine waters, and
- vessel emissions.

Our section 7 consultation considers the number of endangered or threatened marine animals that might be exposed to these different stressors, the nature of those exposures, the animal’s probable responses upon being exposed, and the risks those responses might post to individual animals, the populations those individuals represent and the species those population comprise. In total, 49 listed resources – 40 listed species and nine critical habitat designations—occur within the action area identified for this consultation.

3.5.1 Exposure Analyses

We started our analysis by describing where the HI SSLL fishery activities are likely to occur over time. This forms the *Action Area* for this consultation. Within this *Action Area*, we identify 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 the how 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 HI SSLL fishery location and its stressors. Interactions by hooking represent the best data available on the HI SSLL fishery because it has been collected under 100% observer coverage since 2004. In contrast, few data are available to characterize exposure to derelict gear or discharges of waste, although the proposed action is a documented source of these stressors (NMFS unpublished data).

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., insular false killer whale critical habitat, which is wholly contained within the Main Hawaiian Islands longline exclusion zone). We can sub-divide these data into four general categories: potential coastal exposures, potential pelagic exposures, and observed coastal exposures, and observed pelagic exposures. We do not know to what degree listed resources interact with waste or derelict gear from the HI SSLL fishery. We discuss potential exposures to these diffuse sources of stressors like waste and derelict gear briefly, and through our analyses we conclude that there is a low likelihood of exposure so we focus our attention on the primary threat, the observed interactions, and characterizing the effects of those interactions on listed resources.

3.5.2 Response Analyses

The next step in our assessment consists of response analyses, which identify how listed species and their designated critical habitat are likely to respond once exposed to the action's stressors. In these analyses, we review the scientific and commercial data available to determine whether, how, and to what degree listed resources are likely to respond given their exposure to the action's stressors. These analyses distinguish between species that are exposed to a stressor and are not likely adversely affected and those that are exposed to a stressor and are likely to be adversely affected by their exposure and if exposed, whether their responses are immediate, later in time, chronic or acute.

The stressors associated with the HI SSLL fishery produce responses that range from likely exposed and not likely adversely affected – perhaps, opportunistic successful depredation of bait or catch; interactions with predators and prey; accidentally being hooked and then released alive unharmed; hooked and released injured, and death (immediate, or later in time following injury).

Survival from injury is a function of an individual's prior health condition, environmental conditions, severity of injury, indicators of the severity of stress and injury (such as manner of capture, handling and release) and other variables (Swimmer and Gilman 2012).

Figure 3 presents our conceptual model of how we translate an interaction between a sea turtle's exposure and the HI SSLL 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 HI SSLL 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.

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 HI SSLL 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 Pucia and Levins 1985, and others for more on causal loop models). Arrows in these diagrams (see Figure 4 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.

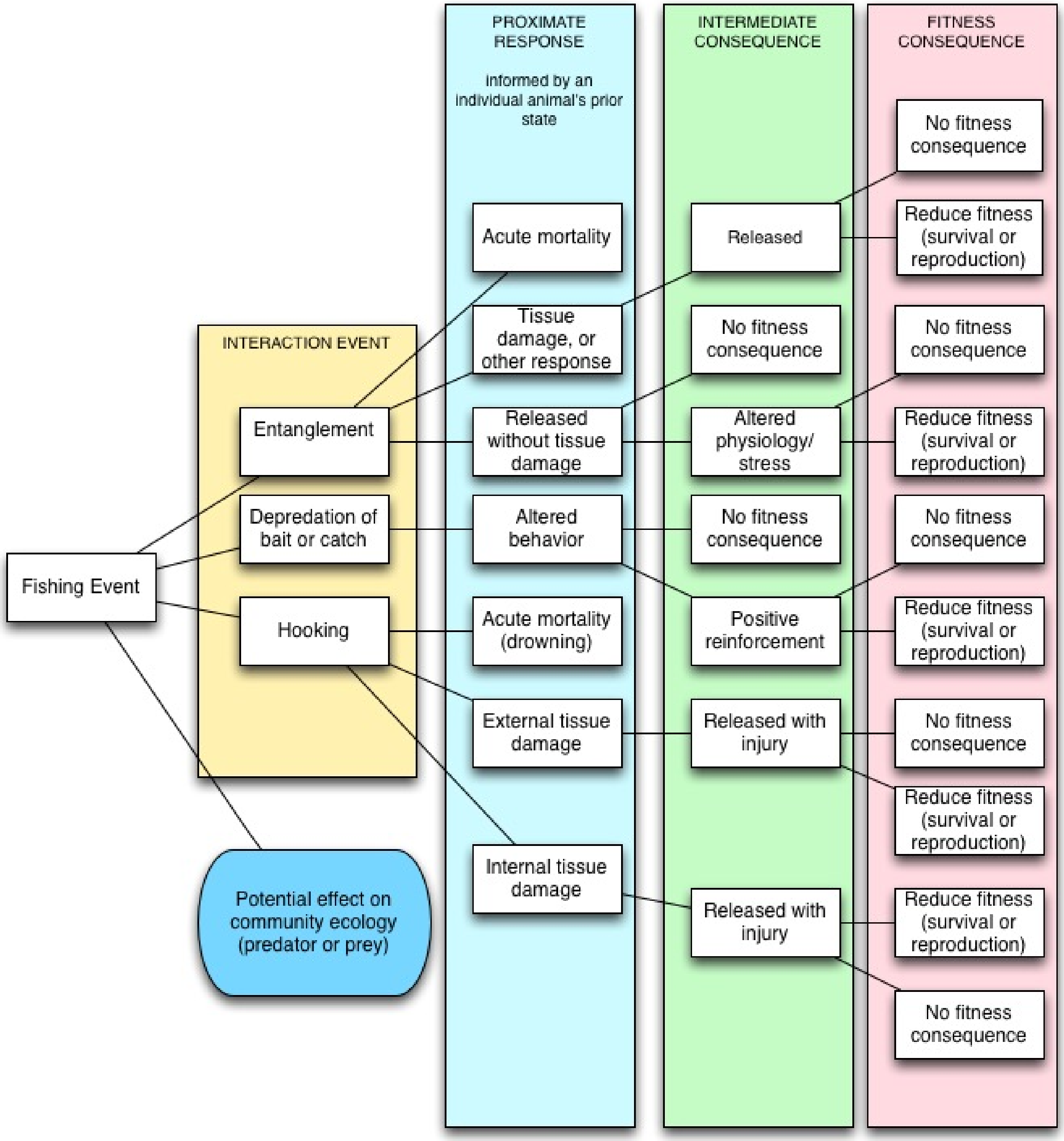


Figure 3. Conceptual model of the potential responses of sea turtles to exposure in the HI SSLL fishery and the pathways between exposure (the interaction event the individual animal has with the fishing gear), responses, and potential fitness consequences to the individual animals

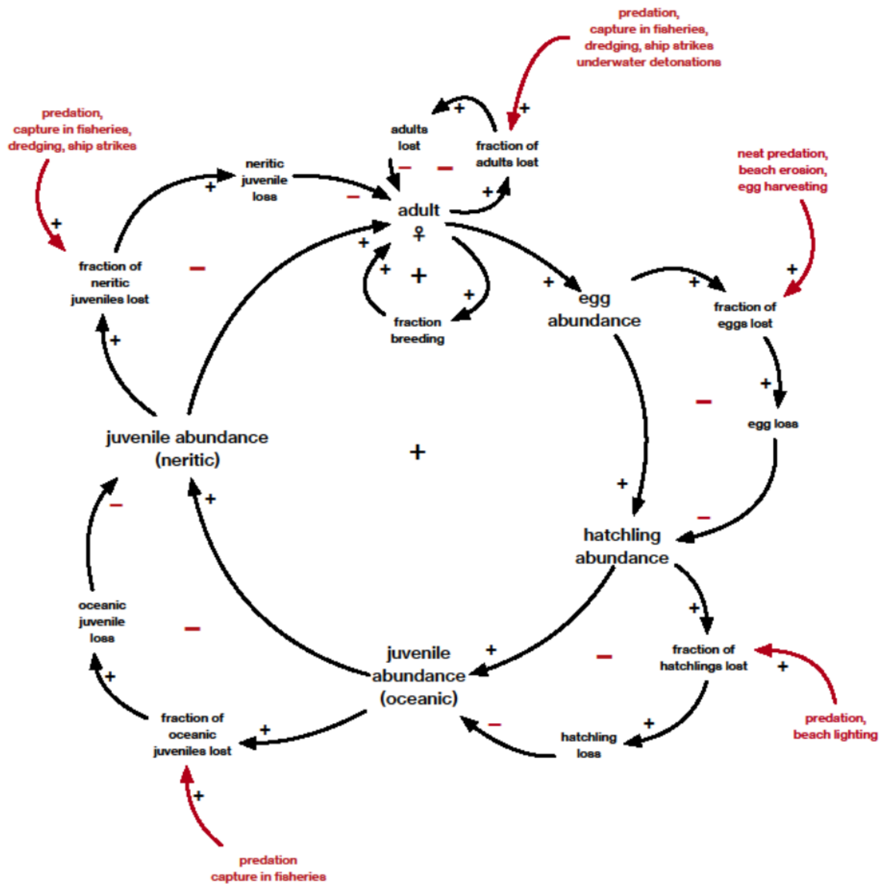
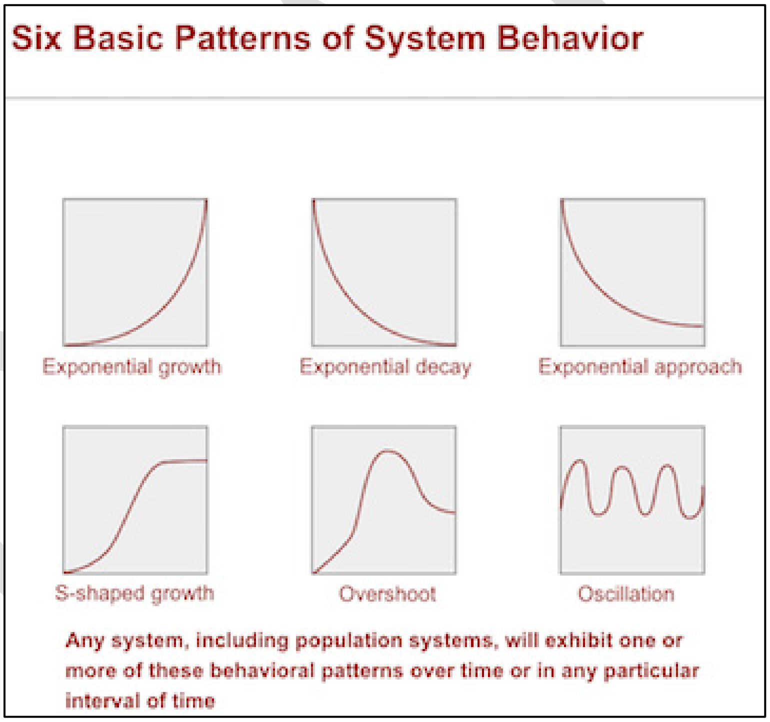


Figure 4. Example causal loop model for loggerhead sea turtles

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 its habitat, 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 areas where identify where they 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 5). 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 5. Six common patterns of system behavior (recreated from Ford 2009)

3.5.3 Simulations using Bayesian state-space population viability analyses for loggerhead and leatherback sea turtles

We modeled, using a Bayesian state-space PVA, the current population status of two species: the endangered leatherback sea turtle and the endangered North Pacific loggerhead sea turtle to examine the long term viability of species. The purpose of the PVA is to bring together current knowledge of the ecology of the species of concern with data like nest counts to forecast the species health and extinction risk (growth and decline) over defined time frames. A Bayesian PVA addresses some of the shortcomings of poor data, imprecise parameter estimates and sampling and process variance in demographic parameters by using posterior probability distributions for parameters of interest. Specifically, we estimated: (a) the time until the population declines to 50%, 25%, and 12.5% of current abundance estimates; (b) the median time for each species to reach those thresholds; and (c) the species' probability of reaching those thresholds (50%, 25%, and 12.5% of current abundance estimates) in 5, 10, 25, 50, 100-year time intervals with associated 95% credible intervals. We also estimated (d) the species' mean log growth rate and variance in mean log growth (with 95% credible intervals) and (e) the species' finite rate of increase (λ), also with 95% credible interval for the estimate.

We used annual nest counts for each species as the foundational data for these analyses because this represents the most reliable long term data set for these species. Counts between years, and beaches (nest areas) are incomplete or missing values during some periods. Therefore, we used statistical time-series analyses to estimate nest counts for months that lack data from leatherback turtles nesting in Indonesia, and used a parametric model to estimate missing counts in a time series. For loggerhead sea turtles, because the data was provided in aggregate form we assumed that there was consistent monitoring effort in all months.

Ideally, we would have key demographic parameters on each of the species to support robust analyses of abundance and trends, such as: age at maturity, generation length, survivorship across life stages, adult and hatchling sex ratios, clutch frequency, nesting success, remigration intervals and conversion factors for each subpopulation that comprises the population. However, comprehensive population-specific data sets are not available for either species. Nest counts provide an index of female abundance because females do not return to nest every year, but also because we have data for only a portion of the subpopulations that comprise the population. Even within a subpopulation for which we had data, missing values during a nesting season needed to be estimated to fill in gaps. No data are available to provide an index of abundance for other age classes. Due to this data limitation, estimating *r* from this data does not represent the true population growth rate for the population of interest. Importantly, monitoring effort among and within beaches, months, and years, can vary widely and complicate comparison of the data and development of robust estimates of population sizes and trends (SWOT Scientific Advisory Board 2011).

3.5.4 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/m2 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. 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.

Table 2. Projections for certain climate parameters under Representative Concentration Pathway 8.5 (values from IPCC 2015).

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

Given the limited data available on sea turtle populations, and other listed species like 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, we've learned a few key important lessons: the climate based model fixed age (lag) is unrealistic given variability in ages at sexual maturity for each 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; Briscoe 2016a, 2016b; Jones 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 the exposure, response, and risk analyses. In the *Status of Listed Resources* and the *Environmental Baseline* we present an extensive review of the best scientific and commercial data available to describe how the listed species and its designated critical habitat is affected by climate change—the status of individuals, and its demographically independent units (subpopulations, populations), and critical habitat in the action area and range wide.

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

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

3.5.5 Cumulative Effects

Cumulative effects as defined by Section 7 (50 CFR 402.02) as those effects of future state or private activities, not involving Federal activities, that are reasonably certain to occur in an action area. In contrast, under NEPA cumulative effects are the incremental impact of the action when added to other past, present, and reasonably foreseeable future actions, regardless of what agency or person undertakes such actions. To distinguish between the two, herein we call the former *Cumulative Effects* and the latter *Cumulative Impacts (in the sense of NEPA).*

For our *Cumulative Effects* analysis for this consultation we will look to state and regional maps and land use plans, or other similar sources of information that will indicate plans for the harbor areas where the fishery vessels are moored, or other changes to state fisheries and state waters, to the extent practicable. We will document our searches and consider the effect of any relevant actions in the *Cumulative Effects* section of the biological opinion on the HI SSLL fishery. For activities farther to the center of the action area (pelagic areas) based on our prior experience, we expect most of the private or non-federal activities would amount to foreign fisheries, which is considered under the *Environmental Baseline* section of this opinion and would be expected to continue into the future.

In contrast, several organizations have argued that several of NMFS' biological opinions on such activities as the US Navy's use of active sonar failed to consider cumulative impacts (in the sense of the NEPA term) of active sonar on the ocean environment and its organisms, particularly endangered and threatened species and critical habitat that has been designated for them (see for example NRDC 2007 and Ocean Mammal Institute 2007). The effects analyses of biological opinions, however, consider the "impacts" on listed species and designated critical habitat that result from the incremental impact of an action by identifying natural and anthropogenic stressors that affect endangered and threatened species throughout their range (the *Status of Listed Resources*), and within the action area (the Environmental Baseline, which articulate the pre-existing impacts of activities that occur in an action area including the past, contemporaneous and future impacts of those activities). We assess the effects of a proposed action by adding their direct and indirect effects to the impacts of the activities that we identify in an *Environmental Baseline* (50 CFR 402.02), in light of the impacts of the status of the listed species and designated critical habitat throughout their range; as a result, our effects analyses are equivalent to those contained in the "cumulative impact" sections of NEPA documents.

3.5.6 Risk Analyses

The final steps of our analyses--establishing the risks the responses pose endangered and threatened species or designated critical habitat—normally begin by identifying the probable risks actions pose to listed individuals that are likely to be exposed to an action's effects. Our analyses then integrate those individuals' risks to identify consequences to the viability of the demographically independent units (subpopulations or demes) and populations that those individuals represent. Our risk analyses conclude by determining the consequences of changing the viability of the demographically-independent units (subpopulations or demes) and populations, and the species those populations comprise.

We measure risks to listed individuals using the concept of current or expected future reproductive success which, as we described in the preceding sections, integrate survival and longevity with current and future reproductive success. In particular, we examine the scientific and commercial data available to determine if an individual‘s probable response to stressors produced by an action would reasonably be expected to reduce the individual‘s current or expected future reproductive success by increasing the individual‘s likelihood of dying prematurely, having reduced longevity, increasing the age at which individuals become reproductively mature, reducing the age at which individuals stop reproducing, reducing the number of live births an individual produces during any reproductive bout, reducing the number of times an individual is likely to reproduce over the reproductive lifespan (in animals that reproduce multiple times), or causing an individual‘s progeny to experience any of these phenomena. When individual plants or animals would be expected to experience reductions in their current or expected future reproductive success, we would also expect those reductions to also reduce the abundance, reproduction rates, or growth rates (or increase variance in one or more of these rates) of the populations those individuals represent (see Stearns 1992). If we conclude that listed plants or animals are *not* likely to experience reductions in their current or expected future reproductive success, we would conclude our assessment.

If we conclude that listed plants or animals are likely to experience reductions in their current or expected future reproductive success, we would integrate those individuals risks to determine if the number of individuals that experience reduced fitness (or the magnitude of any reductions) is likely to be sufficient to reduce the viability of the populations those individuals represent (measured using changes in the populations‘ abundance, reproduction, spatial structure and connectivity, growth rates, or variance in these measures to make inferences about a population‘s probability of becoming demographically, ecologically, or genetically extinct in 10, 25, 50, or 100 years). For this step of our analyses, we would rely on the population‘s base condition (established in the *Environmental Baseline* and *Status of Listed Resources* sections of this opinion) as our point of reference.

Our risk analyses normally conclude by determining whether changes in the viability of one or more populations are or are not likely to be sufficient to reduce the viability of the species (measured using probability of demographic, ecological, or genetic extinction in 10, 25, 50, or 100 years) those populations comprise. 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 as well as a suite of population viability models. Our assessment is designed to establish that a decline, collapse, or extinction of an endangered or threatened species is not likely to occur; we do not conduct these analyses to establish that such an outcome is likely to occur.

In our assessment, we rely on quantitative and qualitative methods for evaluating the action's effects on listed species and their designated critical habitat. For instance we combine information on the risks to individual animals from the proposed action, and integrate those at higher demographic units for each species, within the context of the *Environmental Baseline* and the *Status of Listed Resources* to explain the relationship between the proposed action and the consequences of changing the viability of the subpopulations and populations, and the species those populations comprise.

Importantly, available nesting count data provides one indicator of the population's performance; however, its limitations are significant and as such, in contrast to the 2012 biological opinion on this same fishery, we are not using this data to build a PVA-based take estimate for use in our jeopardy analysis. In contrast we use available data to characterize the species' status, and recognize the limitations of the data for further extrapolation (e.g., for use in a "take" model because the available does not contain sufficient resolution or precision to assess the effect of the action to the species as a whole, nor does the available data allow distinction of the effect of take on distinct populations or even unique life history strategies). In this assessment, we try to incorporate information to characterize and understand the demographic performance the demographically-independent units that comprise the species in our jeopardy analyses (ideally, we would incorporate all of them in our analysis) because often with imperiled species stronger subpopulations will mask the stability or extinction risk of the smaller subpopulations. We cannot assume that what is true of a whole species is also true of their parts (populations, subpopulations). Thus, two of the primary variables that inform the status of the species are (a) the number of demographically-independent units (e.g., populations, subpopulations) that comprise a species, and (b) the demographic performance of these demographically independent units over time (Lomolino and Channel 1998; McElhaney et al. 2000).

Reproductive isolation forms a continuum that is not smooth; it is a continuous process, yet there exists for many species a biological basis for designating demes—biologically discrete, at some level, interbreeding units—which may be smaller than the population and species (or DPS) because there is a degree of independence that can be important to the persistence of the species. For the leatherback sea turtle, while the species is listed globally, the West Pacific Ocean population is recognized as reproductively isolated (Dutton 2007; NMFS 2013 status update). In this assessment, we examine the action's effects on the individuals captured within the HI SSLL fishery, recognizing that these are animals nest on many small (isolated) beaches in areas of Papua New Guinea, Vanuatu, Papua Barat in Indonesia, and the Solomon Islands (Dutton 2007). Although West Pacific Ocean leatherback sea turtle mtDNA studies indicate the population is genetically homogenous, the population is comprised of smaller, demographically distinct units (Dutton 2007; Dutton and Shanker in Spotila and Tomillo 2015). Microsatellite investigations and more sampling, however, may reveal finer population substructuring in this metapopulation.

The data available on nesting leatherback sea turtle females comes from Jamursba-Medi and Wermon nesting aggregations, which represents only a portion of the West Pacific Ocean leatherback sea turtle population. If we were to use this data to in our effects analysis and create a take-based PVA using the same data set, we would create an appearance of precision that fails to consider the unique circumstances of other subpopulations and the stressors that affect their persistence. Therefore, our evaluation is designed to locate, combine, and rely-on several different data sources, combined into a meta-analysis that considers the number of demographically-independent units that comprise the species, their performance, and characterizes the risk the proposed action poses the individual listed species, the populations they represent, and the species as listed, whether we have sufficient data to model their population viability, or not.

Importantly, this change in procedure between the 2012 biological opinion and this assessment are justified because it also recognizes that 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). Other important factors:

- abundance—because small populations are often at greater risk than larger populations
- growth rate—a measure of performance, closely tracks with abundance
- diversity—spatial, temporal, genetic, environmental, within and among populations (from McElhaney et al. 2000).

Because other species in our assessment lack robust demographic data, our evaluation for these species also relies upon a mix of qualitative and quantitative analytical methods to examine these factors that influence a species extinction risk.

3.5.6.1 Method for Estimating Probable Future Interactions (Capture, Hookings, Entanglement) of Seven Protected Species

We employed Bayesian inference techniques to estimate the annual interactions of the following species: loggerhead sea turtles, leatherback sea turtles, green sea turtles, olive ridley sea turtles, Guadalupe fur seals, oceanic white tip sharks, and giant manta rays (McCracken 2018). We used this method because we expect it is adequate to describe the situation of interest. Specifically, the model provides a joint probability distribution of the variable of interest, interaction (aka take level), and the unknown parameters of the data distribution (probability distribution function that is assumed to have generated the data). This joint density can be written as a product of two distributions that are commonly referred to as the prior distribution and the data distribution. The intent of the prior distribution is to capture our knowledge or beliefs about these parameters, without reference to the data. Our method consisted of the following steps:

1. Segment the year to address the times that the fishery was closed
2. Using Bayesian inferences to estimate the expected interactions

Segment the year to address the times that the fishery was closed

Based on the dates the fishery was closed; March 20, 2006 and November 18, 2011, the calendar year was split into 3 segments: (1) 1 January - 20 March, (2) 21 March - 18 November, and (3) 20 November – 31 December. For each species, the take levels within each segment were examined across years and a decision was made between four options:

- Option 1: Estimate the ATL for each period defined by the dates of the 3 segments.

- Option 2: Exclude year 2006 and estimate the ATL for two periods. The first period is defined by combining segments 1 and 2, and the second period is defined by segment 3.
- Option 3: Exclude year 2011 and estimate the ATL for two periods. The first period is defined by segment 1, and the second period is defined by combining segments 2 and 3.
- Option 4: Exclude years 2006 and 2011 and estimate the ATL using the annual take levels for the remaining years.

For the rays and oceanic whitetip shark, Options 3 and 4 were not considered as year 2017 would also need to be excluded since the take level for the third segment in 2017 was not available.

If the calendar year is broken into periods, the take levels for the different periods are treated as if they are independent observations; that is, they are assumed to be uncorrelated with one another even for successive time periods. If years 2006 and 2011 are excluded, information is lost. For loggerheads and leatherbacks, the year the fishery closed because their take limit was reached is a year with a larger take level. Although what would have happened if the fishery remained open is unknown, the take level would have been at least the value of the relevant take limit. In summary, there is a trade-off between breaking the year into periods and deriving each period's ATL at the cost of complexity and excluding years 2006 and 2011 and estimating the annual ATL at the cost of losing information.

As the fishery was closed in the first quarter of 2006, this year was only included if a species showed a propensity for relatively higher take levels in the first segment of the year, or the first segment in 2006 had an unusual number of takes (higher or lower than other years). In 2011 the fishery was closed near the end of the year, and for many species, one would expect few, if any, bycatch events during the last segment of the year.

Use of Bayesian inference to estimate annual interactions of seven protected species

We use Bayesian inference techniques to estimate annual interactions of seven protected species, loggerhead sea turtles, leatherback sea turtles, green sea turtles, olive ridley sea turtles, Guadalupe fur seals, oceanic white tip sharks, and giant manta rays, as an anticipated probability distribution. We assumed that the underlying process that generates the interactions does not change, and that the random values must come up randomly, independently across years, with a single fixed probability distribution. Since interactions are count data, all outcomes will be nonnegative integers; hence, the probability distribution is a discrete distribution. This discrete distribution can be thought of as the relative frequency (probability) of each possible outcome from a long-run of random interactions observations.

We use the COM-Poisson (Conway-Maxwell-Poisson or CMP) distribution (Conway and Maxwell 1962) model that allows for count data being both overdispersed and underdispersed to derive the posterior annual interaction levels of interest. We expect the mean of the posterior annual interaction level would be greater than the average annual interaction level for the historical data since the annual interaction level assumes that the fishery does not close during the year, which happened in 2006 and 2011 (McCracken 2018). This approach allowed us to estimate the likely rate of annual interactions for each of these three species, and then based on this number derive the 2 and 3 year estimated interaction rate for each loggerhead sea turtles, leatherback sea turtles, green sea turtles, olive ridley sea turtles, Guadalupe fur seals, oceanic white tip sharks, and giant manta rays.

3.5.6.2 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. take level) is within a specified range of values (confidence interval) and, more specifically, to describe the uncertainty with the statistic. Bayesian predictions were used to estimate the credible interval, or the probability that the true value (i.e. take level) is within a specified range of values (credible interval).

Section 7(a)(2) of the ESA requires federal agencies to *insure* that any action they authorize, fund, or carry out is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of critical habitat of such. As part of this process, agencies are required to base their conclusion on the best scientific and commercial data available (16 U.S.C. 1539(a)(2)). 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 insuring that the action satisfies the requirements of section 7(a)(2) of the ESA, we chose the 95% confidence level and 95% credible intervals when estimating the take level. By selecting a 95% level we can be confident that 95% of the confidence intervals produced from the data will contain the true population parameter. In other words, the calculated 95% confidence interval for the mean number of animals captured or killed will contain the true value 95% of the time. By using a 95% credible interval there is a 95% probability that, given the data, the true population parameter is contained within the credible interval. In terms of take, this means that there is a 95% probability that the true number of animals captured or killed is within the credible interval.

3.6 Action Area

The action area includes all areas where HI SSLL fishery vessels operate, including transiting and fishing. Vessels fishing from 2009-2016 fishing occurred between 180°- 125° W and 17°-45° N (Figure 6). The action area also includes waters prohibited by Federal regulations to longline fishing where vessels travel to and from fishing grounds, including the EEZ seaward of California (50 CFR 660) and the Main Hawaiian Islands (MHI) Longline Fishing Prohibited Area, which prohibits longline fishing year-round (50 CFR 665). We also analyze transit as part of the action which includes transit to the ports of Honolulu, Long Beach, San Francisco, and San Diego.

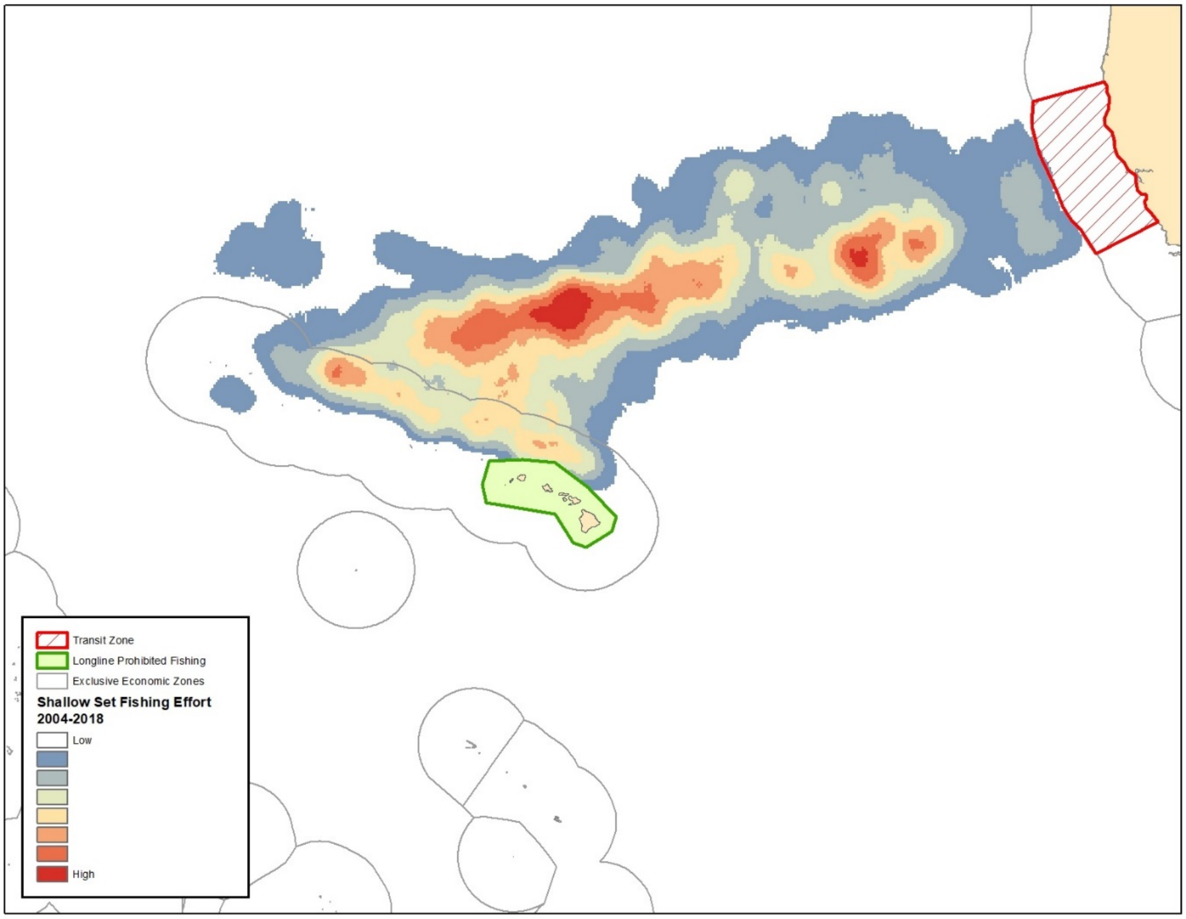


Figure 6. Action area with location of shallow sets made by the Hawaii longline fishery from 2004– 2018.

4 STATUS OF LISTED RESOURCES

NMFS has determined that the action that NMFS SFD proposes to authorize, the HI SSLL fishery, may affect the threatened and endangered species and their designated critical habitat 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
Loggerhead sea turtle, North Pacific DPS	*Caretta caretta*	Endangered
Green sea turtle, Eastern Pacific DPS	*Chelonia mydas*	Threatened
Green sea turtle, Central North Pacific DPS		Threatened
Green sea turtle, East Indian - West Pacific DPS		Threatened
Green sea turtle, Central West Pacific DPS		Endangered

Green sea turtle, Southwest Pacific DPS		Threatened
Green sea turtle, Central South Pacific DPS		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		
Guadalupe fur seal	*Arctocephalus townsendi*	Endangered
Hawaiian monk seal	*Neomonachus schauinslandi*	Endangered
MHI insular false killer whale DPS	*Pseudorca crassidens*	Endangered
Humpback whale, Central America DPS	*Megaptera novaeangliae*	Endangered
Humpback whale, Mexico DPS		Threatened
Fin whale	*Balaenoptera physalus*	Endangered
Blue whale	*Balaenoptera musculus*	Endangered
North Pacific right whale	*Eubalaena japonica*	Endangered
Sei whale	*Balaenoptera borealis*	Endangered
Sperm whale	*Physeter macrocephalus*	Endangered
Killer whale, southern resident DPS	*Orcinus orca*	Endangered
FISH		
Scalloped hammerhead shark, Eastern Pacific DPS	*Sphyrna lewini*	Threatened
Oceanic whitetip shark	*Carcharhinus longimanus*	Threatened
Giant manta ray	*Manta birostris*	Threatened
Central California coast coho salmon	*Oncorhynchus kisutch*	Endangered
Central Valley spring-run Chinook salmon	*Oncorhynchus tshawytscha*	Threatened
Sacramento River winter-run Chinook salmon	*Oncorhynchus tshawytscha*	Endangered
Central California coast steelhead	*Oncorhynchus mykiss*	Threatened
California Central Valley steelhead	*Oncorhynchus mykiss*	Threatened
California coast steelhead	*Oncorhynchus mykiss*	Endangered
North American green sturgeon, Southern DPS	*Acipenser medirostris*	Threatened
MARINE INVERTEBRATES		
Black abalone	*Haliotis cracherodii*	Endangered
White abalone	*Haliotis sorenseni*	Endangered
DESIGNATED CRITICAL HABITAT		
Leatherback sea turtle critical habitat		
Hawaiian monk seal critical habitat		
MHI false killer whale critical habitat		
Steller sea lion critical habitat		
Central California coast coho salmon critical habitat		
Sacramento River winter run Chinook salmon critical habitat		
California coast steelhead critical habitat		
N. Am. green sturgeon critical habitat		
Black abalone critical habitat		

4.1 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 species or critical habitat that are not likely to be adversely affected by the HI SSLL fishery. The first criterion was exposure or some reasonable expectation of a co-occurrence between one or more potential stressors association with the HI SSLL fishery and a particular listed species or designated critical habitat. If we conclude that a listed species or designated critical habitat is not likely to be exposed to the HI SSLL fishery, we must also conclude that the species and critical habitat is not likely to be adversely affected by those activities. The second criterion is the 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 HI SSLL fishery.

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., insular false killer whale critical habitat, which is wholly contained within the Main Hawaiian Islands longline exclusion zone). Next we reviewed whether we had data (observations) on the species exposure or the exposure of their designated critical habitat, or reasoned information that exposure could occur (potential) to one or more of the action’s stressors: fishery interactions (e.g., vessel noise, vessel collision, hooking or entanglement in gear); vessel waste, discharge, and emissions. We ended up with four general exposure scenarios (Table 4) to which a species or designated critical habitat 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 designated critical habitat is unique, and may represent different combinations of stressors of a different magnitude or exposure to those stressors. Given the nature of vessel waste, discharge and emissions, these stressors have the potential to affect all exposure profiles. Even so, some patterns emerged, specifically when we focused on the primary stressors within each profile.

For example, the domestic longline vessel exclusion zone (57 FR 7661) around the MHI (Figure 7) and the state of California’s prohibition of pelagic longline fishing within its EEZ (Highly Migratory Species Fisheries Management Plan), restrict longline vessels from fishing in coastal waters. Therefore, the species listed in the potential coastal exposure would not be exposed to hooking or entanglement from fishing gear; however, they would likely be exposed to vessel transit and vessel noise. The unoccupied observed coastal exposure profile reflects the number of documented interactions (vessel collisions) with coastal species. There were no accounts in the observer database of any observed vessel collisions with any species, much less the species within the coastal exposures.

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

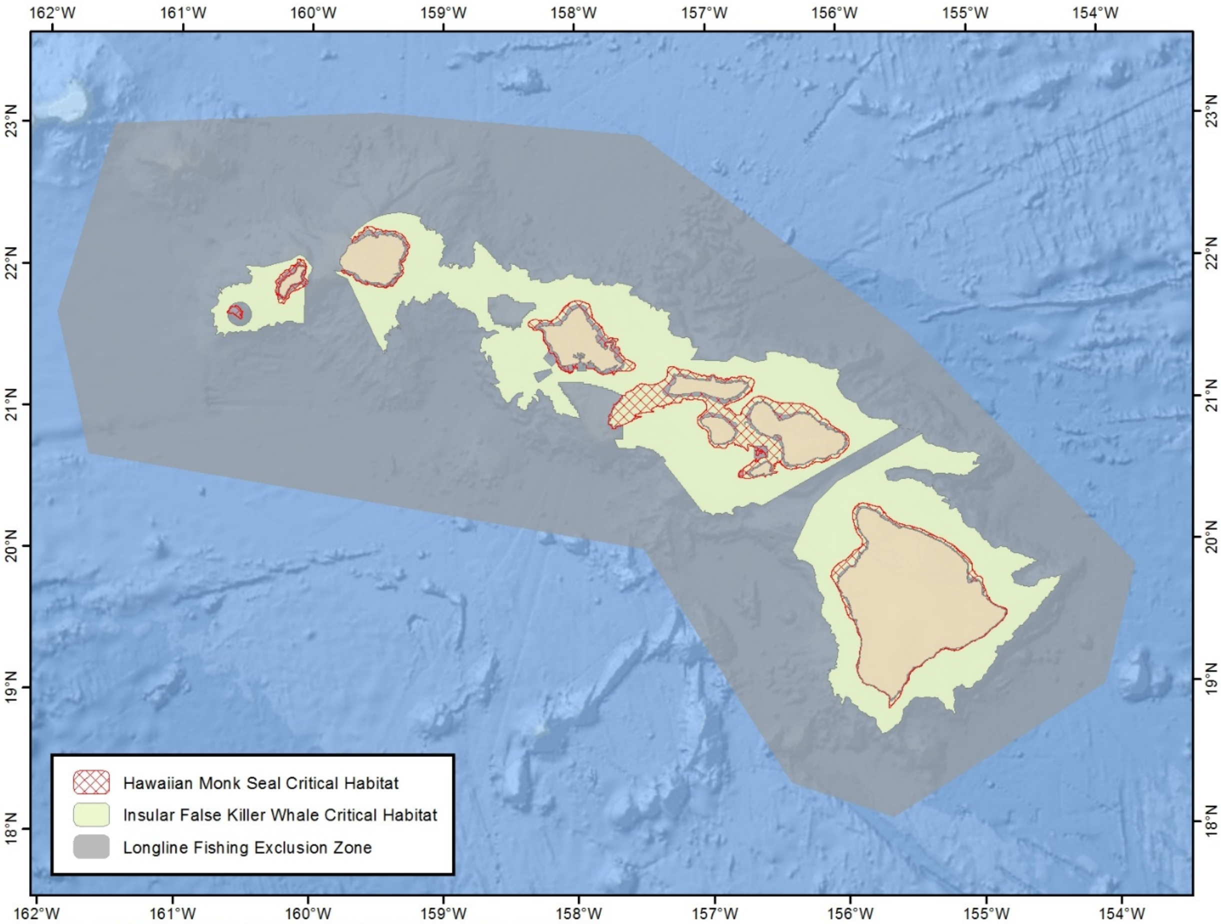
This subsection also summarizes the results of our evaluations of the probability of a response given exposure, which considers susceptibility. Here we briefly describe the species and stressors that are not likely to adversely affect listed species or their designated critical habitat, and our reasoning for this conclusion. Based on our evaluation, the following species are not likely to be adversely affected by the proposed action: hawksbill sea turtle; fin whale; sperm whale; humpback whale, Mexico DPS; humpback whale, Central America DPS; blue whale; sei whale; North Pacific right whale, killer whale, southern resident DPS; Hawaiian monk seal; scalloped hammerhead shark, Eastern Pacific DPS; Central California coast coho salmon; Central Valley spring-run Chinook salmon; Sacramento River winter-run Chinook salmon; Central California coast steelhead; California coast steelhead; North American green sturgeon, Southern DPS; black abalone, and white abalone. Additionally, based on our evaluation of the proposed action, the following designated critical habitat is not likely to be adversely affect by the proposed action: leatherback sea turtle critical habitat, Hawaiian monk seal critical habitat, MHI false killer whale critical habitat, Steller sea lion critical habitat, Central California coast coho salmon critical habitat, Sacramento River winter run Chinook salmon critical habitat, California coast steelhead critical habitat, N. Am. green sturgeon critical habitat, black abalone critical habitat.

In addition, vessel noise, wastes and discharges, and vessel emissions would have discountable effects on leatherback sea turtles, loggerhead sea turtles, green sea turtles, olive ridley sea turtles, giant manta rays, oceanic whitetip sharks, and Guadalupe fur seals. However, because interactions with longline gear are observed for all of these species (that is, these species fall into observed pelagic exposures category) we discuss these species in greater detail later in this biological opinion (see section 6, *Effects of the Action*).

Table 4. General location (coastal or pelagic waters) of likely exposure of species and critical habitat to proposed action's stressors. *These species fell within two exposure profiles.

Observed Pelagic Exposure	**Potential Pelagic Exposure**
Fin whale	Humpback whale, Mexico DPS
Sperm whale	Blue whale
Loggerhead sea turtle	N. Pacific right whale
Leatherback sea turtle*	Sei whale
Green sea turtle*	Scalloped hammerhead, Eastern Pacific DPS
Olive Ridley sea turtle	Hawksbill sea turtle*
Guadalupe fur seal*	
Oceanic white-tip shark	
Giant manta ray*	
Observed Coastal Exposure	**Potential Coastal Exposure**
none	Hawksbill sea turtle*
	Hawaiian monk seal
	Hawaiian monk seal critical habitat
	MHI Insular false killer whale DPS
	MHI Insular false killer whale DPS critical habitat
	Killer whale, southern resident DPS
	Stellar sea lion critical habitat

	Leatherback sea turtle*
	Leatherback sea turtle critical habitat
	Green sea turtle*
	Guadalupe fur seal*
	Giant manta ray*
`	Central California coast coho salmon
	Central California coast coho salmon critical habitat
	Central Valley spring-run Chinook salmon
	Sacramento River winter-run Chinook salmon
	Sacramento River winter-run Chinook salmon critical habitat
	California coast steelhead
	California coast steelhead critical habitat
	California Central Valley steelhead
	Southern California coast steelhead
	North American green sturgeon, Southern DPS
	North Am, green sturgeon, Southern DPS, critical habitat
	Black abalone
	Black abalone critical habitat
	White abalone



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

Figure 7. The longline fishing exclusion zone around the Main Hawaiian Islands

4.1.1 Potential Coastal Exposure

By far the largest category, 13 species and eight critical habitat designations fall into the category of potential coastal exposures. This includes, hawksbill sea turtles, Hawaiian monk seals and their critical habitat, MHI Insular false killer whales and their critical habitat, southern resident killer whales, Stellar sea lion critical habitat, Central California coast coho salmon and their critical habitat, Central Valley spring-run chinook salmon, Sacramento River winter-run Chinook salmon and their critical habitat, California coast steelhead and their critical habitat, California Central Valley steelhead, southern California coast steelhead, southern DPS of North American green sturgeon and their critical habitat, black abalone and their critical habitat and white abalone. As mentioned above, fishing is restricted in the coastal areas. As such, these species are exposed to all or any of the stressors emanating from vessel transits (e.g., vessel collision, vessel noise, vessel waste, discharge and emissions). These stressors may result in immediate exposures, or exposures later in time (e.g., interaction with derelict gear or vessel waste). However, NMFS has determined that for all the species in the category of potential coastal exposures, effects from all or any stressors related to vessel transiting to be highly unlikely and therefore discountable. For detailed information, see the applicable stressor discussion below.

4.1.2 Observed Coastal Exposure

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

4.1.3 Potential Pelagic Exposure

Six species fall into the potential pelagic exposure category: Eastern Pacific DPS scalloped hammerhead sharks, Mexico DPS humpback whales, blue whales, North Pacific right whales and sei whales, and hawksbill sea turtles. Although, we expect the exposure of the hawksbill sea turtle to be primarily in coastal areas, based on its occurrence in oceanic waters, we have included it in the pelagic profile as well. These species could be exposed to all or any of the stressors emanating from the proposed action: fishery interactions (e.g., vessel noise, vessel collision, hooking or entanglement in gear); and vessel waste, discharge, or emissions. These stressors may result in immediate exposures, or exposures later in time (e.g., interaction with derelict gear or vessel waste).

Blue whales, North Pacific right whales and sei whales feed nearly exclusively on krill and would therefore not be actively depredating on bait fish or captured fish hooked on longline gear. The densities for these species is extremely low and there have been no known interactions between these whales and the HI SSLL fishery. Likewise, the density of hawksbill sea turtles in the action area is also low and there are no know interactions with the HI SSLL fishery.

The HI SSLL has observed four interactions with the humpback whale, although, these interactions were attributed to the Hawaii DPS, which is not listed. Due to the northern migrations of the Central America and Mexico DPS of humpback whales there is a low probability that some individual animals could be exposed to the HI SSLL fishery; however, most animals from these two DPSs are expected to migrate close to the coast (within the EEZ) and utilize the California/Oregon (OR/CA) feeding area rather than the higher latitude feeding areas (Wade et al. 2016). Given the Hawaii DPS population is approximately 3.5 times greater than that of the Mexico DPS, the probability that the humpback whale is in Hawaii DPS is significantly higher than that of a humpback whale in the Mexico DPS.

Similarly, the HI SSLL has observed 15 interactions with scalloped and unidentified hammerhead sharks. The seven hammerheads could be identified to species were from the non-listed Central Pacific DPS, and eight that were unidentified hammerheads, were captured in the vicinity of the non-listed central Pacific DPS. There has never been an observed interaction with a scalloped hammerhead shark in the endangered Eastern Pacific DPS. This is likely due to the species more coastal nature and its lack of long distance migrations across deep pelagic waters.

None of the species listed above have been hooked or entangled in HI SSLL gear. Therefore, NMFS has determined that for all the species in the potential pelagic exposure profile effects from all or any stressors related to fishery interactions to be highly unlikely and therefore discountable. For more information, see the applicable stressor discussion below.

4.1.4 Observed Pelagic Exposure

Nine species fall into the observed pelagic exposure category: fin whales, sperm whales, loggerhead sea turtles, leatherback sea turtles, green sea turtles, olive ridley sea turtles, Guadalupe fur seals, oceanic white-tip sharks, and giant manta rays. These species could be exposed to all or any of the stressors emanating from the proposed action: fishery interactions (e.g., vessel noise, vessel collision, hooking or entanglement in gear); and vessel waste, discharge, or emissions. These stressors may result in immediate exposures, or exposures later in time (e.g., interaction with derelict gear or vessel waste). As previously described, there are documented observations of interactions with fishing gear resulting in hooking and/or entanglement with the species listed in the observed pelagic exposure profile. Based on the analysis below, NMFS has determined for the fin whale and sperm whale listed in the observed pelagic exposure profile, effects from all or any stressors related to fishery interactions to be highly unlikely and therefore discountable.

Fin whales do not depredate on bait fish or captured fish hooked longline gear, however, they can become entangled or foul hooked in fishing gear and may break through or carry gear away. From 1994 to February, 2015, there were no observed or reported interactions with fin whales in the shallow or deep-set fishery. The February 2015 interaction is unique and extremely rare. The observer aboard the shallow-set vessel documented the main line becoming entangled in a young whale's mouth (no branch hooks or branch lines were involved). The crew was able to release the whale within approximately 5 minutes and watched it swim away with no gear attached and with only superficial wounds. NMFS determined that this injury was non-serious under the MMPA (Bradford and Forney 2017). Since the HI SSSLL fishery re-opened in 2004 with 100% observer coverage, there has been only one interaction with a fin whale in nearly 17,244 sets, with an interaction rate of 0.000058 fin whales per set, so interactions are exceptionally uncommon.

Likewise, sperm whales are present in the action area; however, interactions between the Hawaii-based longline fishery and sperm whales are rare and unpredictable events. Since 1994, there have been three observed interactions between sperm whales and the entire Hawaii longline fleet. Prior to the separation of the management of the longline fisheries, there was an interaction in 1999 with a vessel that was targeting swordfish, and one in 2002 with an experimental fishery that was testing sea turtle mitigation gear similar to what is used in the HI SSLL fishery now. The 2002 interaction occurred on a control set and the sperm whale was entangled in the mainline. The mainline was cut and the animal escaped with no line attached (Boggs 2002). Sperm whales have been recorded depredating on catch in the longline sable fishery in the Gulf of Alaska. However, no incidents of depredation have been recorded in the Hawaii-based longline fishery and since the HI SSLL fishery re-opened in 2004 with 100% observer coverage, there have not been any interactions with sperm whales.

Finally, observed interactions between the HI SSLL fishery and loggerhead sea turtles, leatherback sea turtles, green sea turtles, olive ridley sea turtles, Guadalupe fur seals, oceanic white-tip sharks, and giant manta rays are covered in detail in section 6 of this biological opinion, *Effects of the Action*.

4.1.5 Collision with vessels

The proposed action would expose all ESA-listed marine species under NMFS' jurisdiction found in both the coastal and pelagic exposure categories to the risk of collision with vessels. Depending on the severity of contact, the collision could cause injuries including bruising, broken bones or carapaces, lacerations, or even death in severe cases.

Propeller strikes from large vessels are likely to be fatal, while propeller injuries from small boats may cause disfigurement of the dorsal fin or other parts of the body without killing the whale outright (Wells et al. 2008). Vanderlaan and Taggart (2007) report that the severity of injury to larger whale species is directly related to vessel speed. They found that the probability of lethal injury increased from 21% for vessels traveling at 8.6 knots, to over 79% for vessels moving at 15 knots or more. We assume collisions at higher speeds would result in more severe injuries for all animals. Slow-moving animals, or animals that associate with vessels, i.e., to ride the bow or depredate catch on fishing gear being retrieved, are likely the most susceptible to ship strikes. Most of the ESA-listed whales in the action area are baleen whales and will be solitary and widely scattered throughout the action area. The sperm whale has been known to interact with Alaska longline fisheries, however, there have been no documented depredation events in the HI SSLL fishery.

Although more commonly observed in large whales, vessel strikes also have potential to kill or injure smaller cetaceans. False killer whales commonly travel in pods and are known to approach vessels. False killer whales in waters surrounding Hawaii (belonging to both insular and pelagic stocks) are known to ride the bow or stern wake of vessels and may come into proximity of propellers (Baird pers. comm.). However, no ship-strike related injuries or deaths of false killer whales have been documented in Hawaiian waters, but Baird (2009) reported a fresh head wound on one individual from the insular population photographed off Oahu in September 2009 that may have been caused by a propeller strike. The density of MHI IFKW is expected to be low since the transit routes are in low-use areas according to Baird et al. (2012) who describe the north side of the island of Hawaii, a broad area extending from north of Maui to northwest of Molokai, and a small area to the southwest of Lanai as high-use areas preferred by MHI insular false killer whale.

Studies on scalloped hammerhead sharks have shown that they have well-developed electrosensory systems and vision (Kajiura 2001) that presumably enables them to detect activity in the water at a distance and to quickly move away from slow-moving vessels. While specific studies have not been conducted for oceanic white tips or giant manta rays for vessel avoidance, they are elasmobranchs and are highly mobile species. The lateral line in manta rays is poorly understood, however they also have a suite of other biological functions which are considered highly sophisticated sensory systems (Bleckmann and Hoffmann 1999; Deakos 2010). This suggests that they possess similar capabilities of detection and could avoid slow moving vessels as well.

Research suggests that sea turtles cannot be expected to consistently notice and avoid vessels that are traveling faster than 2 miles per hour (Hazel et al. 2007). Given the high vessel traffic volume around Hawaii, collisions between turtles and 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

2008). This calculates to a 0.006% probability of a vessel strike with sea turtles for all vessels and trips, many of who are not reducing speeds or employing lookouts for listed species.

ESA-listed species in the California EEZ are also exposed to the risk of collision with HI SSLL vessels. However, the annual number of trips transiting through the California EEZ in and out of ports on the California coast is relatively low. For example, the year with the most trips was in 2015, with 35 trips completed where the vessel either left from or returned to Long Beach or San Francisco ports.

The vessels in the HI SSLL fishery range in size from 20-24 m (Ito et al. 1998). They are steel-or fiberglass-hull vessels that travel at speeds less than 10 knots (SF BE). Since 2004, there has been a fluctuation of active vessels participating in the HI SSLL fishery. Similarly, there is variability in the amount of fishing trips each year. In 2006, there were 35 active vessels, the most amount of vessels to participate in the HI SSLL fishery in a single year. The most amount of fishing trips completed in a single year was 112 in 2009.

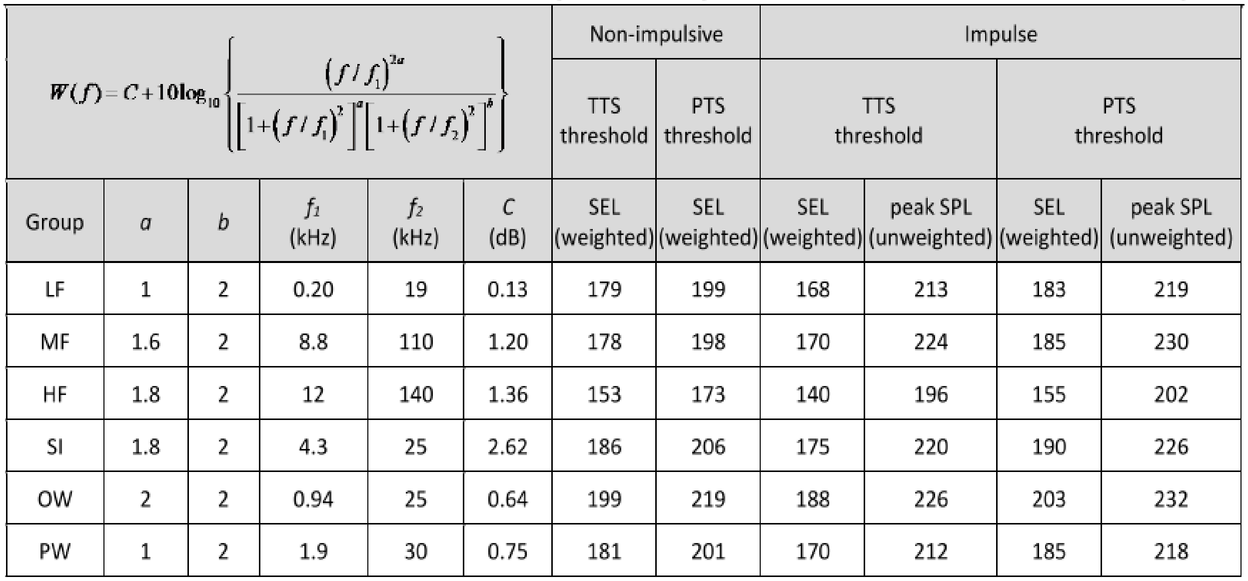
Given the small number of vessels participating in the fishery, the small number of anticipated vessel trips, the slow vessel speeds during fishing operations and vessel transiting, the expectation that ESA-listed marine species would be widely scattered throughout the proposed action area, and the limited populations of the ESA-listed species in these areas, the potential for an incidental vessel strike is extremely unlikely to occur. Thus, the effects of this stressor to any ESA-listed marine species under NMFS' jurisdiction listed in pelagic and coastal exposure categories are discountable.

4.1.6 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). Of these physical effects, the one measurable effect that is most likely to occur at the lowest noise intensity, would be TTS hearing loss. Therefore, we used TTS as a surrogate for all physical effects from noise from the proposed activities in order to assess the likelihood or extent of adverse effects from vessel noise.

When analyzing the auditory effects of noise exposure, noise is generally categorized as either impulse or non-impulsive noise. Acoustic thresholds for impulsive sounds use dual metrics: cumulative sound exposure level (SELcum) and peak sound pressure (PK). For non-impulsive sounds, acoustic thresholds are presented using the SELcum metric (NMFS 2016, 2018). All species under consideration in this opinion, except for sea turtles, are represented by categorical groups in Table 5. For example, the TTS level for 8-hours of exposure to noise for phocid pinnipeds (i.e. Hawaiian monk seal) is 181 dB SEL_{cum} and 199 dB SEL_{cum} for otariids (i.e. Guadalupe fur seal) (NMFS 2016, 2018). Sea turtles on the other hand have values of 232 dB_{peak} and 204 dB SEL_{cum} (U. S. Navy 2017). Cetaceans can be referenced by determining their applicable frequency group. However, it should also be noted that we do not expect any ESA-listed animals to be exposed to this level of sound continuously for 8 hours.

Table 5. Summary of weighting function parameters and TTS/PTS thresholds. SEL thresholds are in dB re 1 µPa2s and peak SPL thresholds are in DB re 1 µPa. Source: Table AE-1, NMFS 2016, 2018.



$W(f) = C + 10\log_{10}\left\{\frac{(f/f_1)^{2a}}{\left[1+(f/f_1)^2\right]^a\left[1+(f/f_2)^2\right]^b}\right\}$						Non-impulsive		Impulse			
						TTS threshold	PTS threshold	TTS threshold		PTS threshold	
Group	*a*	*b*	*f₁* (kHz)	*f₂* (kHz)	*C* (dB)	SEL (weighted)	SEL (weighted)	SEL (weighted)	peak SPL (unweighted)	SEL (weighted)	peak SPL (unweighted)
LF	1	2	0.20	19	0.13	179	199	168	213	183	219
MF	1.6	2	8.8	110	1.20	178	198	170	224	185	230
HF	1.8	2	12	140	1.36	153	173	140	196	155	202
SI	1.8	2	4.3	25	2.62	186	206	175	220	190	226
OW	2	2	0.94	25	0.64	199	219	188	226	203	232
PW	1	2	1.9	30	0.75	181	201	170	212	185	218

Based on the maximum vessel size, we expect the RMS SPL to be 180 dB with an average around 160 dB during normal vessel operations. These noises are generally classified as mobile and non-impulsive. We used the maximum value of 180 dB for our calculations as this value would be the most significant noise produced. Two values were determined for source velocity (meters per second) based on the type of action, hauling (2.5722) versus transiting (5.1444). Lastly, a weighting factor adjustment of 2.5 kHz was used as non-impulsive vessel noise sources are comparable to vibratory pile drivers (NMFS 2016, 2018) and is the standard value used by NMFS based on Blackwell (2005) and Dahl et al. (2015). While fishing operations are intermittent, can be delayed due to weather or other variables, require soak times for the gear, etc., we analyzed continuous noise to determine maximum threshold values for expected isopleths. Sets typically take approximately six hours to set, and nine hours to retrieve in the shallow set fishery (NMFS unpublished data). The data is a compilation of 17,878 shallow set sets, of which 343 instances vessels steamed back to the beginning of the set to haul gear and 36 other times gear was retrieved from an alternate location – most likely due to line parts. After setting, a vessel may move to another area, circle back to the other side of the line, remain in neutral or free drift, etc. Additionally, operations typically occur in poor weather conditions which can affect noise distribution from the source (i.e. the vessel), travel times, etc. We understand that variability can occur between sets based on weather conditions and fishing operations; however we assessed an ideal state where the maximum noise exposure would potentially occur. Noise from other proposed activities would be less intense, and thus less likely to adversely affect any species under consideration.

NMFS has suggested behavior response thresholds of 120 dBRMS threshold for continuous sounds and 160 dBRMS for impulsive sounds for marine mammals and 160 dBRMS for all types of sounds for sea turtles (Finneran and Jenkins 2012; NMFS 2016, 2018). The primary cause of potential effects to listed marine resources from this proposed action is from underwater sound generated by the vessels generator or engine(s) through vibration transference, or from the propeller. Since NOAA's behavioral response threshold is so small, the isopleths to 120 dB are very large. There is high uncertainty regarding the accuracy of spreading models especially at long distances and behavioral responses at projected noise levels.

ESA-listed species in offshore areas are generally moving and their exposure to a moving source of sound would be relatively short lived. Furthermore, most of the action area is subjected to international vessels and international freight lanes where extremely large commercial vessels often transit.

The resultant isopleths for vessel hauling and setting are displayed in Table 6 below. TTS values are only applicable to low-frequency cetaceans (1.5 m), high frequency cetaceans (2.7 m), and phocid pinnipeds (0.7 m). Isopleths for constant vessel transit are displayed in Table 7 and are reduced further as the level of exposure is reduced as the vessel maintains course. TTS values are applicable to high-frequency cetaceans (1.4 m) and phocid pinnipeds (0.4 m).

Table 6. Sound pressure thresholds in decibels (dB) and stand-off distances in meters as determined by NMFS[2]. Resultant isopleths for hauling or setting gear.

Hearing Group	**Low-Frequency Cetaceans**	**Mid-Frequency Cetaceans**	**High-Frequency Cetaceans**	**Phocid Pinnipeds**	**Otariid Pinnipeds**	**Sea turtles**
SEL_{cum} Threshold	199	198	173	201	219	220
SELcum Threshold (TTS)	179	178	153	181	199	200
Behavioral Response threshold	120	120	120	120	120	160
PTS Isopleth to threshold (meters)	0.0	0.0	0.0	0.0	0.0	0.0
TTS isopleth to threshold	1.5	0.0	2.7	0.7	0.0	0.0

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[2] Key: Pink – threshold levels as defined by NMFS 2018 and U.S. . Navy 2018 (sea turtles); blue – resultant isopleths of calculations.

Table 7. Sound pressure thresholds in decibels (dB) and stand-off distances in meters as determined by NMFS[2]. Resultant isopleths for constant vessel transit.

Hearing Group	**Low-Frequency Cetaceans**	**Mid-Frequency Cetaceans**	**High-Frequency Cetaceans**	**Phocid Pinnipeds**	**Otariid Pinnipeds**	**Sea turtles**
SEL_{cum} Threshold	199	198	173	201	219	220
SELcum Threshold (TTS)	179	178	153	181	199	200
Behavioral Response threshold	120	120	120	120	120	160
PTS Isopleth to threshold (meters)	0.0	0.0	0.0	0.0	0.0	0.0
TTS isopleth to threshold	0.8	0.0	1.4	0.4	0.0	0.0

All species under consideration may respond to noises by avoiding, halting their activities, experience reduced hearing by masking, or attraction to source noises. Avoidance is most likely, and a common natural reaction by listed species and considered low risk. Although it is difficult to determine whether avoidance behavior is a result of noise or other cues. The results of both calculations show resultant isopleths would be very close to the vessel, and consider that the vessel is moving. Nevertheless, all species under consideration are large agile animals and capable of swimming away safely from disturbances that would harm them. Attraction to sounds are unusual but sometimes happen in certain species like sperm whales or false killer whales.

We understand little about the impacts of noise or the underwater noise thresholds for sea turtles, but their hearing ranges are generally limited and they are noted as being less reliant on hearing than marine mammals who regularly communicate by vocalization (NMFS 2016, 2018). Sea turtle ears are also more primitive than that of marine mammals and less capable of detecting sound pressures (Popper et al. 2014). For these reasons, sea turtles are less likely to detect and avoid sound pressures from underwater noise that could injure them, and less likely to behaviorally respond.

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 2009). 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 (Casper et al. 2003; Casper and Mann 2006, 2009; Casper et al. 2012; Ladich and Fay 2013; Myrberg

2001). 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). Particle motion dynamics and effects to species are poorly understood and merit further research (Popper and Hawkins 2018). Given their assumed hearing range, elasmobranchs are anticipated to be able to detect the low frequency sound from the proposed vessel noise if exposed. 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).

In regards to the MHI IFKW critical habitat, noises that would significantly impair use or occupancy are those that inhibit their ability to receive and interpret sound for the purposes of navigation, communication, and detection of predators and prey (83 FR 35062). Such noises are likely to be long-lasting, continuous, and/or persistent in the marine environment and, either alone or added to other ambient noises, significantly raise local sound levels over a significant portion of an area.

False killer whales rely on their ability to produce and receive sound within their environment to navigate, communicate, and detect predators and prey. With a foraging strategy that is adapted to the shelf and slope habitat of the MHI, these large marine predators travel in subgroups that are dispersed from each other but converge when prey resources are found. Accordingly, these animals rely on their ability to receive and interpret acoustic cues to find prey at a distance and convey information about available prey resources to other dispersed subgroups of IFKWs. Habitats that contribute to the conservation of MHI IFKWs allow these whales to employ underwater sound in ways that support important life history functions, such as foraging and communicating.

A large body of scientific information on the effects of anthropogenic noise on the behavior and distribution of toothed whales, including false killer whales, demonstrates that the presence of anthropogenic noise can adversely affect the value of marine habitat to MHI IFKWs (Shannon et al. 2016; Erbe et al. 2016; Gedamke et al. 2016; Hatch et al. 2016). Of particular concern are those noises that are chronic or persistent and cause cumulative interference such that the animals' ability to receive benefits (e.g., opportunities to forage or reproduce) from these habitats is sufficiently inhibited.

The density of MHI IFKW is expected to be very low along the transit routes since these whales are generally found in deeper areas just offshore, (median preferred depth is 1679 m) rather than nearshore areas (Baird et al. 2010; Baird et al. 2012). In addition, the transit routes are in low-use areas according to Baird et al. (2012) who describe the north side of the island of Hawaii, a broad area extending from north of Maui to northwest of Molokai, and a small area to the southwest of Lanai as high-use areas preferred by MHI IFKW.

We do not expect the transiting of fishing vessels through MHI IFKW critical habitat to produce noises that would significantly impair the use or occupancy of the habitat by MHI IFKWs based on the values shown in Table 7. Of important note, designated critical habitat is within the boundaries of established fishing exclusion zones, thus no fishing operations occur within the designated critical habitat and reduce the opportunity for noise disturbances from fishing operations.

MHI IFKW have been known to ride bows of vessels (Baird 2009). As the data suggests, the resultant TTS isopleths for mid-frequency cetaceans are 0 m regardless of transiting versus hauling, thus MHI IFKW would not be disturbed by vessel noise at these transit speeds. No other species under consideration in this biological opinion are known to ride vessel bows. High frequency cetaceans like the baleen whales would not be expected to traverse so close to a vessel for extended periods of time.

MHI IFKWs are anecdotally known to follow vessels for short periods of time while longline gear is being retrieved, as shown through satellite tracking data of a small number of individuals (Baird pers. comm. 2018; data presented at April 11, 2018 TRT meeting in Honolulu, HI; Anderson et al. in prep). The whales displayed a rapid approach to the vessel during hauling although no specific variable appears to clarify which environmental preferences the whales display corresponding to interactions with the vessel (Baird pers. comm. 2018). The rapid approach suggests the whales are using acoustic cues to detect the gear (Baird pers. comm. 2018). Expanding the literature review, Thode et al. (2007) determined that vessel hydraulic systems were not the primary acoustic cue which attracted sperm whales to longline fishing vessels in Alaska after hydraulic signatures were not detected. Thode et al. (2007) surmised that the cavitation noise produced by the vessel's propeller during hauling operations would likely be attracting the whales instead. However, no literature is present on MHI IFKWs that suggests either propeller or hydraulic noise produced from the Hawaii longline vessels attract MHI IFKWs to a vessel at this time. This is further substantiated by Thode et al. (2015) whom discuss the difficulties in assessing this type of information specific to pelagic longlines and the false killer whale.

While 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; we believe it is highly unlikely noises generated from vessel operations would illicit significant behavioral reactions from ESA-listed species considered in this consultation. Any potential behavioral responses from stressors that create noise are expected to have discountable effects on listed species.

In conclusion, the only way any species under consideration would be affected by noise to the level of TTS is from being within two meters of the vessel for a significant time period, which is highly unlikely given the nature of the animals, movement of the vessel, etc. Therefore, the effects of this stressor to any ESA-listed marine species under NMFS' jurisdiction listed in pelagic and coastal exposure categories is considered to be insignificant.

Introduction of vessel wastes and discharges, gear loss and vessel emissions

The diffuse sources of stressors associated with the longline fisheries: vessel waste discharge, gear loss, and carbon emissions and greenhouse gasses, affect both pelagic and coastal areas. Equipment spills, discharges, and run-off from the project area could contain chemicals such as fuel oils, gasoline, lubricants, hydraulic fluids and other toxicants, which could expose ESA-listed species. Depending on the chemicals and their concentration, the effects of exposure may range between animals temporarily avoiding an area to death of the exposed animals. In the unlikely event of a leak, we expect vessel crew will act accordingly to quickly minimize and contain the problem. It is anticipated that leaks or spills would be infrequent, small, and quickly cleaned. Any resulting discharges would be at extremely low concentrations, exposure to which is expected to cause no effect on an exposed individual's health, and result in no behavioral response.

Local and Federal regulations prohibit the intentional discharge of toxic wastes and plastics into the marine environment. However, the U.S. Department of Justice and the U.S. Coast Guard fined a Hawaii-based commercial longliner $475,000 over the discharge of oily waste. This was the fourth MARPOL enforcement action that the U.S. Coast Guard has brought against a Hawaiian longline fishing firm in 2018 (The Maritime Executive 2018). MARPOL ANNEX V limits the discharge of garbage, (primarily food waste, cargo residues, cleaning agents and animal carcasses), based on the distance from the vessel to the nearest land mass. The amount of vessel waste discharge from the HI SSLL fishing fleet is difficult to quantify with any accuracy. However, the observer program maintains a database they share with the U.S. Coast Guard on MARPOL violations documented by fishery observers. The database contains 37 entries for the HI SSLL from 2008-2017, and includes the following violations: discarding of plastic, fuel leaks, dumping oil, discarding plastic lined bait boxes and dumping chemical lights (lights attached to the mainline to attract swordfish or swordfish prey).

Loss of gear occurs when there is a break in the mainline and either a radio buoy is not attached to the lost strand of gear, or the lost strand has drifted so far that the radio cannot detect any radio signal, or the radio buoy has failed and is not emitting a signal. A break in the mainline can be caused by old monofilament weakened by the ultraviolet rays of the sun, a tangle with a large marine species, rough weather, etc. Lost gear could include monofilament mainline, hard and/or soft floats attached to the mainline, monofilament branch lines with hooks and a radio buoy. This gear can continue to fish and incidentally hook and entangle marine species. Loss of gear can cost a fisher anywhere from hundreds to thousands of dollars not including loss of potential catch and time while searching for gear. Not surprisingly, fisherman and crew are motivated to maintain their gear and avoid setting gear in adverse weather conditions.

HI SSLL vessels burn fuel and emit carbon into the atmosphere during fishing operations and transiting. The HI SSLL vessels have diesel engines with an average of 460 horsepower (Ito et al 1998). According to an economic review of the Hawaii-based longline fleet by O'Malley and Pooley (2000), HI SSLL captains that were interviewed reported an average of 258 gallons of fuel used per day while traveling to the fishing grounds. Parker et al. (2018), estimates that in 2011, the worlds 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 HI SSLL, we expect the contribution to global greenhouse gases to be relatively inconsequential based on the low number of participants in the fishery.

Given the small number of vessels participating in the fishery, the small number of anticipated vessel trips, the expectation that ESA-listed marine species would be widely scattered throughout the proposed areas of operation, and the limited populations of the ESA-listed species in these areas, the potential exposure to vessel waste, discharge, and emissions is extremely unlikely to occur. Thus, the effects of this stressor to any ESA-listed marine species under NMFS' jurisdiction listed in pelagic and coastal exposure categories are discountable.

4.2 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 HI SSLL fishery (these species fall into observed pelagic exposures). 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 or not the action's direct or indirect effects are likely to increase the species' probability of becoming extinct.

4.2.1 Leatherback Sea Turtle

4.2.1.1 Distribution

The leatherback sea turtle is globally listed as endangered, and the species is comprised of seven putative populations: the Northwest Atlantic Ocean, the Southeast Atlantic Ocean, the Southwest Atlantic Ocean, Northeast and Southwest Indian Ocean, and the West Pacific and the East Pacific Ocean populations (35 FR 8491; Wallace et al. 2013, Wallace et al. 2013 supplemental information). 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 (NMFS 2004a, 2005a, 2006a; Eckhert et al. 2012; NMFS and FWS 2013a). At sea the species has been documented between about 71º N to 47º S (Eckhert et al. 2012).

Based on genetic analyses leatherback sea turtles captured in the HI SSLL fishery seem to be from one population, the West Pacific Ocean population. Of the 65 genetic samples from the HI SSLL fishery, four unique haplotypes have been identified resulting in 98.3% of the turtles concluded to be from the West Pacific population. One haplotype found in one animal occurs in low frequency in both West Pacific and East Pacific Ocean nesting populations, and until further analyses are done cannot be resolved (P. Dutton, pers. Comm, 12 December 2017).

The East Pacific population occurs within the action area, along the coast of California, and exhibits some overlap in distribution with the West Pacific population (Tiwari et al. 2013). There is also evidence of two interactions from the East Pacific population of leatherback sea turtles in the Hawaii longline fisheries. One occurred in 1995 before the fisheries were separated and one sample is from the deep-set fishery, which was captured in 2011 and just recently confirmed to be from the Eastern Pacific population (Dutton pers comm. April 30, 2018). It was 62 cm SCL, caught to the Northwest of Kauai, hooked in the flipper and was released alive with no gear attached (NMFS unpublished data).

Differential Distribution

Throughout its life cycle the leatherback sea turtle relies on developmental habitats that include nesting beaches, and coastal and pelagic waters. Four basic life cycle stages are described: egg, hatchling, juvenile, and adult. These life history stages form the foundation of our life history model, and causal loop diagram (Figure 12), which we describe in more detail later in this chapter. Frequently, the species is described as transoceanic in its behavior. In fact, the adult leatherback sea turtle has the most extensive biogeographical range of any extant sea turtle (Eckert et al. 2012). However, the species also spends a considerable amount of time in neritic waters depending on life history stage. Movement patterns are varied, but distinct, within and among populations and populations, and this variation may influence demographic performance of populations that are geographically separated (use different LMEs of the Indo-Pacific) (Benson et al. 2011). Below we describe different distributional patterns of the two-leatherback sea turtle populations in the action area, the West Pacific Ocean population and the East Pacific Ocean population.

West Pacific Ocean Population

Adult leatherback sea turtles

Leatherback sea turtles mostly nest in tropical latitudes (Eckert et al. 2012). West Pacific Ocean leatherback sea turtles nest year round and exhibit two distinct migration patterns that seem to vary based on the nesting season. Nesting aggregations include Papua Barat, Indonesia, Papua New Guinea, the Solomon Islands, and Vanuatu (Eckert et al. 2012; Tiwari et al. 2013). Most nesting occurs in Jamursba Medi, Indonesia (70% or greater depending upon conclusions of the author and method of estimate) although numbers have steadily declined. The IUCN review evaluated six populations (nesting (index) aggregations), and Jamursba-Medi comprised more than 90% of the nesting abundance estimate in 2010, and about 70% of the total population in earlier years (see past estimates in Table 1 of Tiwari et al. 2013). IUCN estimated an 80% decline in this population across the past three generations. The nesting aggregation at Terengganu, Malaysia, is considered functionally extinct (Tiwari et al. 2013). Declines in nesting turtles would suggest the overall distribution of the population has likely experienced concomitant declines.

During the nesting season, adult leatherback sea turtles of the West Pacific Ocean population are found during inter-nesting in narrowly defined coastal areas, adjacent to their nesting beaches (Benson et al. 2011). West Pacific Ocean leatherbacks generally stayed within 300 km or less from nesting beaches in Indonesia (JM and Wermon beaches, Papua Barat), PNG, and the Solomon Islands (Benson et al. 2011). In PNG, however, some females visited additional nesting areas both within and outside of the Huon Coast, and may have also deposited nests along the northern coast of Madang province, Huon Peninsula, and Bougainville and Woodlark Islands (Benson et al. 2007b). This is an interesting nesting strategy whereby females disperse nests in other locations which increases population resilience, but may also increase exposure to threats (e.g., harvest, predation, or beach erosion).

The marine habitat distribution for the population extends north into the Sea of Japan, northeast and east across the North Pacific to the west coast of North America (predominantly shores of 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). 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, nesting occurs year round. Based on Benson et al. 2011, populations (nesting aggregations) to the north are summer nesters, and populations to the south are winter nesters (see Figure 1 in Benson et al. 2011). Summer nesters migrate north and off the west coast of the U.S, while winter nesting leatherback turtles tend to migrate to the southern hemisphere post-nesting (Benson et al. 2007a, 2011). What appears to be a latitudinal cline in the occurrence of life history strategies may, however, be a result of incomplete monitoring. There may be year-round nesting in other places in the West Pacific, including the Solomon Islands and PNG.

Turtles nesting in Papua Barat, Indonesia during the boreal summer months migrate to the South China Sea and to the North Pacific Ocean, with some traveling across the Pacific past Hawaii to foraging grounds in temperate waters off North America (Benson et al. 2007a,b; Benson et al. 2011). This Bird's Head nesting population exhibits strong site fidelity to the central California foraging area (Benson et al. 2011) which puts them at risk of interacting with Hawaii longline fisheries during migrations. Among foragers tagged in coastal waters off California (n = 27 female and 10 male adults), the majority moved north and spent time in areas off northern California and Oregon, before moving towards the equatorial Eastern Pacific, then eventually westward presumably towards West Pacific Ocean nesting beaches (Benson et al. 2011).

Feeding and Diving Behavior

Adult leatherback sea turtles typically feed on pelagic soft-bodied animals, especially sea jellies, siphonophores, and tunicates. Despite the low nutritive value of their prey, leatherback sea turtles grow rapidly and attain large sizes, hence they must consume enormous quantities of prey. Most water content of the prey is expelled before swallowing to maximize nutritive value per unit volume. 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 (NMFS 2006). 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).

Satellite tracking and stable isotope analysis suggests that West Pacific Ocean leatherbacks are likely comprised of demographically discrete foraging ''units'' (Benson et al. 2011, Seminoff et al. 2012). For the West Pacific population, seven ecoregions (South China/Sulu and Sulawesi Seas, Indonesian Seas, East Australian Current Extension, Tasman Front, Kuroshio Extension, equatorial Eastern Pacific, and California Current Extension) were identified as important seasonal foraging areas (Benson et al. 2011). In the Pacific Ocean, leatherbacks spent long periods transiting over widely dispersed areas, indicating food availability was patchy (Bailey et al. 2008, 2012). By contrast, in the Atlantic Ocean, leatherback transit was frequently interspersed with foraging behavior, indicating leatherbacks frequently encountered areas of high prey density (Bailey et al. 2012). Prey abundance and distribution may explain why the Pacific populations are in worse shape than the Atlantic populations (Bailey et al. 2012).

Juveniles (Hatchlings, Juveniles and Sub-Adults)

The hatchling life history stage may be the most poorly understood, and generally includes a rest period following emergence and before entering the ocean, swimming away from land (frequently, described as a "frenzied" swim) for about 24 hours, after which they adopt a diel swimming pattern. Observations have indicated that as they grow, the hatchling will reduce the amount of time actively swimming to about week 8 when they will spend nearly the same amount of time swimming versus resting. Telemetry-based studies of juvenile leatherback turtles are limited, but a growing body of evidence suggests that thermal and geographic range is affected by body size, and the movements of juveniles and sub-adults (and adults) likely reflects the distribution of prey resources, and temperatures (Eckert et al. 2012)

Passive drifter models have been used to predict the trajectories of hatchlings offshore (e.g., Gaspar et al. 2012; Hamann et al. 2011; Shillinger et al. 2012). Passive drifter model predictions, combined with analysis of sighting, genetic, bycatch and satellite tracking information, indicate hatchlings emerge from nesting beaches in Jamursba-Medi, Indonesia, and Kamiali, Papua New Guinea, and are entrained by highly variable oceanic currents into the North Pacific, South Pacific, or Indian Oceans (Gaspar et al. 2012). After 1 to 2 years, these currents may take small juveniles into temperate regions where water temperatures in winter drop well below the minimum temperature likely tolerated by such small individuals. Eckert (2002) summarized the records of nearly 100 sightings of juvenile leatherbacks and found that animals less than 100 cm curved carapace length (CCL) are generally found in water warmer than 26˚C indicating that the first part of a leatherback's life is spent in tropical waters. Gaspar et al. (2012) hypothesize that after an initial period of mostly passive drift, juveniles begin to actively swim towards warmer latitudes before winter and back again towards higher latitudes during spring. This simulated migration pattern is used by adult leatherbacks from Jamursba-Medi and Kamiali (Gaspar et al. 2012). Scientists have theorized that an adult's choice of migration patterns are influenced by the currents they experienced as a hatchling—known as the "hatchling drift scenario" (reviewed by Saba 2013).

Feeding and Diving Behavior

The diet of young leatherback sea turtles has had limited study. For the first few days to weeks of the while in the frenzy swimming stage, the hatchling is sustained by their yolk. Salmon et al. (2004) found juvenile leatherbacks feeding on ctenophores, gelatinous eggs, and jellyfish, while diving and feeding at depths of 0.5 to 14 m, with depths increasing with age.

East Pacific Ocean Population

Adult leatherback sea turtles in the East Pacific population nests off the Pacific coast of North, Central and South America, from Mexico to Ecuador; marine habitat is generally described as between 130º W and 40 º S (Wallace et al. 2013 [East Pacific status]). The population generally occupies a distribution distinct from the West Pacific population, and is mostly located outside of the action area for the proposed action. However, there are some areas where East and West Pacific populations can overlap, and there is evidence of East Pacific leatherback sea turtles occurring in the action area, albeit on rare occasion.

Tagging studies have shown that Eastern Pacific post-nesting females migrate southward to the south Pacific after nesting in Costa Rica (Shillinger et al. 2008, 2011), The adult turtles commonly forage offshore in the South Pacific Gyre in upwelling areas of cooler, deeper water and high productivity (Shillinger et al. 2011). During the nesting season, they stay within the shallow, highly productive, continental shelf waters (Shillinger et al. 2010). Tagging studies have shown that adults sometimes migrate to highly productive upwelling areas near continental shelves, such as off Oregon and Washington (Benson et al. 2007a, 2011). There are also data sets on at-sea distribution that were collected via observers and fishers onboard fishing vessels in the Eastern Pacific. The primary dataset available was developed by IATTC and shows a wide distribution of leatherback sea turtles throughout the Eastern Pacific, ranging from the Gulf of

California, Mexico to Peru (IATTC 2012). However, genetic analyses of juvenile and adult leatherback sea turtles caught in fisheries off Peru and Chile indicate that a proportion (approximately 16% of sampled turtles) are from West Pacific rookeries (NMFS and FWS 2013a).

Like the West Pacific Ocean population, very little is known for the hatchling to sub-adult stage for the Eastern Pacific stages. As described above for the West Pacific Ocean population, there is generally a rest period following emergence and before entering the ocean, swimming away from land (frequently, described as a “frenzied” swim) for about 24 hours, after which they adopt a diel swimming pattern. Observations have indicated that as they grow, the hatchling will reduce the amount of time actively swimming to about week 8 when they will spend nearly the same amount of time swimming versus resting. Telemetry-based studies of juvenile leatherback turtles are limited, but a growing body of evidence suggests that thermal and geographic range is affected by body size, and the movements of juveniles and sub-adults (and adults) likely reflects the distribution of prey resources, and temperatures (Eckert et al. 2012).

Drift models simulating hatchling dispersal from Eastern Pacific nesting beaches found that they were widely dispersed to productive oceanic habitats like the Costa Rica Dome when leaving beaches from Playa Grande, Costa Rica, and Chacocente, Nicaragua. Models of dispersal for hatchlings leaving from beaches north or south of Playa Grande show they are most likely transported offshore with large eddies known as “hatchling highways” (Shillinger and Bailey 2015). The East Pacific Ocean population has less diversity in their migration routes compared to the West Pacific Ocean population; they only travel to the south Pacific to areas off the coast of South America (Bailey et al. 2012, Shillinger et al 2009, 2011).

Based on the above description, individual East Pacific Ocean leatherback sea turtles are not considered likely to be hooked or captured in the HI SSLL fishery because: (1) 100% of the sampled leatherback sea turtles (35/35) were of West Pacific origin; (2) the one East Pacific interaction in the deep-set fishery was 6° of latitude south of the HI SSLL fishery fishing area; and (3) a recent study of 46 tagged leatherbacks tracked over 12,095 cumulative tracking days demonstrated that East Pacific leatherbacks migrate south of the HI SSLL fishery fishing area after nesting (Shillinger et al. 2008). Individual East Pacific Ocean leatherback sea turtles may be exposed in coastal areas to vessels transiting to and from California ports, which means they are at risk of exposure to vessel collision, vessel noise, vessel waste, discharge and emissions. These stressors may result in immediate exposures, or exposures later in time (e.g., interaction with derelict gear or vessel waste). However, like other species in the category of potential coastal exposures, the exposure of the East Pacific Ocean leatherback sea turtle is considered highly unlikely and therefore discountable. Therefore, the bulk of our analysis focuses on the effects of the action on the West Pacific Ocean population of leatherback sea turtles.

4.2.1.2 Population Structure

Defining the substructure of a species has long challenged conservation biologists, particularly for species like the leatherback sea turtle with its global distribution (Wallace et al. 2011). The IUCN’s Marine Turtle Specialist Group conducted extensive meta-analyses on marine turtles during which they integrated data on genetics, nesting sites, and tagging studies and identified this finer demographic substructure to the globally listed leatherback sea turtle (Wallace et al. 2010, 2011, 2013). Dutton et al. (1999) suggested that based on their analysis of mtDNA and global distribution of the species, the leatherback sea turtle evolved from animals in the Indo-Pacific during the early Pleistocene. Although listed globally, conservation and extinction risk should be evaluated at lower demographic units. In contrast to the global entity, Wallace et al. (2013) indicated that the population level should be given priority for assessing the extinction risk of the leatherback sea turtle. However, further intrapopulation variation at smaller scales also likely occurs (e.g., some of the Atlantic populations) and likely warrants population management. Below we provide some details of the structure of the leatherback sea turtle populations found within the action area of the proposed action.

Pacific Populations

Leatherback sea turtles captured in the HI SSLL fishery seem to be from the West Pacific population. While the East Pacific Ocean leatherback population and the Terengganu, Malaysia population would also likely occur in the action area, the Terengganu population is considered functionally extinct (Wallace et al. 2013) and the few leatherback sea turtles that still nest in Malaysia are part of that functionally extinct population, although they have previously been described as an independent stock (Dutton 2006 [SWOT report] and are included in the abundance estimates for the West Pacific population (Wallace et al. 2013). Despite its relationship in the demographic hierarchy, the loss of the Terengganu leatherback sea turtle represents a serious reduction in the spatial and genetic variation that once comprised leatherback sea turtles in the Pacific Ocean.

At least three nesting aggregations make up the majority of the population West Pacific Ocean leatherback sea turtle population: Papua Barat, Indonesia, Papua, New Guinea, and the Solomon Islands (Dutton et al. 2006 [SWOT]; Tiwari et al. 2013). The Papua Barat, Indonesia, leatherback sea turtles exhibit both a summer and winter nesting life history strategies. The extent to which these life history strategies affect population structure, or structure affects the occurrence of these strategies is not yet clear. However, we do know that there is diversity among nesting females in use of foraging habitats (Benson et al. 2011; Lontoh 2014). Leatherbacks that exploit distant temperate foraging habitats (e.g., central California) appear to require multiple years of seasonal foraging before remigrating to nesting beaches due to greater energetic demands. In contrast, leatherbacks exploiting geographically closer, year-round prey resources (e.g., Sulu Sulawesi and South China Seas) in more tropical habitats remigrate more frequently. The foraging region used by a particular individual turtle may result in different growth advantages (or conversely, tradeoffs). Benson et al. (2011) revealed that the CCL and curved carapace width of boreal winter nesters were larger than the summer nesters in a study that examined 126 leatherback sea turtles. Generally, temperate foragers had greater curved carapace width over tropical foragers for all life history types combined (Benson et al. 2011)

Stable isotopes linked to particular foraging regions have confirmed nesting season fidelity to specific foraging regions in leatherback sea turtles (Seminoff et al. 2012). Approximately 38 to 57% of summer nesters forage in waters off the U.S. West coast during summer and fall (Benson *et al.* 2007b, 2011, in prep). Lontoh (2014) found that variation in body size and reproductive output is associated with foraging region, which is likely a reflection of productivity and energetic costs associated with migration. For example, leatherback turtles that foraged in the northeast Pacific Ocean exhibited greater body size and longer remigration intervals than those that foraged in the South China Seas or North Pacific Transition Zone. Further, leatherback turtles that foraged in the North Pacific Transition Zone laid more clutches and had shorter remigration intervals than others (Lontoh 2014). Body size of turtles from Lontoh (2014):

160.8 ± 0.9 cm CCL (117.2 ± 0.6 cm CCW) for Northeast Pacific foragers;

156.9 ± 1.0 cm CCL (114.2 ± 0.6 cm CCW) for North Pacific Transition Zone foragers;

156.3 ± 1.0 cm CCL (113.2 ± 0.7 cm CCW) for South China Sea foragers

Seminoff et al. (2012) suggested based on satellite tagging and stable isotope analysis that northeastern foragers likely remained within relatively restricted foraging areas, and the western foragers showed more variation in movements between the North Pacific Transition Zone and the South China Seas.

Leatherback sea turtle age at maturity is uncertain. Estimates range widely between 9-15 years and 26-32 years based on skeletochronological analyses (Jones et al. 2011) and inferences from mark-recapture studies (e.g., approx. 15 yrs.; Dutton et al. 2005). Extrapolations of captive growth curves under controlled thermal and trophic conditions suggested size at maturity could be reached in 7-16 yrs. (Jones et al. 2011). Thus, a high degree of uncertainty remains about leatherback sea turtle age at maturity in the wild. Likewise, leatherback sea turtle lifespan is unknown. The minimum size when West Pacific Ocean leatherback females first breed is estimated to be >120 cm SCL (Jones et al. 2011; Stewart et al. 2007; Van Houtan 2011).

Phenotypic and genotypic variation occurs within and across subpopulations that comprise the West and East Pacific Ocean populations of leatherback sea turtles. Regional variation is inherent to several traits, is apparent in other areas covered by other populations that comprise the species (see Eckhert et al. 2012 for a comprehensive review of biological data on the global leatherback sea turtle). Table 8 contains a subset of the biological data assembled by Eckert et al. (2012) for leatherback sea turtles specific in the West and the East Pacific populations, with specific reference to the data used in the recent IUCN assessment for these populations. Clutch size, which is the total number of eggs that a female lays in a clutch is more important than clutch frequency or egg size (Wallace et al. 2007; Spotila and Tomillo 2015). Larger female leatherback turtles will produce larger clutches, rather than larger eggs, although there may be a relationship between female body size and egg size (Rostral in Spotila and Tomillo 2015). Rostral (in Spotila and Tomillo 2015) suggest that for the leatherback sea turtle “selection appears to have favored larger numbers of relatively smaller eggs as well as an increased number of clutches.”

Table 8. Population and subpopulation traits as recorded by various researchers and summarized by Eckert et al. 2012. Population numbers in bold were derived by the IUCN Marine Turtle Group and used in the 2013 status assessment for the subpopulation (Tiwari et al. 2013). Numbers in parentheses are study specific sample sizes. A full list of contributing authors is in Eckert et al. 2012).

Population or Subpopulation	Sex Ratio	Clutch Size (sample size)	Clutch Frequency	Remigration Interval (nesting seasons)	Internesting interval (days between nesting events of same season)	Yolkless eggs	Emergence Success (%)
West Pacific	**3 to 1**		**5.5**	**2.5**			
Terengganu		82.3 (627)		2	9 to 10		
East Coast, AUS		87.0 (5)			9.1	42.4 (5)	
New So Wales, AUS		97.7 (3) range: 94-104					
Queensland, AUS		86.1 +/- 15.7 (16) range: 64-121					
Wreck Rock, AUS		82.8 +/-13.1 (14)				46.5 +/- 14.5 (13)	
Papua		72 (25)			9.5	67 (25)	34.7 (25)
Papua: JM		79.6 +/- 16.3 (48)					
PNG		94.6 +/- 27.28 (44), range: 16-150			11		58(10)
PNG		94.7 +/- 21.9 (94); range: 43-156			14.7		
PNG		88.2 +/- 20.2 (37); range: 42-118					
Eastern Pacific	**3 to 1**		**7.2**	**3.7**			
PNMB, CR			9.5		9 to 10		
Playa Naranjo, CR		68.6 (6)					
Playa Langosta, CR		65.3 +/-15.9			9	39.4 +/-22.1	26.5 (33)

Population or Subpopulation	Sex Ratio	Clutch Size (sample size)	Clutch Frequency	Remigration Interval (nesting seasons)	Internesting interval (days between nesting events of same season)	Yolkless eggs	Emergence Success (%)
		(48)				(48)	
Playa Langosta, CR		64.5 +/-15.7 (131)			9.6		31.6 (26)
Playa, Grande, CR		64.7 (1389)		3.7 (92)	9	38.5 (1389)	54.2 +/-23.2 (164)
Playa Grande, CR		61.8 +/-16.3 (334)		3.7 (448)	9.5	947g +/- 393.8 (334)	50.4 +/- 9.2 (53)
Playa Grande, CR							41.0 +/- 25.2 (334)
Playa Grande, CR							0.38 +/-0.27 (414)
MEX			5.5		9 to 10		66.4 (151)
Jalisco, MEX		66 (8)			9.7		
Michoachán, Guerrero, Oaxaca, MEX		62.0 +/- 17.9 (1098)					

Reliable estimates of survival or mortality at different life history stages are not easily obtained. Annual female survival probability has only been estimated for PNG. Pilcher and Chaloupka (2013) used capture-mark-recapture data for 178 nesting leatherback sea turtles tagged at Lababia, Kamiali beach, PNG over a 10-year boreal winter nesting period (2000-2009) to estimate female survival. The annual survival probability was constant over the 10-year period (Pilcher and Chaloupka 2013), but was lower than those estimated for Atlantic rookeries (Dutton *et al*. 2005; Rivalan *et al*. 2005). The reason for the lower annual survival rate is unknown and may be due to several factors such as greater anthropogenic impacts or lower site fidelity (Pilcher and Chaloupka 2013).

4.2.1.3 Status

The leatherback sea turtle is listed as endangered throughout its range. The Services completed a 5-year review of the species status in 2013 and concluded the species was still endangered (NMFS and FWS 2013a). Some subpopulations are stable or increasing, but other populations for which information is available are either decreasing or have collapsed (NMFS and FWS 2013a, TEWG 2007, PLAWG 2012). The IUCN estimates a 40.1% decline over the past three generations of leatherback sea turtles in their 2013 assessment of the globally listed species (Wallace et al. 2013). Although, categorized as globally “vulnerable” according to IUCN red list criteria, four of the seven populations that comprise the species are categorized as “critically endangered by the IUCN. Wallace et al. (2013) note that one population, the Northwest Atlantic, is doing better than the remaining populations and masks the true extinction risk of the global species. Because the species is composed of a number of distinct populations, they caution that the appropriate scale to evaluate the extinction risk of the leatherback sea turtle should not be higher than the population level. In the sections that follow, we discuss the status of the two populations that occur in the action area of the HI SSLL fishery.

West Pacific Ocean Population

According to the IUCN, based on nest and female counts, the West Pacific Ocean population has declined 83% during the past three generations. An estimated 1,438 individuals (Tiwari et al. 2012) make up the West Pacific Ocean mature adult leatherback sea turtle population (includes males, females and non-breeding females). Wallace et al (2013) noted the current average annual number of nests as 2,379, reduced from 14,276 nests three generations prior (estimated). The primary threats to the population have not ceased so the population is categorized as “critically endangered”. The IUCN predicted the population is likely decline by 96% by the year 2040 (e.g., 572 nests, and about 104 females per year nesting, or 260 adult females total [Tiwari et al. 2013; Wallace et al. 2013]).

Indonesia

Birds Head population harbors the last remaining nesting aggregation of significant size in the Pacific (Dutton et al. 2007, Hitipeuw et al. 2007) with approximately 75% of regional nesting occurring along the northwest coast of Papua Barat, Indonesia (also known as the Bird’s Head Peninsula) (Hitipieuw et al. 2007). The Bird’s Head Peninsula has four main beaches used as index beaches, three comprise the Jamursba Medi complex, and the fourth is Wermon beach (Dutton et al. 2007; Hitipeuw et al. 2007). Nesting data from Jamursba-Medi are highly variable from year to year, and no data are available from 1998 due to a lack of survey effort that year. For the 17-year period 1993-2010, nesting fluctuated annually, with the overall trend declining by a rate of 5.5% per year (Tapilatu et al. 2013). The total number of nests per year for the Jamursba-Medi leatherback sea turtle nesting population ranged between a high of 6,929 nests in 1996 and a low of 1,596 nests in 2011 (Hitipeuw et al. 2007, Tapilatu et al. 2013). Based on nest counts and clutch frequency per season (mean = 5.5 +/- 1.6 nests per female), Tapilatu et al. (2013) estimated that 464 to 612 females nested at Birds Head beaches during the 2011 nesting season.

Nesting beach monitoring at Wermon began in November 2002 and ran through June 2003 with 1,788 nests recorded (Hitipeuw et al. 2007). Monitoring was conducted again from November 2003 through September 2004, which resulted in the highest number of nests recorded for 2003-04 totaling 2,881 nests (Hitipeuw et al. 2007). Monitoring resumed in November 2004 and continued year round thereafter. Nesting declined during 2005 to approximately 1,300 nests (Wurlianty and Hitipeuw 2007), although Tapilatu et al. (2013) report 1,497 nests laid at Wermon in 2005. Since 2005, nesting has declined by 11.6% per year at Wermon to 1,096 nests laid in 2011, representative of 189-249 females, or a 62% decline since monitoring began in 2002 (Tapilatu et al. 2013).

Since 2012, beach monitoring activities have been somewhat variable, however, the last three years of nest counts (2015 - 2017) by the University of Papua (UNIPA) field monitors indicates that nesting trends have continued to decline. The current estimate of total nester abundance of females nesting between 2015 and 2017 (i.e., one remigration interval), is 723 females at Jamursba Medi and 554 females at Wermon (UNIPA unpublished data). The overall nesting trend has continued to decline by 5.6% per year between 2003 and 2017; however, there appears to be an uptick in nesting since 2013 at Jamursba Medi (Tiwari *et al.* in prep). In the Manokwari region, West Papua, Indonesia, nesting occurs year round and the number of nests recorded from 2008 through 2011 ranged from 84 to 135 (Suganuma et al. 2012); however, survey effort was limited and not consistent across years. The total number of adult females in the Bird's head region is estimated to be 1,949 based on summer nests (April –September) by Van Houtan (2013) (from Tapilatu et al. 2013).

A recent NOAA funded, WWF-Indonesian assessment team characterized leatherback nesting on three north coast beaches of Buru Island in Central Maluku (i.e., Waenibe, Waspait, and Wamlana villages, Fena Leisela District, Buru Regency). Initial monitoring of these beaches suggest that this 10.6 km stretch of shoreline supports the first substantial nesting population discovered outside of Papua, Indonesia in the last decade. Nesting activity appears to be year round with a primary summer nesting peak (May to July) and a secondary winter peak (December to February). During 2017, 203 nests were documented of which 114 were predated, and 16 were depredated (WWF 2018). Monitoring activities are ongoing, including genetic sample collection.

Papua New Guinea

Papua New Guinea hosts approximately 20% of West Pacific leatherback nesting activity, which occurs predominately along the Huon Gulf coast (Dutton et al. 2007). Long-term nesting trends are difficult to determine given changes in monitoring effort since 2000 (Pilcher 2012). In 2004, an aerial survey counted 415 nests along the 4,516-km flown, with 71% of nests within the Huon Gulf coast (Benson et al. (2007a). For a 2-km stretch at Kamiali, Papua New Guinea, Benson et al. (2007a) reported that between 2000/2001 and 2003/2004 the total number of females estimated to have nested ranged from 41 to 71. Ground surveys at Kamiali recorded a total of approximately 215 nesting events for 1999/2000 through 2003/2004 (Benson et al. 2007a), and 178 individual leatherback females were tagged over 10 nesting seasons (2000−2009; Pilcher and Chaloupka 2013), with an additional 16 females tagged between 2010 and 2013 (Pilcher 2010, 2011, 2012, 2013). The total number of leatherback turtles tagged, at all sites of the Huon Coast, between 1999 and 2013 was 576 nesting females (Pilcher 2006, 2008, 2009, 2010, 2011, 2012, 2013).

During the 2010-2011 nesting season along the Huon Coast, 79 leatherbacks nested 29 times laying a total of 527 nests (Pilcher 2011b). Of the 211 nests laid between October 2012 and February 2013, 22% were lost to erosion, poaching, or did not hatch (Pilcher 2013). Although the population fluctuates each year, it appears generally stable since the 2006-2007 nesting season, when monitoring effort was standardized. But overall nest counts have declined approximately 93% since 1980 estimates when approximately 300 females were estimated to nest annually (Bedding and Lockhart 1989; Hirth et al. 1993; Pilcher 2009) suggesting a decline in nesting numbers. For example, Hirth et al. (1993) recorded 76 nests at Labu Tali during a 15 day period in 1989 over a distance of 725 meters of beach. In 2010, Pilcher (2011a) recorded 59 nests at Labu Tali over a distance of 3.2-km of beach during a 218 day period. Given the greater spatial and temporal survey conducted in 2010, the nesting numbers at Labu Tali likely represent a decline in nesting activity. While these results may be an artifact of sampling inconsistencies, it is entirely possible that the leatherback population in Papua New Guinea experienced near total nest harvest for some 40+ years through egg collection after World War II until conservation efforts were first implemented in 2000 (Bedding and Lockhart 1989; Bellagio Steering Committee 2008; Hirth et al. 1993; Quinn et al. 1983). In the Autonomous Region of Bougainville Island, Papua New Guinea, aerial surveys conducted in during the 2006-2007 nesting seasons, estimated 160-415 nests were laid per year (Dutton et al. 2007). Ground surveys conducted in 2009, found only 46 leatherback nests with a high level (83-100%) of nest harvest and relatively frequent harvest of adult leatherback turtles (Kinch et al. 2012).

The most reliable trend information begins from the 2006 - 2007 nesting season, and since then nesting activity has been stable or slightly increasing (Pilcher 2011). During the 2010-11 nesting season, 527 nests were recorded between October and May, the highest number of nests recorded since 2000 (Pilcher 2011). Night-time surveys between late-November through mid-February of the same nesting season resulted in 79 identified nesting females, of which 30 were remigrants (turtles from previous seasons), 15 were new turtles never tagged before, and 34 were renesting events of turtles already identified previously in the season (Pilcher 2011). However, monitoring activities again halted during the 2013-2014 nesting season due to community discord and disagreements, and monitoring has not resumed. Of nests laid in PNG during the 2012-2013 nesting season, 22% were lost to erosion, poaching or did not hatch. Overall, total nest counts for these years reflect a decline of approximately 93% in nesting activity since 1980 estimates (Pilcher 2009, 2013).

Solomon Islands

In the Solomon Islands, nesting 30 years ago occurred at more than 15 beaches (Vaughan 1981). Dutton et al. (2007) estimated that approximately 640 - 700 nests were laid annually in the Solomon Islands in 1999 – 2006 representing approximately 8% of the total West Pacific Ocean leatherback metapopulation at that time. Important nesting areas remain on Isabel Island at two principal beaches, Sasakolo and Litogarhira, with additional nesting occurring on Rendova and Tetepare in the Western Province (Dutton et al. 2007).

Nesting beach monitoring began in 1993 at Sasokolo by the Department of Fisheries where an average of 25 females deposit approximately 100 nests per season (Pita and Broderick 2005; Ramohia et al. 2001). The Tetepare Descendants' Association turtle monitoring program has operated since 2002 supporting beach rangers to monitor nesting activity at Tetapare and Rendova and has permanently closed a 13-km beach to harvest. At Tetapare, approximately 30-50 leatherback nests are laid seasonally (Goby et al. 2010; MacKay 2005).

At Rendova, 79 nests were laid during the 2009-10 winter nesting season of which only three hatched (Goby et al. 2010), and during the 2003-04 winter nesting season, 235 leatherback turtle nests were recorded of which only 14 hatched (Pilcher 2010), strongly suggesting that low hatch success poses significant impact to the current nesting population in the Solomons. During a January 2011 site visit, 315 nests were documented at Sasakolo and Litogahira (Tiwari 2011 unpublished). Between July 1, 2012 and April 30, 2013, Tetepare Descendants' Association undertook 257 beach surveys and found 44 leatherback nests (TDA 2013). Recently, nesting activity has also been documented at the southeastern side of Isabel where approximately 52 females may nest annually (TNC-Solomons 2018 unpublished). While monitoring efforts may be ongoing, data management and analysis remains a key challenge for these isolated communities (TDA 2013; Pilcher 2010b).

The community on Vangunu Island documented a total of 23 nests and 11 females between June 2011 and July 2014 (Jino *et al.* 2018). Nesting occurred during two distinct seasons from May-July and from November-January, and of the females tagged one nested successfully six times and another nested five times (Jino *et al.* 2018). The other nine turtles were only observed nesting once or twice and it is likely that some nesting events were not recorded or the females nested on surrounding unmonitored beaches (Jino *et al.* 2018).

Vanuatu

In Vanuatu there are low levels of scattered nesting on several beaches with a total of approximately 50 nests laid per year (Dutton et al. 2007; Petro et al. 2007; WSB 2011, 2015). Leatherbacks nest in small numbers on many of the islands, but approximately 10-15 females nest at the primary nesting beach at Votlo on Epi Island where surveys have been conducted since 2002/03. During 30 the 2010/11 nesting season, 41 nests were laid at Votlo, although only 8 nests hatched (Petro 2011).

In Malaysia, the major nesting rookery at Rantau Bang in Terengganu has collapsed from over 10,000 nests in the 1950s to 10 or fewer nests in recent years (reviewed by Eckert et al. 2012). The decline is attributed to fisheries interactions and egg harvest (reviewed by Eckert et al. 2012). In Fiji, 20-30 individuals are estimated to nest each year (Rupeni et al. 2002). In southeastern Australia, nesting is sporadic with less than a handful of nests each year (Dobbs 2002). Wreck Rock Beach in southern Queensland, Australia, reported an average 0-3 nests annually from 1969 to 1995, but no nests have been observed since 1995, despite regular monitoring (Flint et al. 2012). Nesting is irregular in northern Australia (Hamann et al. 2006a).

Within the Pacific basin, the central California foraging habitat has been studied over a 27 year period to examine population trends of foraging leatherback turtles. Approximately 38 to 57% of summer nesters forage in waters off the U.S. West coast during summer and fall (Benson et al. 2007b, 2011, 2018, in prep; Seminoff et al. 2012; Lontoh 2014). From 1995 to 2003, an estimated 12 to 379 individuals (mean = 178) foraged in this area (Benson et al. 2007); however, from 2004 to 2017, an estimated 23 to 112 individuals foraged in this area, representing a decline of 5.6% annually (Benson et al. in prep).

Leatherback sea turtles have been reported nesting in Vanuatu. Petro et al. (2007) reviewed archival data and unpublished reports, and interviewed residents of coastal communities, all of which suggested that leatherback sea turtle nesting has declined in recent years. There appears to be low levels of scattered nesting on at least four or five beaches with approximately 50 nests laid per year (Dutton et al. 2007). The primary leatherback sea turtle nesting site in Vanuatu is at Votlo on Epi Island where nesting beach surveys have been conducted since 2002-03. During the 2010-2011 nesting season 41 nests were laid at Votlo, although only eight nests hatched (Petro 2011).

Population Viability

For this consultation, Jones et al. (2018) modeled the current status of the West Pacific leatherback sea turtle population. To do this, Jones et al. (2018) assembled a team of six scientists to review previous models used in several previous biological opinions, review available data to conduct the current assessment, and recommend and complete a population model to help assess the extinction risk of the West Pacific leatherback sea turtle population. Although nesting (and monitoring) has occurred in Jamursba-Medi, Wermon, Manokwari, Buru Island, Japen Island, Wewe Koor, Huon Coast, Bougainville Island, New Britain Island, Madang Province, Isabel Island, Rendova (Western Province), Tetapare, Vangunu Island, Choiseul Island, Malaita Island, Epi Island, and Bamboo Bay, NMFS used the data from Jamursba-Medi and Wermon because these data sets are some of the longest and most complete data sets available on the West Pacific leatherback. Of these two "index" beaches, the summer nesters predominate and drive the data. The most consistent monitoring effort has been at Jamursba-Medi and its nesters are primarily summer nesters. Wermon has a stronger bimodal pattern of nesting, with summer and winter nesters in roughly equal proportions.

Even so, data was inconsistent and annual counts were not always available, so we used monthly proportions (nesting patterns) to fill in missing values. The goal was to estimate a long-term trend, which is the population growth rate, r, in the stochastic exponential growth equation $N_{t+1}=N_te^{rt}$, where N is the population size and t is time. For the leatherback sea turtle, r will not represent the true growth rate of the population because annual nest counts, which represent the bulk of data on turtles, only represent a portion of the population. Because mature females do not make nesting remigrations every year, and data is lacking to index other age classes, or the male portion of the population, there is inherent uncertainty and we consider r an estimate of population growth rate based on index data.

Given the inherent challenges of uncertainty and variability, NMFS carefully considered how several models would perform under the circumstances. Jones et al. (2018) considered using the diffusion-approximation population viability model (PVA, Snover 2008), which NMFS used in its 2008 biological opinion on the HI SSLL fishery; demographic models by Chaloupka (2002a,b), which NMFS considered in its 2002 and 2004 biological opinions on the HI SSLL fishery; the stochastic exponential growth model and the climate model by Van Houtan, which NMFS considered in the 2012 biological opinion on the HI SSLL fishery. In 2004, NMFS used a Dennis model in the FMP consultation, and this modeling approach. Table 9 lists several key reasons why we did not rerun or advanced these models for this consultation. Ultimately, to estimate the long-term population trend of leatherback sea turtles, Jones et al. (2018) developed a Bayesian state-space model after Boyd et al.(2017) because this method can separate the possible sources of variation in the annual nest count data that result from observation error (imperfect data collection) and process error (demographic and environmental stochasticity). The approach is a Bayesian adaptation of multivariate autoregressive state space (MARSS) models used for analysis of ecological time series.

Bayesian methods are designed to produce posterior probability distributions for estimated parameters based on data, and these distributions can be directly used to generate future projections. Additionally, the "state-space" aspect of the model allows specification of both a biological process model and an observation model, which allows the sources of variability in the annual nest count data to be parsed out and estimated separately. Consequently, the estimate for the long-term trend parameter is better isolated from noise in the data, and the variance surrounding its mean more accurately capture the variation in the trend itself without being conflated with data collection errors and natural variability (stochasticity).

Table 9. Models used or considered in previous consultations on the Hawaii shallow-set longline fishery.

Model Considered	**Author**	**Select Reasons Not Advanced in this Consultation**
Climate-based	Van Houtan 2011	▪ Short prediction interval (22 years) ▪ Couldn't produce long-term estimates of population viability and variance ▪ Limited to two climate parameters ▪ 2012 Modeled predictions did not occur (the model predicted the opposite of what has happened in the intervening years) ▪ Does not explain uncertainty in model parameters ▪ Lack of transparency and reproducibility
Demographic models	Chaloupka 2002a, b	▪ Limited biological data available to populate the model ▪ Require estimates for large number of parameters or use of surrogate species ▪ Does not explain uncertainty in model parameters
Diffusion Approximation	Snover 2008	▪ More sensitivity analyses could be performed (suggested in CIE review), especially for the extension of the approach that incorporates fisheries take ▪ Model estimates a fraction of the population, and does not account for variation in survey effort
Dennis Model	NMFS 2004	▪ Model estimates a fraction of the population, and does not account for variation in survey effort

The approach Jones et al. (2018) used to estimate the long-term population trend, i.e., population growth rate, is based on a stochastic density-independent exponential growth model, which we implement within a Bayesian state-space modeling framework, following Boyd et al. (2017). Estimation of the Bayesian state-space model was completed in JAGS (Plummer 2003), and projections (see below) were completed in R (R Development Core Team 2012) using the 'coda' and 'jagsUI' packages (Plummer et al. 2006, Kellner 2015). This type of model is commonly used for long-lived, slow-growing, late-maturing species such as sea turtles whose populations have been depleted to relatively low levels. The framework allows for estimation of both process variation (i.e., environmental and demographic variability) and observation uncertainty (i.e., imperfect data collection), and provides parameter estimates with probability distributions. The only inputs to the model are time series of nest count data, which we assume are an index of abundance for reproductive females in the population. The basic exponential growth equation underlying this approach is outlined below, and the model structure is described in Figure 8.

Exponential growth equation:

$Nt+dt = Nt*exp(r*dt)$
Nt = population size (number of individuals) at time t
dt = delta t, or difference in time (unit = years)
r = population growth rate, or log growth rate (i.e., long-term trend)
exp(r) = λ or "lambda", the finite rate of increase

Log-transformed equation & solved for growth rate:

$log(Nt+dt) = log(Nt) + r*dt$

$r = [log(Nt+dt) - log(Nt)]/dt$

From the model outputs, Jones et al. (2018) were most interested in the posterior distributions estimated for r, the population growth rate or long-term trend; Q, the process error variance; and $Nfinal$, the predicted nesting female count for the final year of observed data. Figure 8 is a schematic illustration of the Bayesian state-space model based on stochastic density independent exponential population growth. There are two major components: a process model describing the true number females nesting each year and an observation model relating the observed count data to the true number of nesting females. Three parameters are estimated: (1) r, the long-term trend, i.e., population growth rate, (2) Q, process error variance, and (3) R, observation error variance. The model also provides a predicted count (number of nesting females) with a probability distribution for each year of observed data. The predicted value for the final year of observed data is then used as a starting point for future projections. They used these three posterior distributions to project forward 100 years into the future (details below). The parameter distributions were estimated through a Bayesian model fitting process in which the program employs a Markov Chain Monte Carlo simulation to fit the model parameters to each set of data separately (i.e., the three time series of Annual Females for loggerhead turtles and two time series for leatherback turtles for a total of 2 models, 1 per species). Models were checked for suitable performance using the following diagnostic tools: examination of trace plots, effective sample sizes, Geweke statistics, Gelman statistics, and Deviance Information Criterion.

Jones et al. (2018) generated future projections using a simulation approach with 10,000 runs for each model. Each run began with a draw from each of three model-estimated distributions: the nesting female count in the final observed year, N_{final}; the growth rate, r; and the process error variance, Q. For each future year, new values of r and Q were drawn, making the projected growth rate and process error variance dynamic and best reflecting interannual variation in the observed data as well as the biology of the species. For each run, the number of nesting females for each future year was calculated according to the exponential growth equation but with the addition of a process error (see model diagram below). In addition, a current abundance estimate was generated for each projection run by summing the last 3 estimated count values (N_{final}, $N_{final-1}$, and $N_{final-2}$) from the same MCMC run as where the N_{final} value was drawn for the projection. The three year running sum was based on assuming a three year remigration interval (applicable to both loggerheads and leatherbacks), with the assumption that summing all females that nest during that period provides a snapshot of Total Reproductive Females in the population.

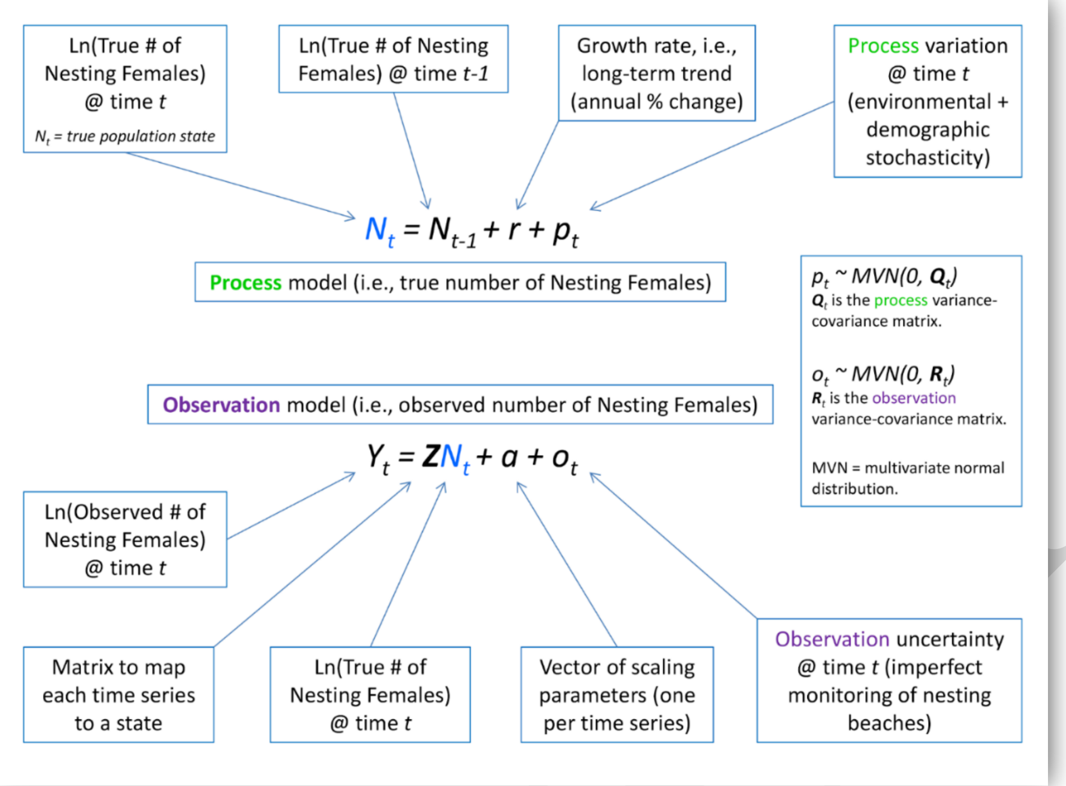
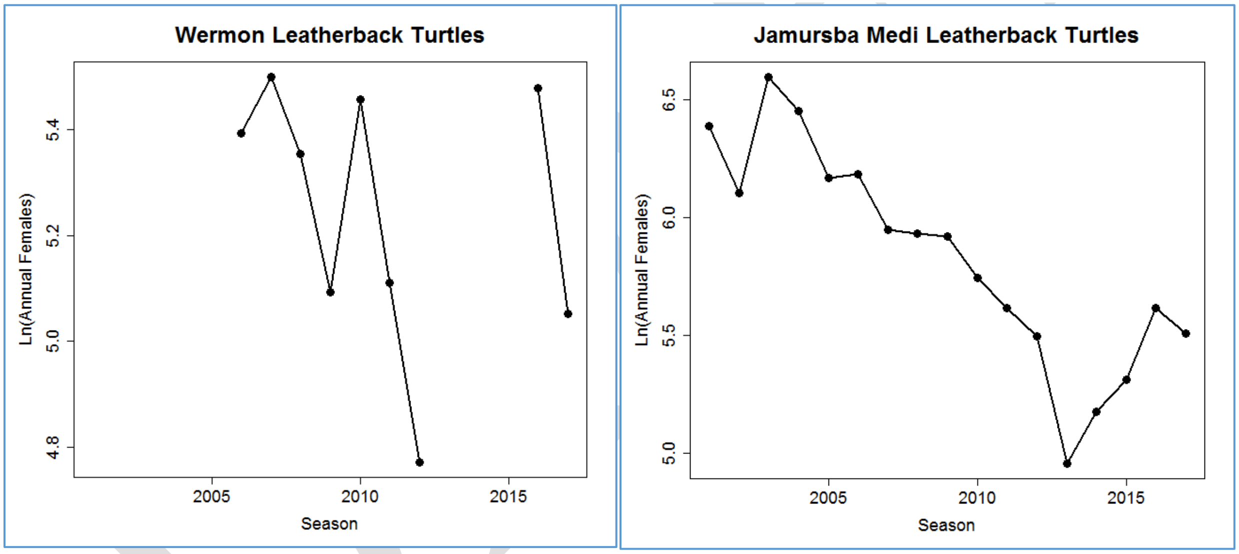


Figure 8. Schematic illustration of Bayesian state-space model based on stochastic density-independent exponential population growth.

Jones et al. (2018) calculated 3-year running sums from the annual female projections to create an index of total reproductive females. This allowed for the comparison of the projections to current abundance estimates. To do so, Jones et al. (2018) computed the proportion of runs for which the projected total reproductive females fell below (and remained below) 50%, 25%, and 12.5% of current abundance by 100 years in the future. For the set of runs ending below a threshold, we calculated the mean, median, and 95% credible interval for the number of years until the population fell below the threshold. We 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.

Annual female projections were derived from annual nest counts divided by the clutch frequency of 5.5. nests per female. Jones et al. (2018) used annual counts instead of a running sum to minimize assumptions about remigration interval. Raw data are comprised of time series from two nesting beaches in Bird’s Head, West Papua, Indonesia – Jamursba Medi (2001-2017) and Wermon (2006-2017, except 2013-2015). The data are shown in Figure 9 as the natural log of annual nesting females; this is how they were input to the model.

Figure 9. Natural log of annual nesting female leatherback sea turtles in Jamursba Medi and Wermon aggregations



Jones et al. (2018) fit the population growth rate, ‘r’, using the Bayesian state-space model approach based on density-independent stochastic exponential population growth as in Boyd et al. (2017) (see Figure 9). Figure 10 illustrates the predicted model median (blue line) of the natural log of annual nesting females with 95% credible intervals (gray shading) are overlaid on data points (black dots) for each time series, scaled appropriately by the model-estimated scaling parameters.

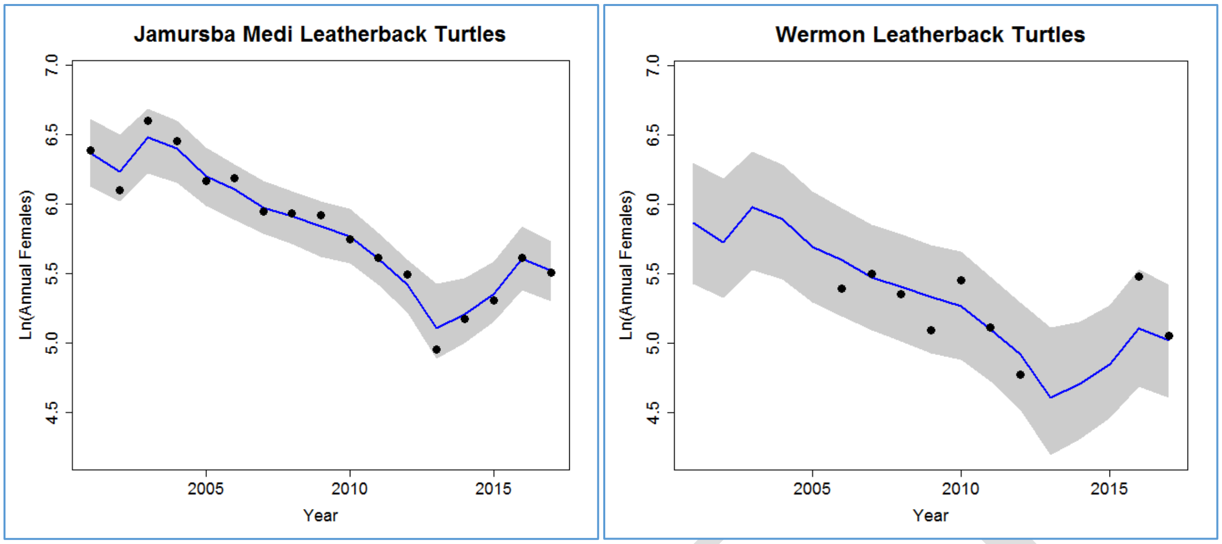


Figure 10. Predicted trends in nesting female leatherback sea turtles from Jamursba Medi and Wermon (median (blue line) of the natural log of annual nesting females with 95% credible intervals (gray shading)).

Jones et al. (2018) used model-estimated annual female distributions for 2015 to 2017 to estimate an index of current total reproductive female abundance; this is computed as a 3-year run sum (3 year assumed remigration interval). The estimated distributions for 2015-2017 annual females and the summed total reproductive females estimate are summarized in Table 10.

Table 10. Model-estimated female leatherback sea turtle numbers and the associated 95% credible intervals.

Observed data year	**Median**	**Lower 95% CI**	**Upper 95% CI**
Final Data Yr -2	340	275	433
Final Data Yr -1	439	350	552
Final Data Yr -0	401	324	495
Sums=CurAbundEst	1180	949	1479

Figure 11 below shows the 10,000 model runs projecting 100 years into the future from the final data year (2017). Model projections start with a value drawn from the posterior distribution of annual females for 2017; use dynamic ‘r’ (draw new ‘r’ each future year within a simulation run) and dynamic process error variance, ‘Q,’ to add natural variability. Model projections are of annual females in natural log space.

Next, Jones et al. (2018) estimated the mean and median time until the population declines to 50%, 25%, and 12.5% of its current abundance estimates. Abundance thresholds were calculated for each future projection (n=10,000 simulation runs) of total reproductive females (3-year running sum of projected annual females) is compared to the specified fractional decline from the starting estimate of current total reproductive females. Values are presented in Table 11. Whereas Table 12 lists the probability of the population reaching abundance thresholds at 5, 10, 25, 50, and 100 years from the data year (2017).

Figure 11. Model projections (10,000) of annual females in natural log space for 100 years into the future from 2017 (data year).

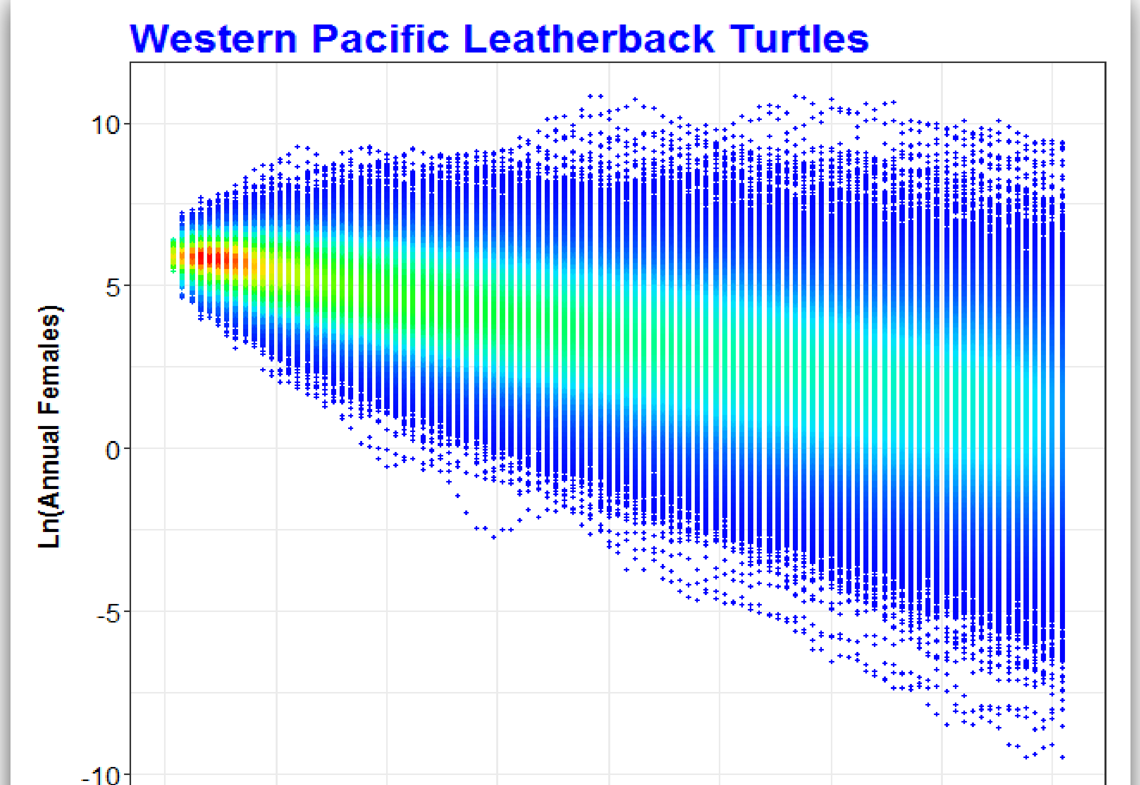


Table 11. Probability of the West Pacific population reaching abundance thresholds within the 100 year projection period, and time in years (mean, median, & 95% credible interval [CI]) to reach the threshold for all runs that fall below the threshold.

Threshold	**Probability of staying above threshold**	**Probability of falling below threshold**	**Years to reach threshold (Mean)**	**Years (Median)**	**Years (lower 95% CI)**	**Years (upper 95%CI)**
50% abundance	2%	98%	26	28	3	86
25% abundance	5%	95%	37	31	7	91
12.5% abundance	9%	91%	46	43	13	95

Table 12. Probability (with 95% credible intervals [CI]) of the West Pacific population reaching abundance thresholds at 5, 10, 25, 50, and 100 years from final data year (2017).

Abundance Threshold	**Yr 5 (%)**	**Yr 10 (%)**	**Yr 25 (%)**	**Yr 50 (%)**	**Yr 100 (%)**
50%	12	36	68	87	98
Lower 95% CI	11	35	67	87	97
Upper 95% CI	13	36	69	88	98
25%	0	8	45	76	95
Lower 95% CI	0	7	44	75	95
Upper 95% CI	1	8	46	76	96
12.5%	0	1	22	61	91
Lower 95% CI	0	1	21	60	90
Upper 95% CI	0	1	23	62	91

Finally, the Bayesian model estimates provide an index of population growth using the best available index of population abundance derived from nest count data. Estimating true population growth rates would require additional data or assumptions about the population age distribution, which are not available. Therefore these growth rates reflect the long-term trend estimated using the annual female count data described above. For *r*, if values are positive then growth is positive; if values are negative then growth is negative. For λ lambda, if values are greater than 1 then growth is positive; if values are less than 1 then growth is negative. The estimates indicate that the West Pacific Ocean leatherback sea turtle population growth is negative.

Log growth rate of population (r):

mean: -0.053
median: -0.053
variance: 0.003
95% CI: -0.164 to 0.059

Finite rate of increase (λ):

mean: 0.949
median: 0.948
95% CI: 0.849 to 1.061

East Pacific Ocean Population

The decline of the East Pacific leatherback population during the past two decades has been extensively documented, and was identified as one of the eleven most endangered regional management units in the world (Wallace et al. 2011). Comprehensive reviews of long-term nesting abundance in Mexico (Sarti Martínez et al. 2007) and Costa Rica (Santidrián Tomillo et al. 2007), which together comprise nearly 90% of all Eastern Pacific leatherback nesting concluded that nesting has declined more than 90% since the 1980s (Spotila et al. 2000, Pritchard 1982, MTSG 2012), from thousands of nesting females per year to no more than 1,000 adult females total in the population, with approximately 150 to 200 females nesting annually per year at primary and secondary nesting beaches (NMFS and FWS 2013; MTSG 2012). Wallace et al. 2013 analyzed the population trends and determined that the population has declined by 97.4% during the past three generations and predict the population will decline by 99.9% over the next generation (2040), which will be fewer than 30 adult females total.

In the East Pacific, major nesting beaches are found in Costa Rica, Mexico, and Nicaragua. At Las Baulas National Marine Park, Costa Rica, which consists of Playa Grande and the smaller nesting beaches of Playa Langosta and Playa Ventanas, Santidrián Tomillo et al. (2007) analyzed data for the area and reported that leatherback numbers declined over 15 years of monitoring (1988-1989 to 2003-2004) with approximately 1,504 females nesting in 1988-1989 to an average of 188 females nesting in 2000-2001 and 2003-2004. An earlier estimate of nesting females at just Playa Grande showed a steady drop from 1,367 females in 1988-1989 to 506 in 1994-1995, and down to 117 by 1998-1999 (Spotila et al. 2000). In Pacific Mexico, Pritchard (1982) conducted an aerial survey of the coastline and derived an estimate of several thousands of nesting females. Although nesting occurs at many sites along the coast (e.g., Tomatal Beach: Vannini and Rosales Jaillet 2009), monitoring on four primary index beaches (Mexiquillo, Tierra Colorada, Cahuitán, Barra de la Cruz) for over 20 years (1982-2004) has shown a decline in nest numbers. Tens of thousands of nests were likely laid on the beaches in the 1980s, but during the 2003-2004 season a total of 120 nests was recorded on the four primary index beaches combined (Sarti Martinez et al. 2007). In Pacific Nicaragua, a monitoring and protection program was established in 2002 at Veracruz beach, followed by Juan Venado in 2004, and Salamina in 2008 (Urteaga et al. 2012). From 2002 to 2010, 420 nests were recorded and 48 individual females were identified. Nesting numbers have decreased since 2006 (Urteaga et al. 2012).

Mexico

At one time Mexico had the largest nesting population in the world with approximately 75,000 reproductively active females (Pritchard 1982). However, Sarti Martinez et al. (2007) recorded

120 nests (combined) at four index monitoring sites during 2002–2003 nesting season. During the 1980s, 30% of the nesting females per season were remigrants, but since the mid-1990s, there has been very little evidence of remigration (Sarti et al. 2000). During the 1999-2000 and 2000-01 nesting seasons, only a small increment in the number of remigrant turtles was observed (Sarti Martinez 2002).

Proyecto Laúd (Project Leatherback) works to survey and gather nesting information of the four index beaches (Mexiquillo, Tierra Colorada, Cahuitán and Barra de la Cruz) to estimate the size of the nesting population, evaluate nesting distribution along the Mexican Pacific (including at secondary non-index sites), and address threats and protect nests where possible. During 2011-2012 nesting season, 98 females were identified at the monitored beaches that laid a total of 386 nests at the four index sites, and 103 nests at secondary beaches (Lopez et al. 2012). Sarti Martinez et al. (2007) estimated that about 43% of the total leatherback nesting in the Mexican Pacific occurs on the four index beaches. If this proportion is consistent, then about 1,137 leatherback clutches were laid along the Mexican Pacific during the 2011-2012 nesting season. Considering an estimated clutch frequency of four, an estimated 284 females may have nested along the Pacific coast of Mexico during 2011-2012, just five less than the 2010-2011 season (Lopez et al. 2012).

Costa Rica

During the 1980s researchers realized that the beaches of Playa Grande, Playa Ventanas and Playa Langosta collectively hosted the largest remaining Pacific leatherback populations in Costa Rica. Since 1988, leatherback turtles have been studied at Playa Grande (in Las Baulas), the fourth largest leatherback nesting colony in the world. During the 1988-89 season (July-June), 1,367 leatherback turtles nested on this beach, and by the 1998-1999 season only 117 leatherback turtles nested (Spotila et al. 2000). The 2003-2004 nesting season showed an increase in nesting abundance from the previous two seasons with an estimated 159 females nested at Playa Grande in 2003-2004 up from 69 and 55 in 2001-2002 and 2002-2003, respectively.

Researchers began tagging females at Playa Grande in 1994. Since then, tagged leatherbacks have had a low return rate (between 16% and 25% in the five or six years following tagging). Spotila et al. (2000) calculated a mean annual mortality rate of 35% for leatherbacks nesting at Las Baulas, and revised to 22% by Santidrian Tomillo et al. (2007). For comparison, at St. Croix, US Virgin Islands nesting grounds, female leatherbacks returned approximately 60% over the same period with a mortality rate of 11% (Reina et al. 2002, Dutton et al. 2005). Thus, comparatively few leatherback turtles are returning to nest on east Pacific nesting beaches and it is likely that Eastern Pacific leatherback turtles are experiencing abnormally high mortalities during non-nesting years. Santidrian Tomillo et al. (2007) confirmed that the number of leatherback turtles nesting at Las Baulas, Costa Rica has declined precipitously over 15 years of monitoring (1988–1989 to 2003–2004), with approximately 1,504 females nesting in 1998-1989 to an average of 188 females nesting in 2000-2001 and 2003-2004, and an estimated annual survival rate of 0.78 (95% CI: 0.75–0.80) or 22% of adults annually. Hence, mortality rates for juveniles and sub adults in the ocean appear to be double those of a stable population. Poaching of turtle eggs at Las Baulas began in the 1950s and continued under intense pressure, with over 90% of eggs harvested before 1991 when the park was established (Santidrian Tomillo et al. 2008). During the 1993-94 nesting season, poaching was reduced and accounted for a loss of only 1.3% of nests on Playa Grande (Santidrian Tomillo et al. 2007). Other losses were due to predation, tidal effects and failure in egg development or infestation by maggots (Schwandt et al. 1996). Bell et al. (2003) found that while leatherbacks at Playa Grande had a high rate of fertility (mean = 93.3% ± 2.5%), embryonic death was the main cause of low hatchling success in this population.

Nicaragua

Nicaragua encompasses the third largest leatherback nesting concentration in the Eastern Pacific after Mexico and Costa Rica. Primary nesting beaches in Nicaragua include Veracruz, Jan Venado and Salamina with Veracruz having the longest time series of monitoring beginning in 2002. At all of these beaches, 100% of nests were harvested prior to implementation of the nesting beach program. An aerial survey conducted during the 1998-1999 season estimated a nesting density in Playa El Mogote of only 0.72 turtles per kilometer (IAC 2004). During the 2005-2006 nesting season, 488 nests were recorded during aerial surveys (Barragan 2006 in Urteaga et al. 2012). Between 2002 and 2010, a total of 420 leatherback nests were recorded and 48 individual females were identified (Urteaga et al. 2012). Of nests laid, approximately 94% were protected from harvest and a total of 48 individual females have been tagged. Unfortunately, the 2009-2010 nesting season was the lowest compared to the previous three nesting season. Overall, the monitoring period is too short to determine a population trend.

4.2.1.4 Threats to the Species

Natural Threats

Natural factors, including the 2004 tsunami in the Indian Ocean (see detailed report by Hamann et al. 2006c) 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). Shifting mudflats in the Guianas have also made nesting habitat unsuitable (Crossland 2003, Goverse and Hilterman 2003).

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

Anthropogenic Threats

Major anthropogenic threats to the species, are fisheries bycatch, direct harvest, alteration of nesting habitat, and predation, which are briefly described below. In addition, habitat changes attributed to changing environmental conditions (i.e., sand temperatures that result in mortality or changes in sex ratios, erosion) and marine debris are also threats to this species (Tiwari et al. 2013), and are described below. Primary impacts to the West Pacific Ocean population in addition to U.S. 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). 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 2013a; Bellagio Steering Committee 2008). Since 2002 the HI SSLL and DSLL fisheries together have captured approximately 251 leatherbacks with an estimated mortality of 83. All of the leatherback sea turtles captured and sampled, except two, are from the West Pacific Ocean population (Dutton pers comm 2018). Observers provide approximate lengths and all estimates have been in the 4-6 ft range. Based on the reported approximate lengths they are about 120-180 cm straight carapace length (SCL), which are sub-adult to adult (Jones pers comm. 2018). The gender of the turtles could not be determined since all of the turtles caught in the HI SSLL fishery are too large to bring aboard.

Between 2006, when the observer program started in American Samoa, and 2017 the American Samoa longline fishery is estimated to have had 50 interactions (35 mortalities) (NMFS 2018). 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 lifestages and ranged from 106.7 cm- 150 cm SCL. The gender was not determined for any of the turtles captured.

The U.S. purse seine fleet is anticipated to interact with 11 leatherback sea turtles annually with zero mortality (NMFS 2006). Between 2008 and 2015 the fishery had an estimate of 16 interactions with leatherbacks, and no mortalities (NMFS unpublished).

The cumulative bycatch estimates for the U.S. 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 analysis indicated almost all of these turtles originated from the West Pacific Ocean population (Dutton et al. 2000).

Although the causes for decline of the eastern and West Pacific Ocean leatherback sea turtle populations are not entirely clear, it is likely the result of historic intensive egg poaching on the nesting beaches, incidental capture of adults and juveniles in fisheries, and natural fluctuations due to changing environmental conditions that influence prey abundance and distribution (Sarti Martinez et al. 2007; Santidrian Tomillo et al. 2007, 2008; Wallace et al. 2010b; Saba et al. 2012). In Costa Rica the emergence of new threats from coastal development on key leatherback sea turtle nesting areas present a serious challenge to efforts to protect leatherback sea turtles in the East Pacific (Wallace and Piedra 2012; Tapilatu et al. 2013; NMFS and FWS 2013a), and egg consumption by humans and domestic animals (e.g., dogs) persist on Nicaragua nesting beaches where protection is incomplete (Urteaga et al. 2012). However, fisheries bycatch is still considered the major obstacle to population recovery (Wallace and Saba 2009; NMFS and FWS 2013a; MTSG 2012).

Eckert and Sarti (1997) speculated that the swordfish gillnet fisheries in Peru and Chile contributed to the decline of the leatherback sea turtle in the Eastern Pacific as the decline in the nesting population at Mexiquillo, Mexico occurred at the same time that effort doubled in the Chilean driftnet fishery; although ongoing leatherback sea turtle bycatch in gillnet and longline fisheries of South America off Peru and Chile continues to impact adults and subadults (Alfaro-Shigueto et al. 2007, 2011; Donoso and Dutton 2010). An assessment of fisheries bycatch impacts on sea turtle populations globally found that bycatch in net gear appears to have the highest population-level impact on the East Pacific population, followed by longlines (Wallace et al. 2013). Roe et al. (2014) highlight potential longline fishery bycatch hotspots in the Pacific that may affect populations at various stages of their life history. For West Pacific Ocean nesting populations, several areas of high risk were identified in the north and central Pacific, with the greatest risk adjacent to primary nesting beaches in tropical seas of Indo-Pacific islands, in the Sulu, Sulawesi, and South China Seas. In the central Pacific region, the largest bycatch risk area was predicted to occur southwest of the Hawaiian Islands, between the equator and up to 15°–20° N, from January through March. The North Pacific Transition Zone between 30° N and 35° N poses moderate risk to leatherbacks at this time. From October through December there is a moderate risk from 140° W to 120° W, and patchy areas of risk broadly distributed throughout the North Pacific from April through December (Roe et al 2014) . For Eastern Pacific nesting populations, the greatest risk was identified in the South Pacific Gyre.

The 2004 management measures have proven to reduce leatherback sea turtle interaction rates by 83% (Gilman et al. 2007a; WPFMC 2009b). Since the HI SSLL fishery re-opened in 2004, an 22 estimated leatherback sea turtles have died. All of the leatherback sea turtles caught were released alive; mortality estimates come from applying the NMFS post-hooking mortality criteria (Ryder et al. 2006) to interactions.

NMFS (2001) regulations implemented a large Central California time/area closure that significantly reduced leatherback interactions in this fishery by approximately 80%. NMFS has continued to refine management measures in this fishery and the current authorized take is up to three leatherbacks annually, or up to 10 leatherback interactions (or seven mortalities) over a five year period (NMFS 2013). Proposed (NMFS 2018) management measures would include limits (“hard cap”) on interactions whereby the fishery would cease immediately if three leatherback turtle takes occur or one observed mortality.

There are around nine fixed gear fisheries operating off the U.S. west coast, which include the California state-managed Dungeness Crab Fishery and the federally-managed Sablefish Pot Fishery. Interactions in the Sablefish fishery is managed by NMFS (2013); there was one documented interaction in 2008. The CA 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 CA 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 2018).

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; IOSEA 2012; Hitipeuw & Lawalata 2006, 2008). 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 (1994) brought attention to this hunt when they reported that approximately 200 animals were harpooned in just three months of 1994 (Table 6), with as many as 13 taken in one day. Over the past three decades, sporadic monitoring efforts have ensued to quantify and assess the take. Such efforts have resulted in estimates of up to 100 individuals taken per year (Suarez and Starbird 1996; Hitipeuw & 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 2013a); however, recent enumerator surveys indicate that harvest continues with conservative estimates of 431 takes over the past 8 years (53.9/yr) and at least 103 leatherbacks harvested in 2017 (Table 13) (WWF 2018). The size range of reported leatherback takes 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 (WWF 2006).

Table 13. Number of direct takes in the Kei Kecil Islands documented since 1994.

Timeframe	Number harvested	Source
3 month period in 1994	200	Starbird 1994
annually	100	Suarez and Starbird 1996
2003-2006	100	WWF 2006
Over 8 years	431	Hittipeuw and Lawalata 2008
2017	103	WWF 2018

Destruction and alteration of leatherback sea turtle nesting habitats are occurring throughout the species' global range, especially coastal development, village sprawl, beach armoring, beachfront lighting, and vehicular/ pedestrian traffic. Coastal development includes roads, buildings, seawalls, etc., all of which reduce suitability of nesting beaches for nesting by reducing beach size. Beach armoring is typically done to protect coastal development from erosion during storms, but armoring blocks turtles from accessing nesting areas and often leads to beach loss. Coastal development and village sprawl also increases artificial lighting, which may disorient emerging hatchlings, causing them to crawl inland towards lights instead of seaward. Coastal development also improves beach access for humans, resulting in more vehicular and foot traffic on beaches, causing compaction of nests and reducing emergence success. Fortunately, some major nesting beaches for leatherback sea turtles, including those for the West Pacific Ocean population, occur in remote areas where development as described above is less prevalent although timber harvest, road construction, and village sprawl remain an issue in these remote areas (NMFS and FWS 2013a; Bellagio Steering Committee 2008; PLAWG 2012).

Harvest of leatherback sea turtles for their meat and eggs has resulted in the extirpation of major nesting aggregations, such as what occurred in the 1980s and 90s in Malaysia and Mexico due to egg collection (and likely exacerbated by simultaneous mortality of adults due to fisheries bycatch). Globally, harvest is reduced from previous levels, but egg harvest continues throughout the range of the West Pacific Ocean population, including hunting of adults near primary nesting beaches and in foraging habitats (i.e., Kei Islands, Indonesia: Starbird and Suarez 1996; Bellagio Steering Committee 2008). Predation of eggs is a major problem for West and East Pacific Ocean leatherback sea turtle, for example by feral pigs in Papua Barat and feral dogs in PNG (NMFS and FWS 2013a; Bellagio Steering Committee 2008). Impacts and threats to leatherback sea turtle conservation and recovery in Papua Barat include: exploitation of turtles and eggs, chronically low hatchling production as a result of predation (pigs, dogs, and monitor lizards), inundation, beach erosion, and lethal incubation temperatures (Starbird and Suarez 1996; Hitipeuw et al. 2007; Tapilatu and Tiwari 2007; Bellagio Steering Committee 2008; PLAWG 2012; NMFS and FWS 2013a). While efforts are underway to coordinate and standardize conservation and monitoring work, there is a need to establish an advisory committee consisting of local stakeholders and to encourage local management authorities to become actively engaged in oversight of nesting beach programs (Bellagio Steering Committee 2008). Despite successes achieved through the HCLTCP in PNG described previously, information indicates continuing impacts to leatherback sea turtles from egg and adult harvest and domestic dog predation in Huon coast communities not part of the project, along with continuing broad-scale impacts from beach erosion, wave inundation, and village sprawl (Bellagio Steering Committee 2008; Pilcher 2009). In Vanuatu and Solomon Islands adult leatherback sea turtles are opportunistically hunted for meat in some areas and leatherback sea turtle eggs are occasionally collected from these beaches, and beach erosion is a serious impact to nesting beaches (Bellagio Steering Committee 2008; NMFS 2008a; NMFS and USFWS 2013a).

Marine debris may be a source of mortality to all species of sea turtles, as small debris can be ingested and larger debris can entangle animals, leading to death. Manmade materials such as plastics, micro plastics, and derelict fishing gear (e.g., ghost nets) that may impact leatherbacks via ingestion or entanglement can reduce food intake and digestive capacity, cause distress and/or drowning, expose turtles to contaminants, and in some cases, cause direct mortality (Balazs 1985; Bjorndal et al. 1994; Keller et al. 2004; Arthur et al. 2009; Wabnitz and Nichols 2010; Parker et al. 2011). While the impact of marine debris on leatherbacks during their pelagic life stage is currently unquantified, it is likely that impacts may be severe, given the increase of plastics and other debris and pollution entering the marine environment over the past 20-30 years. Schuyler et al. (2015) and Wedemeyer-Strombel et al. (2015) documented significant amounts of ingested debris in sea turtles sampled, and although olive riddles were found to have the highest risk of ingestion, other species, including leatherbacks, have similar overall risk.

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 USFWS 2013a). Over the long-term, climate change-related impacts will likely influence biological trajectories in the future on a century scale (Paremsan and Yohe 2003). The study by Polovina et al. (2011), indicates that primary production in the southern biome and in the California current ecosystem are expected to increase by the end of the century (Rykaczewski and Dunne 2010), which may benefit leatherback sea turtles. Increases in their primary prey source, sea jellies, due to ocean warming and other factors are likely (Brodeur et al. 1999; Attrill et al. 2007; Richardson et al. 2009), although there is no evidence that any leatherback sea turtle populations are currently food-limited. Even though there may be a foraging benefit to leatherback sea turtles due to climate change influence on productivity, we do not know what impact other climate-related changes may have such as increasing sand temperatures, sea level rise, and increased storm events. However, a different picture is predicted for Eastern Pacific leatherback turtles. Modeling of climate projections and population dynamics resulted in an estimated 7% per decade decline in the Costa Rica nesting population over the twenty first century. Whereas changes in ocean conditions had a small effect on the population, the increase of 2.5^0 C warming of the nesting beach was the primary driver of the modeled decline through reduced hatching success and hatchling emergence rates (Saba et al. 2012). Furthermore, climate change may compound the effects of interannual climate variability, as governed by El Niño Southern Oscillation (ENSO). Saba et al. (2007) showed that nesting females in Costa Rica exhibited a strong sensitivity to ENSO where as cool La Niña events correspond with a higher remigration probability and warm El Niño events correspond with a lower remigration probability. As a result, productivity at leatherback sea turtle foraging areas in the Eastern Pacific in response to El Niño/La Niña events result in variable remigration intervals and thus variable annual egg production. This phenomenon may render the Eastern Pacific leatherback sea turtle population more vulnerable to anthropogenic mortality due to longer exposure to fisheries than other populations (Saba et al. 2007).

4.2.1.5 Conservation

Considerable effort has been made since the 1980s to document and address leatherback sea turtle bycatch in fisheries around the world. In the U.S., observer programs have been implemented in most federally-managed fisheries to collect bycatch data, and several strategies have been pursued to reduce both bycatch rates and post-hooking mortality. These include developing gear solutions to prevent or reduce capture (e.g., circle hooks) or to allow turtles to escape without harm (e.g., turtle exclusion devices, which may be too small for adult leatherback sea turtles), implementing seasonal time-area closures to prevent fishing when turtles are congregated, modifying existing gear (e.g., reducing mesh size of gillnets), and developing and promoting Sea Turtle Handling Guidelines (NMFS and FWS 2013a). For example, switching to large circle hooks and mackerel bait in 2004 with complimentary fishery-based outreach and education resulted in an approximately 83% reduction in the leatherback sea turtle interaction rate in the HI SSLL fishery (Gilman et al. 2007a; WPFMC 2009b). Protected species workshops are required by NMFS annually of all Hawaii longline vessel operators to provide refresher trainings on the proper sea turtle handling guidelines, among other things. PIR offices in particular, have supported a significant number of international fishery-based projects to identify and promote effective sea turtle bycatch mitigation measures (e.g., circle hooks) or other gear modifications. In the Pacific, such projects have occurred in: Indonesia, Vietnam, Papua New Guinea, Solomon Islands, Malaysia, Palau, Marshall Islands, Federated States of Micronesia, and throughout Latin America in association with the IATTC. Much of this work has been coupled with capacity-building, training, and implementation of regional observer programs aimed to improve the quality of catch and bycatch information from international fleets to better address the requirements of RFMO Conservation and Management Measures (CMMs) (of the WCPFC and IATTC). NMFS together with other regional partners will continue working within the context of RFMOs and U.S. laws to modify and improve international sea turtle bycatch mitigation requirements.

NMFS and partners have been involved in leatherback sea turtle research and conservation activities in the Western Pacific for nearly a decade supporting projects to understand and bolster survivorship, reduce harvest or predation, and to address other priority actions identified in the U.S. Pacific Leatherback Sea Turtle Recovery Plan (NMFS and FWS 1998b). Efforts to recover leatherback sea turtles have been hampered by naturally occurring phenomena, including seasonal spring tide inundation of nests and large earthquakes. A myriad of land ownership, beach access, and local village politics have also hampered monitoring and conservation efforts in all countries. NMFS continues to work toward achieving support and developing fruitful partnerships for leatherback sea turtle conservation throughout the region and has made substantial progress toward understanding population structure and threats. Progress has been achieved by building capacity among international colleagues, implementing studies on the economics of conservation, engaging and supporting nesting beach conservation activities and mitigation measures that include hatching success studies, implementing and encouraging Passive Integrated Transponder (PIT) tagging as a necessary tool to determine annual nesting estimates, undertaking aerial surveys and satellite telemetry research to assess habitat use, and utilizing innovative molecular techniques (genetics and stable isotopes) to assess stock structure and connectivity.

Community-based village rangers at Wermon and Jamursba-Medi in Papua Barat have been hired over the past decade to collect population demographic data (tag turtles and record nesting activity). Through their presence on the beach, projects have been able to guard leatherback sea turtle nests from predation by feral pigs and egg collectors. In Wermon, for example, during the 2006-07 nesting season the project used a few bamboo grids over nests as protection from dog predation (Bellagio Steering Committee 2008); a conservation strategy that has proven effective in PNG (Pilcher 2006). Prior to 2002, 100% of nests laid at Wermon beach were lost as a result of harvest (60%) or predation (40%) (Starbird and Suarez 1996). Therefore, as a result of monitoring efforts the Wermon project may have protected over 12,000 nests that have been laid since the project's inception (NMFS 2011b). Community support in the form of scholarships and church repairs has been provided to encourage local participation in leatherback sea turtle conservation. Other community-based initiatives have been supported and coordinated among the groups working in Papua. This includes socioeconomic research to better understand how to build community capacity to support leatherback sea turtle conservation, and workshops convened to help the leatherback sea turtle conservation program to develop stronger ties between the program and communities (Gjertsen and Pakiding 2012). From 2003 to 2007, the WPFMC supported a project at the Kei Kecil Islands of Papua Barat Indonesia to assess and help reduce traditional harvest of adult leatherback sea turtles in coastal foraging habitats. Starbird and Suarez (1996) estimated that this traditional fishery captured at least 100 leatherback sea turtles per year; however, the Kei Islands project acquired a more accurate harvest estimate of less than 50 turtles per year with the majority being juveniles or subadults (Lawalata and Hitipeuw 2006). In July 2012, Kei Kecil was designated as a marine protected area. It is hoped that the designation, which was supported by a program of USAID aimed at conserving marine resources, will help to address threats posed by forest clearance near nesting beaches, fishing activities, and hunting of turtles for meat (IOSEA 2013).

In PNG, the community-based HCLTCP monitored nesting activity, implemented conservation measures to protect nests from dog predation (e.g., bamboo grids), and has worked to reduce localized harvest through community development incentives (CDI) since its inception in 2003 through 2013. Through CDI, communities at large experience the benefits of the leatherback sea turtle project over time even if they themselves have not personally gained (financially or otherwise) from the project's existence, but in many cases, may have relinquished resource utilization by agreeing to participate in conservation efforts (i.e. no harvest). CDI projects to date have included repairing or improving fresh water supplies, building or expanding school facilities, repairing traditional village meeting houses, and developing or improving church and aid outpost facilities (Pilcher 2011). As a result, nest predation and harvest of eggs was reduced and hatchling production has increased over time in associated communities from close to 0% to approximately 60-70% as a result of the CDI program and concurrent efforts to implement nest protection measures (Pilcher 2009). During the 2010-2011 nesting season, the average hatching success rate was quantified to be 44.0%, resulting in an overall conservative estimate of 80,000 hatchlings released since the project's inception (Pilcher 2011; NMFS 2011b). Monitoring activities were hampered by community discord and disagreements during the 2013-2014 nesting season and have not resumed.

In the Solomon Islands, a program has been initiated at Sasakolo and Litogarhira to relocate nests that would otherwise be destroyed by beach erosion, high sand temperatures, illegal harvest and predation in order to increase hatchling production (a collaborative project between SWFSC and The Nature Conservancy, with additional funding support from the International Sustainable Seafood Foundation, the Ocean Foundation, and NMFS). Additionally, the Tetepare Descendants Association (TDA) has closed 13 km of beach to harvest, continues to protect and monitor nests, and is obtaining training, guidance and encouragement through collaborations with relevant NMFS staff and from funding. Further, efforts are currently underway to launch assessment and monitoring activities with a community that is reported to have summer nesting activities. Preliminary data from the village of Waisurione on the island of Malaita is reporting approximately 5 nesting females per summer (May – July) nesting season, and villagers are learning to protect nests from dog predation (using bamboo grids) and relocating nests laid in erosion prone areas (Marine Research Foundation 2015; Williams et al. 2014).

In Vanuatu, while leatherback sea turtle nesting is limited or unknown, especially on more remote islands, NMFS has supported a local NGO, Wan Smolbag, intermittently since 2007 through 2015 to train local villagers to monitor nesting activity, conserve leatherback sea turtle nesting beaches, and educate local communities to protect leatherback sea turtles and their nests from direct harvest of nesting females and their eggs (Petro 2011; Aromalo and MacKay 2015).

In Mexico, most conservation programs aimed at protecting nesting sea turtles have continued since the early 1980s, and there is little information on the degree of poaching prior to the establishment of these programs. Since the Mexican government instituted protective measures, there has been greater nest protection and nest success. The most recent results (2011-2012 nesting season) indicate that nearly 90% of clutches laid in key index beaches in Mexico were protected (e.g., relocated to hatcheries), with Barra de la Cruz nesting beach receiving the greatest number of nests and the largest number of nests protected (99% or 142 nests). On the priority II beaches during the 2011-12 nesting season, nest protection ranged from 70.6% to 78.8% (López et al. 2012). This is a significant increase since 1996, when only 12% of nests were relocated. From 1982 to 2004 a total of 270,129 leatherback sea turtle hatchlings were released to the wild population (Sarti Martinez et al. 2007). Currently, hope for the future of the population relies on the protection of clutches laid on the priority beaches, the participation of local communities in conservation activities, and increased awareness of the leatherback sea turtle's status among Mexican society.

In Costa Rica, a systematic system of poaching was well established by 1975 that resulted in the removal of 90% of eggs. Intense poaching lasted until 1991, when Las Baulas park was established (Santidrian-Tomillo et al. 2007). Overall, environmental education and conservation efforts through active law enforcement have greatly reduced leatherback sea turtle egg poaching in Costa Rica (Chaves et al. 1996). However, Santidrian-Tomillo et al. (2008) concluded that the Costa Rican population still suffers from the effects of historic poaching rates, which is compounded by current and unsustainable 22% adult mortality rates.

In summary, long-term monitoring and conservation programs at the index nesting beaches in Mexico and Costa Rica have essentially eliminated threats from human consumption of eggs and nesting females, and ongoing efforts at important beaches in Nicaragua are increasing in effectiveness (Urteaga et al. 2012). Nonetheless, the abundance of this population remains perilously low, and continues to decrease slowly toward extinction (MTSG 2012). Reducing leatherback sea turtle bycatch has become a primary focus for many conservation projects around the world, and some mitigation efforts are showing promise (Watson et al. 2005; Gilman et al. 2009; Wang et al. 2010). However, fisheries bycatch is still considered the major obstacle to population recovery (Wallace and Saba 2009; Wallace et al. 2013; NMFS and USFWS 2013a).

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 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 Conservation and Management Measure (CMM 2008-03) to mitigate the impacts on turtles from longline swordfish fisheries in the Western Central Pacific Ocean. The measure includes the adoption of FAO guidelines to reduce sea turtle mortality through safe handling practices and to reduce bycatch by implementing one of three methods by January 2010. The three methods to choose from are: 1) use only large circle hooks; 2) use whole finfish bait; or 3) use any other mitigation plan or activity that has been approved by the Commission. 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 and an increasing number of community-based initiatives are in place to reduce the take of turtles in foraging areas (Gilman et al. 2007b, NMFS and FWS 2013a).

4.2.1.6 Summary of the Status of the Leatherback Sea Turtle

In this section of this biological opinion, we explained that the leatherback sea turtle is endangered throughout its range, and that the Pacific regional populations (East Pacific and West Pacific) are critically endangered, and one subpopulation is functionally extinct (Terengganu, Malaysia). We used our knowledge of the species' demography and population ecology to capture the primary factors that appear to determine the leatherback sea turtle's population dynamics in a causal loop diagram (Figure 12). Where we could, we included in our causal loop diagram a graphic to represent the behavior of the reference variables (data) that influenced the behavior of that particular loop.

Pacific Ocean leatherback sea turtle populations are declining at rapid rates, although in some years, counts of nesting females might increase. Based on NMFS' Bayesian state-spaced model, the West Pacific Ocean population is declining at about 5% per year, and is at risk of falling to less than half of its current abundance in as few as three years (range 3-86 years, mean 26 years; see Table 4; Jones et al. 2018). The trend taken together with other information summarized in the preceding review of the species' status indicates that the population is at risk of extirpation in the next turtle generation or two (within 30-60 years; Jones et al. 2018).

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. 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 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.

Our graphic illustrates the population behavior of leatherback sea turtles adults (females) in S1, which is based on available nest count data. Because nesting females drive the population dynamics of this species, they serve as a surrogate for all of the life stages in the population. Our diagram of the population dynamics of this species also highlights the two nesting and four foraging strategies to better represent the complexity within the population and to differentiate the stressors that play on the individuals that engage in these different migratory behaviors. S2 references the decline observed in California forager abundance.

When we add in the specific threats we can see which life history stages interact with U.S. fisheries. The HI SSLL fishery interacts with subadults and adult summer nesters that forage in the North Pacific and California. The HI DSLL fishery interacts with juveniles and the larger stages that the HI SSLL fishery also interacts with. The American Samoa longline fleet interacts with juveniles, subadults and adult winter nesters that forage in the south Pacific. These and other threats are represented 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.

Despite these conservation efforts the overall trend of the leatherback sea turtle continues to decline. The Pacific Ocean populations is an asymptopically declining population. 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.

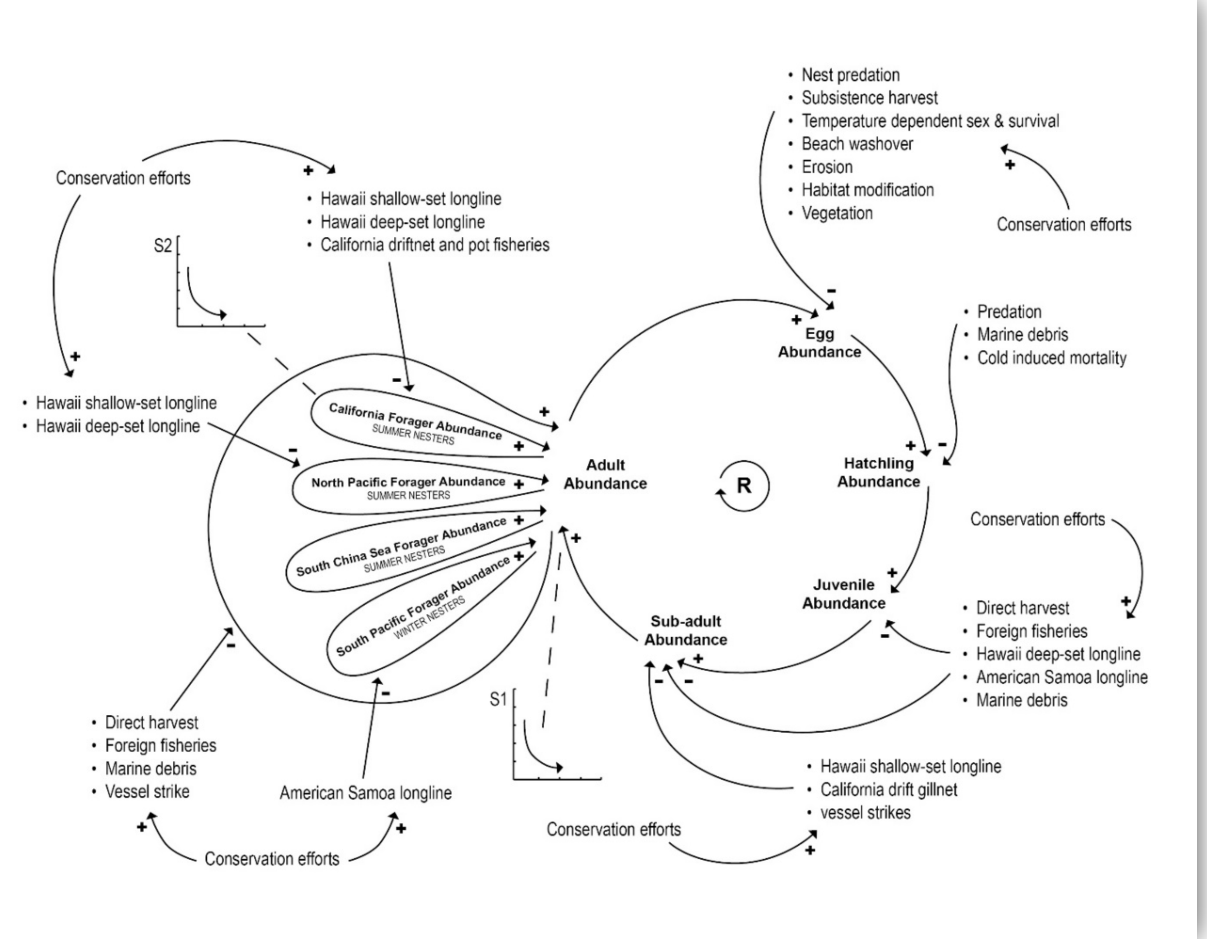


Figure 12. Leatherback sea turtle casual loop diagram depicting various stressor-response relationships for each life cycle stage.

4.2.2 North Pacific Loggerhead Sea Turtle DPS

4.2.2.1 Distribution

Loggerhead sea turtles can be found throughout tropical to temperate waters in the Pacific; however, their breeding grounds include a restricted number of sites. In the Pacific, extensive mtDNA studies show that the northern loggerhead populations are isolated from the southern Pacific populations, and that juveniles from these distinct genetic populations do not disperse across the equator (Hatase et al. 2002a; Dutton 2007, Conant 2009). Loggerhead sea turtles in the action area are from the North Pacific DPS and therefore represent a single species (76 FR 58868).

The North Pacific DPS of loggerheads appears to occupy an ecological setting distinct from other loggerheads, including those of the South Pacific population (DPS). In general, this is the only population of loggerheads to be found north of the equator in the Pacific Ocean, foraging in the Eastern Pacific as far south as Baja California Sur, Mexico (Seminoff et al. 2004, Peckham et al. 2007) and in the Western Pacific as far south as the Philippines (Limpus 2009) and the mouth of Mekong River, Vietnam (Sadoyama et al. 1996). Pelagic juveniles have been found to spend much of their time foraging in the central and eastern North Pacific Ocean.

Differential Distribution

Despite long-distance developmental movements of juvenile loggerheads in the North Pacific, current scientific evidence, based on genetic analysis, flipper tag recoveries, and satellite telemetry, indicates that individuals originating from Japan remain in the North Pacific for their entire life cycle, never crossing the equator or mixing with individuals from the South Pacific (Bowen *et al.* 1995; Hatase *et al.* 2002a; LeRoux and Dutton 2006; Dutton 2007; Boyle *et al.* 2009; Briscoe et al. 2016a, 2016b). This apparent, almost complete separation of two adjacent populations most likely results from: (1) The presence of two distinct Northern and Southern Gyre (current flow) systems in the Pacific (Briggs 1974), (2) near-passive movements of post-hatchlings in these gyres that initially move them farther away from areas of potential mixing among the two populations along the equator, and (3) the nest-site fidelity of adult turtles that prevents turtles from returning to non-natal nesting area

The North Pacific loggerhead sea turtle DPS nests primarily in Japan (Kamezaki et al. 2003), although low level nesting may occur outside of Japan in areas surrounding the South China Sea (Chan et al. 2007; Conant et al. 2009). Along the Japanese coast, nine major nesting beaches (greater than 100 nests per season) and six “submajor” beaches (10–100 nests per season) exist, including Yakushima Island where 40% of nesting occurs (Kamezaki et al. 2003).

Loggerhead sea turtle life history is characterized by early development in the oceanic (pelagic) zone followed by later development in the neritic zone over continental shelves. The oceanic developmental period may last for over a decade, followed by recruitment to the neritic zone of older age classes where maturation is likely reached. In the Central North Pacific Ocean, foraging juvenile loggerhead sea turtles congregate in the boundary between the warm, vertically-stratified, low chlorophyll water of the subtropical gyre; and the vertically-mixed, cool, high chlorophyll transition zone water. The Transition Zone Chlorophyll Front is a favored foraging and developmental habitat for juvenile loggerhead sea turtles (Polovina et al. 2001). Satellite telemetry of loggerhead sea turtles also identified the Kuroshio Extension Current, specifically the Kuroshio Extension Bifurcation Region, as a forage hotspot (Polovina et al. 2006; Kobayashi et al. 2008). The Kuroshio Extension Bifurcation Region is an area of high primary productivity that concentrates zooplankton and other organisms that in turn attract higher trophic level predators, including sea turtles (Polovina et al. 2004). Loggerhead sea turtle habitat in the North Pacific occurs between 28° N. and 40° N. latitude (Polovina et al. 2004) and sea surface temperatures (SST) of 14.45° C to 19.95° C (Kobayashi et al. 2008), but is highly correlated at the 17/18°C isotherm (Howell et al. 2008). An unknown proportion of juveniles eventually transition from the Central North Pacific to the Eastern Pacific where a major foraging aggregation has been documented along the Pacific coast of the Baja California Peninsula (BCP), Mexico (Peckham et al. 2008; Seminoff et al. 2014). Loggerhead sea turtles from the North Pacific DPS have also been documented in the Gulf of California, Mexico (Seminoff et al. 2004; Zavala et al. 2017) and along the coast of southern California, during warm water periods (Allen et al. 2013; Eguchi et al. In-Review).

Baja California Sur, Mexico is an important foraging area for juvenile turtle loggerhead sea turtles (Peckham and Nichols 2006; Peckham et al. 2007; Conant et al. 2009; Wingfield et al. 2011). Abundance estimates by Seminoff et al. (2014) indicated that upwards of 43,000 loggerhead sea turtles may occur in the Gulf of Ulloa, which likely represents a significant portion of the entire North Pacific loggerhead sea turtle DPS. After spending years foraging, potentially two decades (Tomaszewicz et al. 2014), in the central and Eastern Pacific, loggerhead sea turtles return to their natal beaches for reproduction (Resendiz et al. 1998; Nichols et al. 2000) and remain in the Western Pacific for the remainder of their life cycle (Iwamoto et al. 1985; Kamezaki et al. 1997; Conant et al. 2009; Hatase et al. 2002; Ishihara et al. 2011). The East China Sea is a major habitat for post-nesting adult females based on tag-recapture studies (Iwamoto et al. 1985; Kamezaki et al. 1997, 2003; Kobayashi et al. 2008, 2011).

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).

Loggerheads are able to exist on a wide variety of food items with ontogenetic and regional differences in diet. Loggerhead diets have been described from just a few coastal regions, and very little information is available about differences or similarities in diet at various life stages. Very little is known of the diet of oceanic juveniles. Data collected from stomach samples of juvenile loggerhead sea turtles indicate a diverse diet of pelagic food items (NMFS 2006; Parker et al. 2005). They are primarily carnivorous but they also consume some algae as well (Bjorndal 1997; Jones and Seminoff 2013). Loggerheads that feed off the Baja California Peninsula feed on higher trophic level organisms and benthic organisms, like crabs (Parker et al. 2005; Peckham et al. 2011).

4.2.2.2 Population Dynamics

Structure

Pacific loggerhead sea turtles are further partitioned evolutionarily from other loggerhead sea turtles throughout the world based on additional analyses of mtDNA. The haplotypes (a haplotype refers to the genetic signature, coded in mtDNA, of an individual) from both North and South Pacific loggerheads are distinguished by a minimum genetic distance (d) equal to 0.017 from other conspecifics, which indicates isolation of approximately one million years (Bowen 2003). Within the Pacific, Bowen *et al.* (1995) used mtDNA to identify two genetically distinct nesting populations in the Pacific—a northern hemisphere population (North Pacific DPS) nesting in Japan and a southern hemisphere population nesting primarily in Australia (South Pacific DPS).

Loggerhead sea turtles in the action area are from the North Pacific DPS. This DPS is comprised of turtles from Japan's nesting aggregations, which based on significant differences in mtDNA haplotype frequencies, is comprised of at least three putative subpopulations represented by the three distinct rookeries identified by Matsuzawa et al. (2016):

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

The genetic data that define the three subpopulations are characterized by at least two common mtDNA haplotypes that occur at different frequencies but have too much overlap to differentiate using a mixed stock analysis from the sample size from the HI SSLL fishery (Dutton pers comm. May 4, 2018). The Ryuku subpopulation is the smallest and the most genetically distinct and there were no signs that the Ryuku subpopulation was caught disproportionately (Dutton pers comm. May 4, 2018). Results from a (weighted and non) mixed stock analysis of 213 loggerheads sampled from the HI SSLL fishery since 1995 are shown in Table 14 and Table 15 and should be interpreted cautiously due to a small sample size.

Table 14. Flat priors mixed stock analysis for Loggerhead sea turtles (Dutton pers comm. May 4, 2018).

Stock	**Mean**	**2.5%**	**97.5%**
Mainland	0.5538	0.0078	0.9773
Yakushima	0.2630	0.0000	0.9444
Ryuku	0.1832	0.0000	0.6686

Table 15. Weighted Priors based on population size for loggerhead sea turtles (Dutton pers comm. May 4, 2018).

Stock	Mean	2.5%	97.5%
Mainland	0.5019	0.0012	0.9769
Yakushima	0.4055	0.0015	0.9897
Ryuku	0.0926	0.0000	0.5245

Through stable isotope analysis a recent study demonstrated that loggerheads recruit to the neritic area at different ages, meaning some have longer oceanic stages with less time in the neritic environment and some have longer time in the neritic zone versus the oceanic environment (Tomaszewicz et al. 2017). We do not know if there is a difference in the population structure or if it is more of an environmental cue that plays a role in when loggerhead sea turtles recruit to the neritic environment. The two areas have different food sources which may influence growth rates and perhaps survival rates based on different threats. At this time we do not know if the different foraging strategies at the different stages are related and perhaps represent different subpopulations.

Diversity

Loggerheads nest on ocean beaches and occasionally on estuarine shorelines. Although specific characteristics vary between rookeries, loggerhead nesting beaches tend to be wide, sandy beaches backed by low dunes and fronted by a flat, sandy approach from the water (Miller et al. 2003). Nests are typically laid between the high tide line and the dune front (Routa 1968; Witherington 1986; Hailman and Elowson 1992).

Sea turtle eggs require a high-humidity substrate that allows for sufficient gas exchange and temperatures conducive to egg development (Miller 1997; Miller et al. 2003). Mean clutch size varies greatly between populations, but on average is approximately 100-130 eggs per clutch (Dodd 1988). Loggerhead nests incubate for variable periods of time. The length of the incubation period is inversely related to nest temperature, such that between 26°C and 32°C, a change of 1°C adds or subtracts approximately 5 days (Mrosovsky 1980). The warmer the sand surrounding the egg chamber, the faster the embryos develop (Mrosovsky and Yntema 1980). Sand temperatures prevailing during the middle third of the incubation period also determine the sex of hatchlings (Mrosovsky and Yntema 1980). Incubation temperatures near the upper end of the tolerable range produce only female hatchlings while incubation temperatures near the lower end of the tolerable range produce only male hatchlings. The pivotal temperature (i.e., the incubation temperature that produces equal numbers of males and females) in loggerheads is approximately 29°C (Limpus et al. 1983; Mrosovsky 1988; Marcovaldi et al. 1997). Moisture conditions in the nest influence incubation period, hatching success, and hatchling size (McGehee 1990; Carthy et al. 2003). Additionally, soil composition has been shown to influence embryonic development (Marco et al. 2017).

A number of life history variables were used in the Susceptibility to Quasi-Extinction (SQE) analysis of the Status Review (Conant et al. 2009). Hatase et al. (2013) published updated life history parameters (Table 16) based on 362 nesting females over a 26-year period from the

Yakushima Island rookery in Japan, and is considered the most current data at the time of this analysis for this specific DPS.

Table 16. Comparison of body size and fecundity (clutch size and clutch frequency) between recruits (first-time nesters) and remigrants (experienced nesters) of the loggerhead turtle (Caretta caretta) nesting at Yakushima Island, Japan, in 1999, 2008, and 2011. Source: Table 2 pg. 2588, Hatase et al. 2013.

	Oceanic	Oceanic	Oceanic	Neritic	Neritic	Neritic	
Parameter	Mean ± SD	Range	*n*	Mean ± SD	Range	*n*	*P*
Straight carapace length (mm)	791 ± 36	715-902	58	859 ± 41	729-968	282	<0.0001
Clutch size (no. eggs)	103.2 ± 15.6	78.0-134.5	22	115.5 ± 19.8	64.0-164.0	98	<0.005
Emergence success (%)	64.8 ± 13.3	43.0-80.0	7	62.8 ± 17.6	23.3-86.8	32	
No. emergent hatchling per nest	66.3 ± 15.4	34.0-80.0	7	68.9 ± 23.3	24.0-114.0	32	0.94
Clutch frequency (no. clutches)	3.6 ± 1.0	1-5	31	4.3 ± 1.2	1-6	229	0.0005
Breeding frequency (seasons)	1.8 ± 1.2	1-5	16	3.3 ± 2.3	1-10	82	<0.005
Breeding life span (yr)	4.0 ± 3.9	1-12	16	4.8 ± 3.5	1-15	82	0.17
Cumulative reproductive output (hatchlings)†	4.33 ± 16			1029 ± 27			
Remigration interval prior to sampling eggs (yr)	4.9 ± 1.5	3-7	7	1.6 ± 0.6	1-3	125	<0.0001
Mean remigration interval within an Individual (yr)	3.8 ± 0.9	2.8-5.0	8	1.8 ± 0.5	1.0-3.0	69	<0.0001

Notes: Division of turtles into the two groups is based on δ^{13}C and δ^{15}N in egg yolks; *n* indicates sample size. *P* values were calculated using Mann-Whitney *U* tests.

† Cumulative reproductive output = clutch size X emergence success X clutch frequency X breeding frequency.

Loggerhead hatchlings pip and escape from their eggs over a 1 to 3-day interval and move upward and out of the nest over a 2- to 4-day interval (Christens 1990). The time from pipping to emergence ranges from 4 to 7 days with an average of 4.1 days (Godfrey and Mrosovsky 1997). Hatchlings emerge from their nests en masse almost exclusively at night, and presumably using decreasing sand temperature as a cue (Hendrickson 1958; Mrosovsky 1968; Witherington et al. 1990). Moran et al. (1999) concluded that a lowering of sand temperatures below a critical threshold, which most typically occurs after nightfall, is the most probable trigger for hatchling emergence from a nest. After an initial emergence, there may be secondary emergences on subsequent nights (Carr and Ogren 1960; Witherington 1986; Ernest and Martin 1993; Houghton and Hays 2001).

Hatchlings use a progression of orientation cues to guide their movement from the nest to the marine environments where they spend their early years (Lohmann and Lohmann 2003). Hatchlings first use light cues to find the ocean. On naturally lighted beaches without artificial lighting, ambient light from the open sky creates a relatively bright horizon compared to the dark silhouette of the dune and vegetation landward of the nest. This contrast guides the hatchlings to the ocean (Daniel and Smith 1947; Limpus 1971; Salmon et al. 1992; Witherington 1997; Witherington and Martin 1996; Stewart and Wyneken 2004).

Immediately after hatchlings emerge from the nest, they begin a period of frenzied activity. During this active period, hatchlings move from their nest to the surf, swim, and are swept through the surf zone (Carr and Ogren 1960; Carr 1962, 1982; Wyneken and Salmon 1992; Witherington 1995). Orientation cues used by hatchlings as they crawl, swim through the surf, and migrate offshore are discussed in detail by Lohmann and Lohmann (2003).

Neonate loggerheads that have migrated away from land differ from swim frenzy stage hatchlings in that they are largely inactive and only exhibit infrequent low-energy swimming, and they have begun to feed, no longer relying on their retained yolk (Witherington 2002). As post-hatchlings, loggerheads are pelagic and are best known from neritic waters along the continental shelf. This neritic post-hatchling stage is weeks or months long (Witherington 2002) and may be a transition to the oceanic stage that loggerheads enter as they grow and are carried within ocean currents (Bolten 2003).

The oceanic juvenile stage begins when loggerheads first enter the oceanic zone (Bolten 2003). Juvenile loggerheads originating from nesting beaches in the Northwest Atlantic, West Indian, and West Pacific Oceans appear to use oceanic developmental habitats and move with the predominant ocean gyres for several years before returning to their neritic foraging and nesting habitats (Hughes 1974a; Bowen et al. 1995; Musick and Limpus 1997; Pitman 1990; Zug et al. 1995; Bolten 2003). However, the actual duration of the oceanic juvenile stage varies with loggerheads leaving the oceanic zone over a wide size range (Bjorndal et al. 2000). In the Atlantic, Bjorndal and colleagues (Bjorndal et al. 2000, 2003a) estimated the duration of the oceanic juvenile stage to be between 7 and 11.5 years, with juveniles recruiting to neritic habitats in the western Atlantic over a size range of 46-64 cm CCL (Bolten et al. 1993, Turtle Expert Working Group 2009). However Snover (2002) suggests a much longer oceanic juvenile stage duration with a range of 9-24 years and a mean 14.8 years over similar size classes. However, in Japan and Australia, juvenile loggerheads do not disperse to neritic habitats until larger than around 60 cm SCL (Y. Matsuzawa and Sea Turtle Association of Japan, unpublished data) and 70 cm CCL or larger (Limpus et al. 1994), respectively, while in the Mediterranean they do so at around 25 cm CCL (Casale et al. 2008a).

The neritic juvenile stage begins when loggerheads exit the oceanic zone and enter the neritic zone (Bolten 2003). After migrating to the neritic zone, juvenile loggerheads continue maturing until they reach adulthood. Some juveniles may periodically move between the neritic and oceanic zones (Witzell 2002; Bolten 2003; Morreale and Standora 2005; McClellan and Read 2007; Mansfield 2006; Eckert et al. 2008). The neritic zone also provides important foraging habitat, inter-nesting habitat, and migratory habitat for adult loggerheads. Some adults may also periodically move between the neritic and oceanic zones (Harrison and Bjorndal 2006). See Limpus and Limpus (2003a) for reviews of this life stage in the Pacific.

North Pacific loggerheads are slow growing and have delayed sexual maturity. Somatic growth rates range from 0 to 5 cm/year using SCL[3] measurements (Hatase et al. 2004; Turner Tomaszewicz et al. 2017), although small juveniles have been shown to grow up to 10 cm/yr (Turner Tomaszewicz et al. 2015). Turner Tomaszewicz et al. (2015) estimated loggerhead sea turtles in the Gulf of Ulloa range from three to 24 years of age based on skeletochronological analysis, suggesting that individuals moving into the Eastern Pacific during their early development may spend 20 plus years in this region before reaching maturity and returning to the Western Pacific for their adult life phase. Adult female size ranges from 74 cm to 96 cm (Hatase et al. 2002b, 2004) and there is evidence that habitat preference influences size at maturity, with oceanic-foraging adults being slightly smaller than neritic-foraging adults (Hatase et al. 2002). There are also differences among these two different foraging groups in clutch size, clutch frequency, breeding frequency, and remigration intervals, with the neritic foragers having a 2.4-fold greater reproductive output (Hatase et al. 2013). Other stages also have potential different life history strategies.

In the most recent Status Review, Conant et al. (2009) suggested an age-at-maturity of approximately 30 yrs., although this likely varies among individuals due to differences in rate of nutrient acquisition during juvenile life stages and perhaps differences in foraging areas, neritic vs oceanic. Based on skeletochronology, Turner Tomaszewicz et al. (2015) estimated the age-at-maturity for Baja California Peninsula loggerheads to be 25 years old, with prior studies having yielded similar findings based on body size at first nesting, remigration intervals, and climate forcing models (Kamezaki et al. 1995; Hatase et al. 2004; Van Houtan and Halley 2011). Data on reproductive longevity of loggerheads are limited, but are becoming available with increasing numbers of intensively monitored, long-term projects on protected beaches.

4.2.2.3 Status

In coastal foraging habitats off of Mexico, abundance estimates by Seminoff et al. (2014) indicated that upwards of 43,000 loggerhead sea turtles may occur annually in the Gulf of Ulloa, Baja California Peninsula which likely represents a significant portion of the juvenile component of the North Pacific loggerhead sea turtle DPS. Although it is unknown what proportion of the overall DPS occurs in Mexico. Loggerhead turtles in this region range from 3 to 24 years old based on skeletochronological analysis (Turner-Tomaszewicz et al. 2015). After spending years, potentially two decades, foraging in the central and Eastern Pacific (Abecassis et al. 2013;

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[3] In sea turtles, SCLs are measured from the anterior point at the midline (i.e. nuchal scute) to the posterior tip of the supracauadals (Bolten 1999).

Turner-Tomaszewicz et al. 2015); loggerheads return to their natal Japanese nesting beaches for reproduction and remain in the Western Pacific for the remainder of their life cycle (Kamezaki et al. 1997; Conant et al. 2009; Kobayashi et al. 2011).

Nesting beach monitoring in Japan began in the 1950s on some beaches, and grew to encompass all known nesting beaches starting in 1990 (Kamezaki et al. 2003). Along the Japanese coast, nine major nesting beaches (greater than 100 nests per season) and six "submajor" beaches (10–100 nests per season) exist, including Yakushima Island where approximately 40% of nesting occurs (Kamezaki et al. 2003). Census data from 12 of these 15 beaches provide composite information on longer-term trends in the Japanese nesting assemblage. As a result, Kamezaki et al. (2003) concluded a substantial decline (50–90%) in the size of the annual loggerhead sea turtle nesting population in Japan since the 1950s. As discussed in the 2011 final ESA listing determination, current nesting in Japan represents a fraction of historical nesting levels (Conant et al. 2009; 76 FR 58868, September 22, 2011). Nesting declined steeply from an initial peak of approximately 6,638 nests in 1990–1991, to a low of 2,064 nests in 1997. During the past decade, nesting has been variable, increasing and decreasing over time as is typical of sea turtle nesting trends. Nesting increased gradually to 5,167 nests in 2005 (Conant et al. 2009), peaked to 11,082 nests in 2008, declined and then has risen steadily to a record high of 15,396 nests in 2013 (Sea Turtle Association of Japan (STAJ) 2008, 2009, 2010, 2012; Matsuzawa, Y. pers. comm. 2014). Nesting activity declined in 2014 to less than 10,000 nests, and again in 2015 with less than 5,000 nests laid, but has stabilized with a slight increase in 2016 (I Kelly pers. comm. in consultation with Y Matsuzawa 2018).

For the 23-year period 1990-2013, the total number of nests per year for the North Pacific DPS ranged between 2,064 – 15,396 nests. The 2015 IUCN Redlist Assessment estimated the total number of nests in the subpopulation, including beaches with less than 10 years of monitoring (62 nesting beaches in total), between years 2009-2013 was about 9,050 nests yr-1 (Limpus and Casale 2015). Assuming a clutch frequency of four nests per female per year (Van Houtan 2011), the number of nesting females per year between 1990 and 2013 may have ranged from 516 – 3,849. The current median female abundance estimate for 2013-2015 is 3,652 (95% CI, 2976 to 4468) (Jones et al. 2018).

NMFS estimated the long-term population trend using a stochastic density-independent exponential growth model using a Bayesian state-space modeling framework to assist us in understanding the current status of the North Pacific loggerhead DPS (Jones et al. 2018). See the Population Viability section previously discussed in the leatherback sea turtles for discussion on model selection and usage. This section will discuss the data and results specific to the North Pacific loggerhead sea turtle.

The data is based on annual female counts (raw annual nest counts divided by clutch frequency of 3 nests per female; (Conant et al. 2009)). We use annual counts instead of a running sum to minimize assumptions about remigration interval. The model used nesting data from three index beaches in Yakushima, Japan – Inakahama, Maehama, and Yotsusehama; which represent approximately 52% of loggerhead nesting in Japan. Yakushima Island is within the Yakushima Management Unit that is described earlier in this section (Matsuzawa et al. 2016). The data are shown in Figure 13 as the natural log of annual nesting females; this is how they were input to the model.

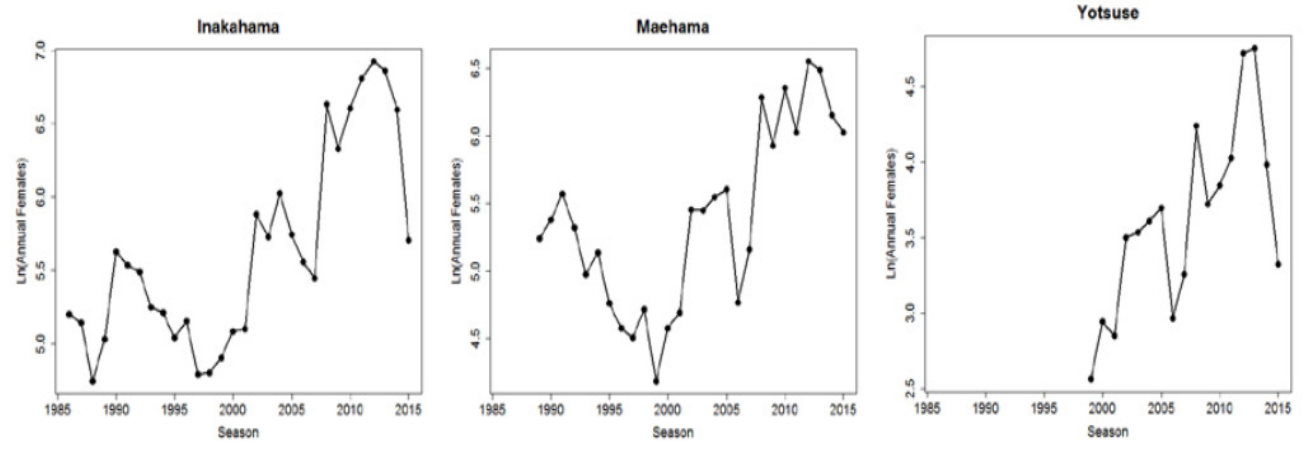

Figure 13. Natural log of annual nesting female loggerhead sea turtles from three nesting beaches in Yakushima, Japan (Source: Jones et al. 2018).

Jones et al. (2018) fit the population growth rate, ‘r’, using Bayesian state-space model approach based on density-independent stochastic exponential population growth as in Boyd et al. (2017). Other model approaches were considered; however, the Bayesian state-space approach incorporates uncertainty better and partitions that uncertainty into both process and observation error components. The predicted model median (blue line) with 95% credible intervals (gray shading) are overlaid on data points (black dots) for each time series, scaled appropriately by the model-estimated scaling parameters (Figure 14).

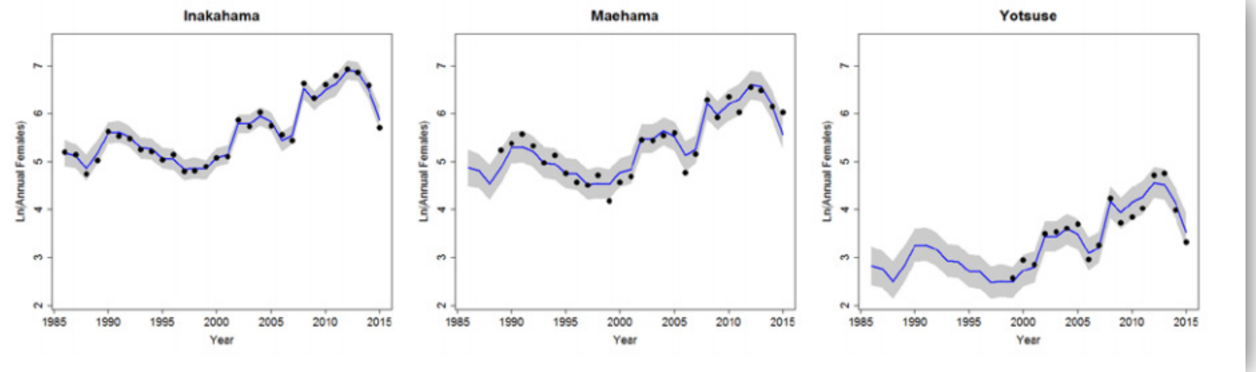

Figure 14. Predicted trends in nesting female loggerhead sea turtles from three nesting beaches in Yakushima, Japan (median (blue line) of the natural log of annual nesting females with 95% credible intervals (gray shading)) (Source: Jones et al. 2018).

We used model-estimated annual female distributions for 2013- 2015 to estimate an index of current total reproductive female abundance; this is computed as a 3-year run sum (3 year remigration interval; (Conant et al. 2009)). The estimated distributions for 2013-2015 Annual

Females and the summed Total Reproductive Females estimate are summarized in the Table 17. Model-estimated female loggerhead sea turtle numbers and the associated 95% credible intervals (Source: Jones et al. 2018)..

Table 17. Model-estimated female loggerhead sea turtle numbers and the associated 95% credible intervals (Source: Jones et al. 2018).

Observed data year	Median	Lower 95%	Upper 95%
Final Data Yr -2	1777	1465	2165
Final Data Yr -1	1203	984	1448
Final Data Yr -0	652	527	855
Sums-CurAbundEst	3630	676	4468

Next, 10,000 model projection runs projecting 100 years into the future from the final data year (2015) and are shown in Figure 15. Model projections start with a value drawn from the posterior distribution of Annual Females for 2015; use dynamic ‘r’ (draw new ‘r’ each future year within a simulation run) and dynamic process error variance, ‘Q,’ to add natural variability. Furthermore, model projections are of annual females in natural log space.

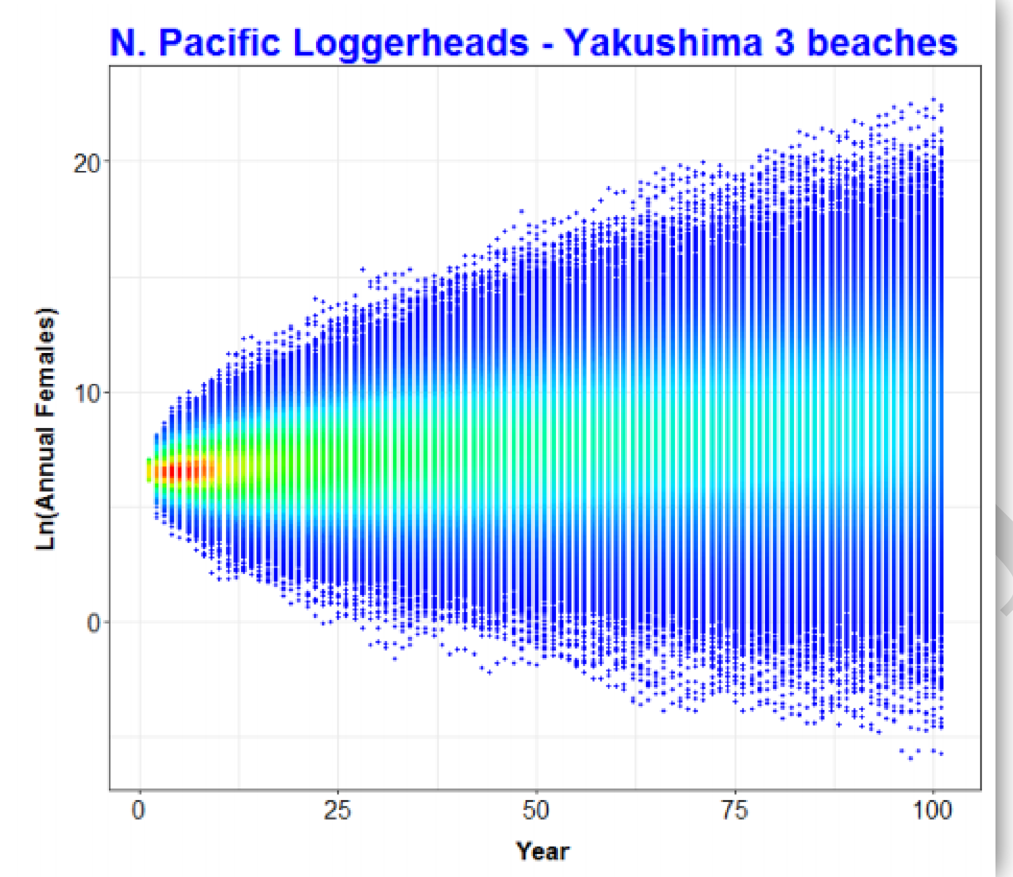

Figure 15. Model projections (10,000) of annual females in natural log space for 100 years into the future from 2015 (data year) (from Jones et al. 2018).

Next, NMFS estimated the mean and median time until the population declines to 50%, 25%, and 12.5% of its current abundance estimates. Abundance thresholds were calculated for each future projection (n=10,000 simulation runs) of total reproductive females (3-year running sum of projected annual females) is compared to the specified fractional decline from the starting estimate of current total reproductive females. Values are represented in Table 18. Whereas Table 19 lists the probability of the population reaching abundance thresholds at 5, 10, 25, 50 and 100 years from the data year (2015).

Table 18. Probability of the population reaching abundance thresholds within the 100 year projection period, and time in years (mean, median, & 95% credible interval) to reach the threshold for all runs that fall below the threshold (Source: Jones et al. 2018).

Threshold	Probability of staying above threshold	Probability of falling below threshold	Years to reach threshold (Mean)	Years (Median)	Years (lower 95% CI)	Years (upper 95% CI)
50% abundance	75%	25%	49	50	2	99
25% abundance	80%	20%	56	59	3	99
12.5% abundance	85%	15%	62	68	7	99

Table 19. Probability (with 95% credible intervals) of the population reaching abundance thresholds at 5, 10, 25, 50, and 100 years from final data year (2015) (Source: Jones et al. 2018).

Abundance Threshold	Yr 5 (%)	Yr 10 (%)	Yr 25 (%)	Yr 50 (%)	Yr 100 (%)
50%	13	14	16	18	25
Lower 95% CI	12	13	15	17	25
Upper 95% CI	13	15	16	18	26
25%	3	6	9	12	20
Lower 95% CI	3	6	9	12	19
Upper 95% CI	4	7	10	13	20
12.5%	0	2	5	8	15
Lower 95% CI	0	2	5	8	14
Upper 95% CI	0	2	6	9	16

Finally, the Bayesian model estimates provide an index of population growth using the best available index of population abundance derived from nest count data. Estimating true population growth rates would require additional data or assumptions about the population age distribution. Rates reflect the long-term trend estimated using the annual female count data described above. For r, if values are positive then growth is positive; if values are negative then growth is negative. For λ lambda, if values are greater than 1 then growth is positive; if values are less than 1 then growth is negative.

Log growth rate of population (r):

mean: 0.024
median: 0.024
variance: 0.005
95% CI: -0.108 to 0.156

Finite rate of increase (ƛ):

mean: 1.024
median: 1.024
95% CI: 0.897 to 1.168

The mean log growth rate is a measure of the population's "average" growth rate assuming that some trajectories will increase, some will remain stable, and others will decrease. If a population's mean log growth rate is greater than zero, then most population trajectories will increase; if less than zero, then most population trajectories will decline (Morris and Doak 2002). A population's finite rate of increase (λ) captures a population's growth rate or the amount by which a population size multiplies from year to year. In the face of stable environmental conditions, this growth rate would be constant and a population would increase geometrically (λ >1), decrease geometrically (λ <1), or remain the same (λ =1). The λ mean is 1.024 (95% CI, 0.897 to 1.168). These results suggest that most trajectories of this subpopulation increase slightly.

4.2.2.4 Threats to the Species

Natural Threats

Natural threats to sea turtles include progressive shifts in climate dynamics (i.e. climate change) causing increased frequency of cyclones, hurricanes, typhoons, and tsunamis; shifts in ocean currents, beach washover and inundation, nesting beach erosion and modification, foraging habitat modification, shifts or alterations in food web dynamics, and harmful algae blooms.

We expect natural predation occurs in the all life cycle stages based on literature review of the global population of loggerheads. However, information specific to the North Pacific loggerhead sea turtle DPS is not available for these life history stages and is not considered a significant threat to the species at this time.

Nest and egg predation has been extensively documented for the various sea turtle species, including the global population of loggerheads, by birds, crocodiles, insects, invertebrates, large lizards, and mammals (Seaman et al. 1962; Fowler 1979; Brown and McDonald 1995; Lutcavage et al. 1997; Moulis 1997; Yerli et al. 1997; Allen et al. 2001; Limpus and Limpus 2003; Marcus et al. 2003; Blamires 2004; Dunlan et al. 2004; Wetterer et al. 2014; Garmestani 2005; Caut et al. 2006 (and references therein); Engeman et al. 2006, 2010; Ficetola 2008; Leighton et al. 2008; Tomillo et al. 2010; Whiting and Whiting 2011; Parris 2013; Wyeneken 2013; Burger and Gochfeld 2014; Sim 2014; Brost et al. 2015; Marco et al. 2015; Engeman et al. 2016; Lei and Booth 2017; O'Conner et al. 2017). Factors that affect nest and egg predation include intraguild predation, nest location, emergence intervals and timing, location, depth, temperature, moisture levels, and soil composition (Gyuris 1993; Brown and McDonald 1995; Chaloupka and Limpus

2001; Dutton et al. 2005; Caut et al. 2006; Whiting 2007; Barton and Roth 2008; Medderea et al. 2010; Leighton et al. 2010; Wyneken et al. 2013; Sim 2014; Brost et al. 2015; Marco et al. 2017). Additionally, egg mortality can be influenced by both fungal and bacterial infections in reptiles (Moreira 2005; Soslau 2011) and specifically in loggerheads (Wyneken et al. 1988, 2013; Peters et al. 1994; Phillot et al. 2001; Phillot and Parmenter 2001; Guclu et al. 2010; Bailey et al. 2018), although egg structure was not shown to influence the direct invasion potential by fungi (Phillott and Parmenter 2006). Moreover, extreme sand temperatures at nesting beaches create skewed female sex ratios and also can cause deformities and mortalities (Conant et al. 2009). It should be noted that, although nest predation in Japan is known to occur by raccoon dogs (*Nyctereutes procyonoides*) and weasels (*Mustela itatsi*) (Kamezaki et al. 2003), quantitative data are not sufficient to assess the degree of impact of nest predation on the persistence of this DPS at this time.

Sea turtle hatchlings face a gauntlet of natural predators, both during the crawl to water, and once they enter the ocean. Predation events have been documented for various species of sea turtles by invertebrates, birds, insects, mammals, and fish (Gyuris 1994; Stancyk 1995; Pilcher et al. 2000; Allen et al. 2001; Madden et al. 2008; Tomillo et al. 2010; Reblo et al. 2012; Wyneken et al. 2013; Burger and Gochfeld 2013; da Silva et al. 2016; Santos et al. 2016). Predation is documented in the global population of loggerheads (Stancyk 1980; Witherington and Salmon 1992; Parris 2002; Stewart and Wyneken 2004; Whelan and Wyneken 2007; Sim 2014; Brost et al. 2015; Silva et al. 2017); however, information specific to the North Pacific loggerhead sea turtle DPS is not available.

Additionally, while reviewing the scientific literature on predation of juvenile and adult sea turtles it was shown that terrestrial mammals, crocodiles, marine mammals like killer whales, and various shark species (tiger, great white, bull, oceanic whitetip, great hammerhead, blue, and dusky), depredate sea turtles; including but not limited to the global population of loggerhead sea turtles (Kauffman 1950; Compagno 1984; Witzell 1987; Stevens and Lyle 1989; Marquez 1990; Cliff and Dutley 1991; Simpfendorfer 1992; Hirth et al. 1993; Keinath and Musick 1993; Autar 1994; Limpus et al. 1994a, 1994b; Long 1996; Lowe et al. 1996; Gelsleichter et al. 1999; Ortiz et al. 1997; Fergusson et al. 2000; Troeng 2000; Brown 2001; Simpfendorfer et al. 2001; Compagno 2002; Heithaus 2001; Heithaus et al. 2002, 2005, 2008; Sutherland and Sutherland 2003; Pitman and Dutton 2004; Fertl and Fulling 2007; Blumenthal et al. 2009; Lopez 2010; Whiting and Whiting 2011; Fitzpatrick et al. 2012; Verissimo et al. 2012; Hammerschlag et al. 2015, 2016; Acuna-Marrero et al. 2017; Butera and Restrepo 2018 (In-review)). However, studies specific to the North Pacific loggerhead sea turtle DPS are lacking. It should be noted that studies have been conducted in range of the species, but fail to identify shark stomach contents to sea turtle species. Conversely, some species like the jaguar, predate juvenile and adult loggerhead sea turtles, however no geographical overlap occurs with the North Pacific loggerhead sea turtle DPS, and would not be an applicable threat. Whereas, interactions between species like crocodiles and the North Pacific loggerhead sea turtle DPS could potentially occur in the Philippines based on geographical overlap of the two species, but has not been documented to date.

Anthropogenic Threats

Sources of mortality specific to the North Pacific loggerhead DPS include: human encroachment and egg harvest/predation on nesting beaches, nesting beach alteration (armoring and habitat degradation, dam construction, sand compaction, disorientation of hatchlings by artificial lighting, etc.), marine debris, ingestion of and entanglement in marine debris, vessel strikes, incidental capture in coastal and pelagic fisheries (including longline, drift gillnet, set-net, bottom trawling, dredge, and trap net) throughout the species' range (Koch et al. 2006; Peckham et al. 2007, 2008; Dutton and Squires 2008; Conant et al. 2009; Ishihara 2009; Ishihara et al. 2011; Van Houtan and Halley 2011; 76 FR 58867). While the Services cannot predict the exact impacts of climate change, sea level rise may present a more immediate challenge for this DPS because of the proportion of beaches with shoreline armoring that prevents or interferes with the ability of nesting females to access suitable nesting habitat.

Loggerhead nesting habitat also has been lost at important rookeries in Miyazaki due in part to port construction that involved development of a groin of 1 kilometer from the coast into the sea, a yacht harbor with breakwaters and artificial beach, and an airport, causing erosion of beaches on both sides of the construction zone. This once excellent nesting habitat for loggerheads is now seriously threatened by erosion (Takeshita 2006).

However, interactions and mortality with coastal and artisanal fisheries in Mexico and the Asian region likely represent the most serious threats to North Pacific loggerhead sea turtle DPS (Peckham et al. 2007, 2008; Conant et al. 2009; Ishihara 2009).

Bycatch and fisheries-related standings' numbering in the thousands annually have been reported from gillnet and longline fisheries operating in loggerhead sea turtle 'hotspots' off of Baja Mexico, where intense coastal fishing pressure overlaps with high densities of loggerhead sea turtles foraging in nearshore habitats, producing among the highest bycatch rates reported worldwide (Peckham et al. 2007, 2008; Conant et al. 2009; Wingfield et al. 2011). Results of a 2014 study suggest that up to 11% of the region's loggerhead sea turtle population may perish each year (Seminoff et al. 2014).

Between 2003 and 2010, annual stranding surveys to assess mortality have documented 3,096 dead loggerhead sea turtles (with a mean of 420 ± 274/year) along a 45 km stretch of beach of Playa San Lazaro in Baja California SUR, Mexico (Peckham 2010). For comparison purposes, along this same beach during same time period, 144 olive ridley and 279 green turtles were documented as stranded (Peckham et al. 2007, 2008). Studies using drifter experiments in Baja, concluded that stranding probability estimates may vary between 5-20%, and it is likely that 150 loggerhead sea turtles may have died during a 15 day period in 2010 (Koch et al. 2013). In July 2012, a record 483 dead loggerhead sea turtles stranded along 43 kilometers of the shoreline of Playa San Lazaro, Baja California Sur. This was a 600% increase over the annual average of 78 loggerhead sea turtles in July since 2003. This prompted the U.S. to identify Mexico under the Magnuson Stevens Reauthorization Act for fishing interactions involving the bycatch of loggerhead sea turtles (Benaka et al. 2012; NMFS 2013). Mexico was given two years (by January 2015) to demonstrate that they have a comparable regulatory program to the U.S. to reduce sea turtle bycatch or they would be negatively certified under the Magnuson Stevens Reauthorization Act. A negative certification could result in prohibitions on the importation of certain fishery products into the U.S. or the denial of port privileges for vessels of that nation. In August 2015, Mexico received a negative certification from NMFS and were advised to revise and resubmit their proposed regulations. In September 2016, NMFS issued a positive determination based on changes Mexico made to their regulatory program and additional commitments regarding the duration of their regulations and co-operative research with the U.S. (NMFS 2016, 2017).

Overutilization for commercial purposes in both Japan and Mexico was likely a factor that contributed to the historical declines of this DPS. Current illegal harvests of loggerheads in Baja, California for human consumption continues as a significant threat to this DPS.

4.2.2.5 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. NMFS has formalized conservation actions to protect foraging loggerhead sea turtles in the North Pacific Ocean which were implemented to reduce loggerhead sea turtle bycatch in U.S. fisheries. Observer programs have been implemented in federally-managed fisheries to collect bycatch data, and several strategies have been pursued to reduce both bycatch rates and post-hooking mortality. 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 (NMFS and FWS 2007a). For example, switching to large circle hooks and mackerel bait in 2004 reduced the interaction rate by approximately 90% in the HI SSLL fishery (Gilman et al. 2007a; WPFMC 2009). In 2003, NMFS implemented a time/area closure in southern California during forecasted or existing El Niño-like conditions to reduce the take of loggerhead sea turtles in the California/Oregon drift gillnet fishery (68 FR 69962; December 16, 2003). On July 25, 2014, El Niño conditions off the coast of California were determined to have been met for the first time which closed the Pacific Loggerhead sea turtle Conservation Area through August 31, 2014 to swordfish drift gillnet fishing (79 FR 43268; July 25, 2014). Efforts to elevate the awareness level of fishermen regarding protected species interactions and the potential impacts to their fisheries (i.e., closures if allowable take levels are exceeded in the shallow-set component of the fishery), and efforts to educate boat owners and operators via annual (and mandatory) protected species workshops cannot be discounted. NMFS has also developed a mapping product known as TurtleWatch that provides a near real time product that recommends areas where the deployment of pelagic longline shallow-sets should be avoided to help reduce interactions between Hawaii pelagic longline fishing vessels and loggerhead sea turtles (Howell et al. 2008).

Since loggerhead sea turtle interactions and mortalities with coastal fisheries in Mexico and Japan are of concern and are considered a major threat to North Pacific loggerhead sea turtle recovery, NMFS and U.S. non-governmental organizations have worked with international entities to: (1) assess bycatch mortality through systematic stranding surveys in Baja California Sur, Mexico; (2) reduce interactions and mortalities in bottom-set gillnet fisheries in Mexico; (3) conduct gear mitigation trials to reduce bycatch in Japanese pound nets; and (4) convey information to fishers and other stakeholders through participatory activities, events and outreach. In 2003, Grupo Tortuguero's ProCaguama (Operation Loggerhead) was initiated to partner directly with fishermen to assess and mitigate their bycatch while maintaining fisheries sustainability in Baja California, Mexico. ProCaguama's fisher-scientist team discovered the highest turtle bycatch rates documented worldwide and has made considerable progress in mitigating anthropogenic mortality in Mexican waters (Peckham et al. 2007, 2008). As a result of the 2006 and 2007 tri-national fishermen's exchanges run by ProCaguama, STAJ, and the Western Pacific Fisheries Management Council, a prominent Baja California Sur fleet retired its bottom-set longlines in 2007 (Peckham et al. 2008; Peckham and Maldonado-Diaz, 2012). Prior to this closure, the longline fleet interacted with an estimated 1,160-2,174 loggerhead sea turtles annually, with nearly all (89%) of the takes resulting in mortalities (Peckham et. al. 2008). Because this fleet no longer interacts with loggerhead sea turtles, conservation efforts have resulted in the continued protection of approximately 1,160-2,174 juvenile loggerhead sea turtles annually (76 FR 58868; September 22, 2011). Additionally, stranding data collected since 2003 at Playa San Lazaro indicates a 60% reduction in standings' during 2010 compared to previous 2003-2009 averages (Peckham 2010).

Led by the Mexican Wildlife Service, a federal loggerhead sea turtle bycatch reduction task force, comprised of federal and state agencies and non-governmental organizations, was organized in 2008 to ensure loggerhead sea turtles receive the protection they are afforded by Mexican law. In 2009, while testing a variety of potential solutions, ProCaguama's fisher-scientist team demonstrated the commercial viability of substituting bycatch-free hook fishing for gillnet fishing. ProCaguama, in coordination with the task force, is working to develop a market-based bycatch solution consisting of hook substitution, training to augment ex-vessel fish value, development of fisheries infrastructure, linkage of local fleets with regional markets, and concurrent strengthening of local fisheries management (Conant et al. 2009). As of 2012, a number of members of the gillnet fleet had retired their gear (a total of 140 gillnets), 18 crews have converted to hook and line fishing (a more sustainable practice in the 'hotspot' area that results in zero bycatch), and local NGO efforts were underway to implement the market-based solutions mentioned above to encourage consumption of sustainably caught sea food (Peckham 2014, Conant et al. 2009). Market-based efforts are underway to promote and support sustainable fisheries that use fishing gear, such as hand line, which have low (or no) sea turtle bycatch in Baja (Peckham 2014).

In Japan, due to concerns of high sub-adult and adult loggerhead sea turtle mortality in mid-water pound nets, researchers with the STAJ, ProCaguama, local fisherman, and NMFS are working together to investigate and test pound net mitigation options to reduce the impact and mortality of sea turtle bycatch. This effort has included public education and outreach activities with media events to raise public awareness of the bycatch problems. The first phase of the project was completed in February 2012, and three promising gear mitigation options for pound net fisheries were discovered (Matsuzawa et al. 2012). Continued collaborative efforts continue to progress development of mitigation measures that include refinement of solutions and testing in an operating net to assess target and non-target catch rates to secure industry uptake and buy-in (Ishihara et al. 2012).

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

Since 2003, WPFMC has contracted with STAJ to protect loggerhead sea turtle nests and increase hatchling survivorship at several nesting beaches in southern Japan, including at the two primary beaches on Yakushima Island. Beach management activities include conducting nightly patrols during the summer nesting season to relocate nests from erosion prone areas, protecting nests from predators and people with mesh and fences, and cooling nests with water and shading to prevent overheating during incubation. STAJ has developed techniques for nest relocation that now result in an average of 60% hatchling success rates (compared to nearly zero survival of the same nests laid in erosion prone areas). Conservation efforts funded by the WPFMC continued through 2012, with approximately 270,000 hatchlings conserved from relocated nests over the nine-year project period that otherwise may have been lost (Ishizaki 2015).

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 Food and Agriculture Organization's Technical Consultation on Sea Turtle-Fishery Interactions, the Inter-American Convention for the Protection and Conservation of Sea Turtles, the Convention on International Trade in Endangered Species, and others. In 2008 the Western and Central Pacific Fisheries Commission (WCPFC) adopted a Conservation and Management Measure (Conservation and Management Measure 2008-03) to mitigate the impacts on turtles from longline swordfish fisheries in the western central Pacific Ocean. The measure includes the adoption of Food and Agriculture Organization of the United Nations guidelines to reduce sea turtle mortality through safe handling practices and to reduce bycatch by implementing one of three methods by January 2010. The three methods to choose from are: 1) use only large circle hooks, or 2) use whole finfish bait, or 3) use any other mitigation plan or activity that has been approved by the Commission. As a result of these designations and agreements, many of the intentional impacts on sea turtles have been reduced: harvest of eggs and adults have been slowed at several nesting areas through nesting beach conservation efforts and an increasing number of community-based initiatives are in place to slow the take of turtles in foraging areas. Moreover, as shown by the above examples from Hawaii, Japan, and Baja Mexico, international efforts are growing to reduce sea turtle interactions and mortality in artisanal and industrial fishing practices (Gilman et al. 2007b; Peckham et al. 2007; NMFS and FWS 2007; Ishihara et al. 2012).

4.2.2.6 Summary of the Status of the NP Loggerhead Sea Turtle

In this section of this biological opinion, we explained that the North Pacific DPS of loggerhead sea turtle is endangered, and that the species' trend appears to be increasing slightly. 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 16). 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.

Based on the NMFS' Bayesian state-spaced model, the North Pacific loggerhead sea turtle population is increasing. Overall, the data shows the population is increasing at an estimated rate of about 2% per year and is at risk of falling to less than half of its current abundance in as few as two years (range 2-99 years, mean 49 years; see Table 18; Jones et al. 2018). The trend taken together with other information summarized in the preceding review of the species' status indicates that there is a low likelihood that the species will be extinct within the next 100 years (Jones et al. 2018).

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. 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 on the apparent dynamics of the of the North Pacific loggerhead sea turtle DPS, 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: North Pacific loggerhead sea turtles appear to be increasing and that trend will tend to reinforce itself.

Our graphic illustrates the population behavior of loggerhead sea turtles adults (females) in S1, which is based on available nest count data. Because nesting females drive the population dynamics of this species, they serve as a surrogate for all of the life stages in the population. Natural threats occur to the global population as a whole and therefore can potentially be relevant to the North Pacific DPS. Natural threats were discussed holistically as these threats pertain to all populations of loggerheads to varying degrees (76 FR 58867), and therefore, we illustrated some of these threats in our causal loop diagram (Figure 16). 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. We graphically reference the increases by oscillating patterns, as this is most likely a more realistic pattern to the population changes.

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 loggerhead sea turtle.

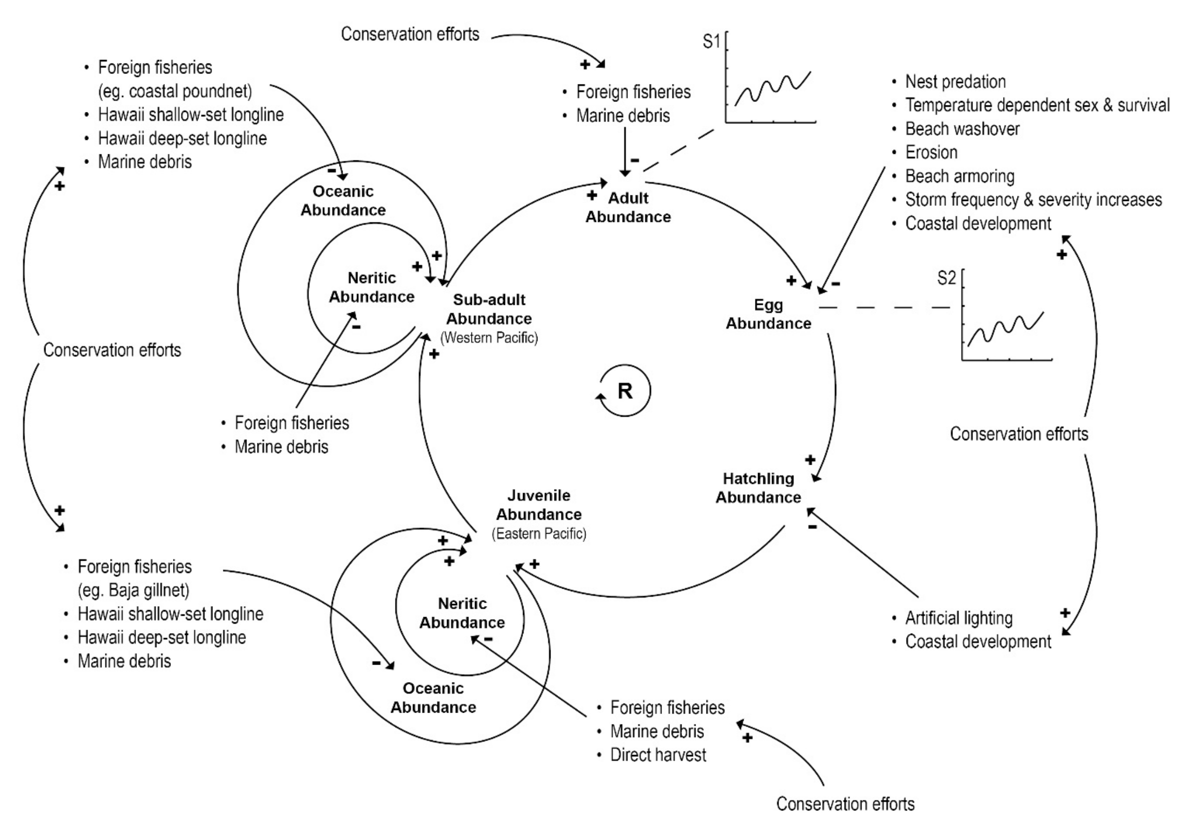

Figure 16. North Pacific loggerhead sea turtle DPS casual loop diagram depicting various stressor-response relationships for each life cycle stage

4.2.3 Green Sea Turtle

The green sea turtle was 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 Distinct Population Segments (DPSs) (Figure 17) that qualify as a “species” for listing. The Services removed the current range-wide listing and, in its place, listed eight DPSs as threatened and three as endangered. The green sea turtles most likely to occur in the range of the HI SSLL fishery are those DPSs that occur in the Pacific Ocean.

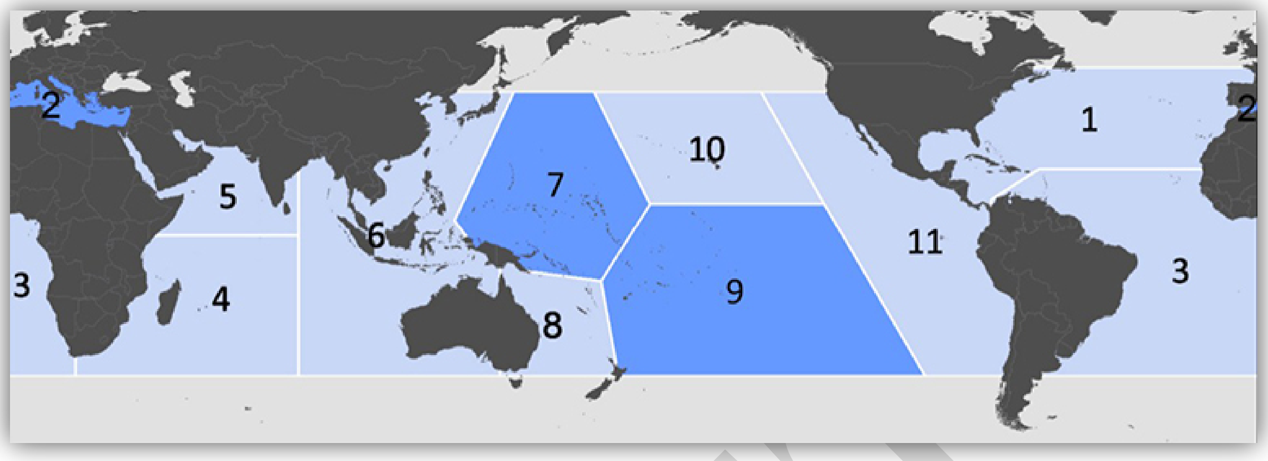

Figure 17. Overview of green sea turtle distinct population segments; (1) North Atlantic DPS (2) Mediterranean DPS (3) South Atlantic DPS (4) Southwest Indian DPS (5) North Indian DPS (6) East Indian - West Pacific DPS (7) Central West Pacific DPS (8) Southwest Pacific DPS (9) Central South Pacific DPS (10) Central North Pacific DPS (11) East Pacific DPS. Light blue indicates threatened populations whereas dark blue indicates endangered populations.

PIRO's observer program collected 19 samples from green sea turtles between 1996 and 2017. Genetic analysis of these samples was conducted by the NMFS Southwest Fisheries Science Center. They used two different approaches: a Bayesian mixed stock analysis (MSA) of pooled data, and a direct count of individual assignments based on haplotype that incorporated photo identification (Dutton pers. comm. August 31, 2016; see Table 20 for summary information from these assessments). Due to the very small sample sizes we cannot determine the percentages of turtles from each DPS with great certainty. Furthermore, due to the low number of available genetic samples and taking into account the migration and foraging behaviors of the various green sea turtle DPSs in consideration, we will use the results from the genetic analysis cautiously..

Table 20. Genetic composition of green turtles sampled from the Hawaii shallow-set longline fishery (Dutton pers. comm. August 31, 2016). The number of genetic samples column is from the direct approach and the MSA% is the mixed stock analysis with the 95% confidence interval.

DPS (ID #)	Number of genetic samples	MSA mean %	95% CI
East Pacific (11)	11	55	32-77
Central North Pacific (10)	8	41	20-64
East Indian-West Pacific (6)	0	1	0-10

Southwest Pacific (8)	0	1	0-8
Central West Pacific (7)	0	1	0-8
Central South Pacific (9)	0	1	0-5

Due to the very small sample sizes we cannot determine the percentages of turtles from each DPS with great certainty as evidenced from the large CI. Therefore we will use the upper 95% CI to analyze the potential impacts to each DPS.

Distribution

The green turtle has a circumglobal distribution; occurring throughout tropical, subtropical waters, and, to a lesser extent, temperate waters (Figure 18). Their movements within the marine environment are not fully understood, but it is believed that green turtles inhabit coastal waters of over 140 countries (Groombridge and Luxmoore 1989).

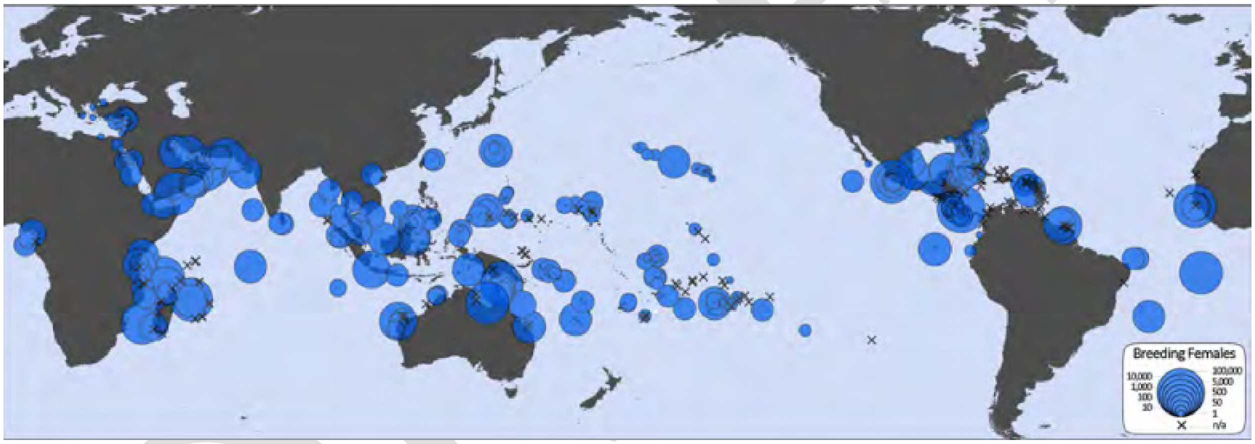

Figure 18. Nesting distribution of green turtles around the globe. Sites marked with an 'X' indicate known nesting sites, but for which no recent nesting abundance data are available.

Most green turtles spend the majority of their lives in coastal foraging grounds. These areas include fairly shallow waters both open coastline and protected bays and lagoons. While in these areas, green turtles rely on marine algae and seagrass as their primary diet constituents, although some populations also forage heavily on invertebrates. These marine habitats are often highly dynamic and in areas with annual fluctuations in seawater and air temperatures, which can cause the distribution and abundance of potential green turtle food items to vary substantially between seasons and years (Carballo et al. 2002). Many prey species that are abundant during winter and spring periods become patchy during warm summer periods. Some species may altogether vanish during extreme temperatures, such as those that occur during El Niño Southern Oscillation events (Carballo et al. 2002). Sub-adults and adults from Pacific Ocean populations have been taken as bycatch in several tuna longline fisheries in the Pacific (Beverly and Chapman 2007), including in the HI SSLL and DSLL fisheries.

Differential Distribution

Throughout its life cycle the green turtle relies on developmental habitats that include nesting beaches, and coastal and pelagic waters. Four basic life cycle stages are described: egg, hatchling, juvenile, and adult. These life history stages form the foundation of our life history model, and causal loop diagram (Figure 26), which we describe in more detail later in this section. Globally, green turtles comprise a mosaic of populations each with unique nesting sites and, in many cases, possessing disparate demographic features (e.g., mean body size, age at first reproduction; NMFS and USFWS 2007). However, despite these differences, green turtles from different populations often mix in common foraging grounds, thus creating unique challenges when attempting to delineate distinct population segments for management or listing purposes under the ESA (Jensen et al. 2013). Distinctive features of morphology, behavior, or life history that might indicate important local adaptations are documented for turtles from the Mediterranean, North Atlantic, South Atlantic, North Indian, East Pacific, and Central North Pacific. Turtles from the East Pacific are morphologically distinctive and exhibit a unique behavior somewhat akin to hibernation (described below). All nesting sites in the Mediterranean and some in the North Atlantic are outside the normal latitudinal range for the species. Different life history and distributional patterns, status and threats for the following green turtle DPSs that occur in the action area are described below: East Pacific, Central North Pacific, Central South Pacific, Central West Pacific, Southwest Pacific and East Indian/West Pacific (Figure 17).

Adult Foraging and Diving

Juvenile and adult green turtles are known to forage in coastal areas, estuaries, lagoons (Seminoff et al. 2002a) and near shore insular habitats (Amorocho and Reina 2007). Even though it was believed that green turtles only foraged in coastal environments, recent studies demonstrated they also forage in open waters as adults (Quiñones et al. 2010). Their diet is mainly sea grass and red and green algae. However, they also eat invertebrates and gelatinous prey (Seminoff et al. 2002b, 2006a). Adult green turtles foraging off shore of Peru have a diet dominated by jelly fish, mollusks and crustaceans (Quiñones et al. 2010).

Seminoff et al. studied the diet of green turtles at Bahía de los Angeles in the Gulf of California, México. From 1995–1999, samples were collected from live-captured turtles and stomach contents were analyzed from stranded carcasses encountered in the study area. Foods ingested did not vary with carapace length. Turtles consumed diverse marine algae, with the filamentous red alga *Gracilariopsis lemaneiformis* the most abundant; and other common genera included *Gracilaria, Codium, Ulva*, and *Chaetomorpha.* Turtles also augmented their diet with animal matter; 25 nonalga food items were identified, including sponges, tube worms, sea pens, and sea hares. Substratum and anthropogenic debris such as plastic bags and nylon cord were commonly recovered in diet samples.

The eastern Tropical Pacific offers areas of high productivity as a result of its particular oceanic features. It is characterized by the presence of coastal and open ocean upwellings, fronts, eddies and meanders (Lavín et al. 2006). The convergence of the trade winds of the intertropical convergence zone (ITCZ) produces a low wind area off Central America where the sea surface temperatures are higher than elsewhere in the area. However, the wind blowing from land to sea produces the Tehuantepec, Papagayo and Panama jets causing eddies that generate productive areas exhibiting a decrease in sea surface temperatures, an increase in chlorophyll levels, and interrupting the warm low productivity "warm pool" (Palacios and Bograd 2005; Lavín et al.

2006). Within this region, the Costa Rican dome is another upwelling region in the Eastern Pacific with high primary and secondary production (Lavín et al. 2006).

Chambault et al. (2015) analyzed the trajectories and diving behavior of 19 adult green turtles during their post-nesting migration from French Guiana and Suriname to their foraging grounds off the coast of Brazil and observed maximum dive depths of from 10 to 100 m. Seventy% of the dives were performed at shallow depths within 30 m of the surface. Data from this study also showed that the most favorable habitat conditions for the foraging activity of green turtles are clear and cool waters, which are associated with low bathymetry. Data confirm that green turtle foraging grounds are located near seagrass beds (Chambault et al. 2015). Seagrass ecosystems are fragile, relying on complex biochemical processes, since seagrass growth is regulated by temperature, light and nutrient availability.

Blanco (2010) used satellite telemetry and deployment of data loggers to document different dive patterns of East Pacific turtles off of Costa Rica including depths, durations and ascent and descent phases. During the internesting period, turtles exhibited a strong diel pattern in their submergence behavior. U-dives are the predominant dive type that sea turtles perform during the internesting period (Hays et al. 1999; Hoscheid et al. 1999; Houghton et al. 2002). U-dives are also commonly related to resting behavior, which appears to be the main activity during the internesting period of sea turtles. To identify U-dives, Blanco (2010) calculated the relation between total duration of an individual dive and the time at the bottom. If the time at the deepest depth was 90% or more of the dive, it was considered a U-dive (Hays et al. 1999). Blanco (2010) compared the proportion of U-dives performed during the day and night with the time spent at the surface and found that turtles spent more time at the surface at night and more time performing U-dives during the day. The data indicated that the internesting interval for the East Pacific turtle population nesting in Costa Rica was 12 days. This was shorter than the 14 days reported at Ascension Island by Carr et al. (1974) and at Tortuguero, Costa Rica (Bjorndal and Carr 1989). Data obtained from the satellite transmitters indicated that, during internesting, turtles dive mostly between depths of 2 and 10 m (90% of the dives) performing mainly short dives. The depth of the majority of the dives corresponded to the depth of the area where the turtles stayed during the internesting interval indicating that diving behavior of the turtles was mainly influenced by bathymetry of the internesting area and turtles mainly dove to the bottom.

U-dives are related to resting behavior because turtles remain at a fixed depth for a period of time; this depth often is the bottom (Minamikawa et al. 1997, Hays et al. 2000). U-dives have widely been considered to represent resting activity on the sea floor, although on occasion some other activities such as foraging or movement along the bottom may occur (Hochscheid, et al. 1999; Seminoff et al. 2006). The duration of U-dives varied with individual turtles from 8 to 23 minutes. East Pacific green turtles in Blanco's (2010) study area spent a large amount of time at the surface and the majority of the surface time occurred at night. A basking sea turtle can increase its body temperature almost 4ºC by exposing a substantial portion of the carapace to the sun (Spotila and Standora 1985). Blanco's (2010) study showed that turtles were at the surface at night indicating that they were not basking, but rather resting. During internesting, turtles need to save energy to produce eggs, crawl to the beach and lay the eggs several times in the season. Green turtles in Blanco's (2010) study remained close to the nesting beach and rested most of the time during the interesting period. These behaviors likely reduced energy expenditure during the nesting season which could be a significant cost for reproduction (Wallace et al. 2005). During the reproductive season sea turtles mainly rely on stored lipids to support metabolism and reproduction (Hamann et al. 2002) especially in areas where food is not plentiful.

Blanco (2010) reported that, during migration, turtles showed a bimodal distribution in duration of dives with major durations between 2-5 min and 40-60 min. The shorter intervals were associated with shallow dives and the longer intervals with deeper dives. Depth most used during migration was 5 m. These short shallow dives allow the turtles to swim more efficiently by reducing drag which increases at the surface and decreases when the depth is 2-3 times the body thickness (Hays et al. 2001).

Information on the diet and nutrition of the Central North Pacific DPS green turtle has been comprehensively reviewed by Mortimer (1982) and Bjorndal (1985). Algae and sea grasses occur in the insular Pacific region, but their distribution, abundance, and utilization by green turtles are only known for a few areas. Seagrasses are rare in the Mariana, Marshalls and Hawaiian Islands and absent in American Samoa and the unincorporated islands. At certain nearshore habitats of the Hawaiian Islands, green turtles feed heavily on benthic algae of the genera *Codium*, *Amansia*, *Pterocladia*, *Ulva*, and *Gelidium*. In addition, two ubiquitous species of exotic algae, *Acanthophora spicifera* and *Hypnea musciformis*, are widely consumed. The former was believed to have been accidentally introduced from Guam on the hull of a barge during the early 1950s. The latter species was intentionally brought from Florida in 1974 to the island of Oahu (and subsequently spread on its own elsewhere) as part of a seaweed farming experiment (Balazs et al. 1987; Russell and Balazs 1994). Green turtles in the Hawaiian Islands also forage on *Halophila hawaiiana*, the endemic sea grass that occurs there, as well as the introduced seagrass, *Halophila decipiens* (Russell et al. 2003). However, seagrass is not widespread nor abundant except for a few localized sites such as in Kaneohe Bay on Oahu, South Molokai, and Anini on Kauai. Green turtles resident to Johnston Atoll feed almost exclusively on the green algae, *Caulerpa racemosa* and *Bryopsis pennata*. These two species grow in abundance along the south shore of the main islet (Johnston Island) where an untreated human sewage outfall and chemical weapons incineration plant are located (Balazs 1985a). Green turtles living in nearshore reef habitats of Hawaii primarily eat red and green algae or seaweed (locally known as limu). Of approximately 400 species of seaweed present in the Hawaiian archipelago, nine species account for the majority of green turtle diet and a non-native red algae, *Acanthophora spicifera,* is a common item in their diet. Other food items they consume in lesser amounts include jellyfish, salps, mollusks, sponges, and tubeworms. Important foraging areas are located along the coastlines of Oahu, Molokai, Maui, Lanai, Hawaii, Lisianski Island, and Pearl and Hermes Reef.

In the Hawaiian Islands, green turtles migrate to FFS from foraging pastures located both to the northwest and the southeast, involving one-way distances up to 1300 km. In 1991, three females were satellite-tracked from their nesting grounds at FFS to foraging grounds in Kaneohe Bay and Johnston Atoll. Green turtles from the nesting beaches in the FFS nesting site remain resident to foraging pastures throughout the archipelago (with the possible exception of the oceanic juvenile phase, for which there are no available data and which genetic sampling has yet to reveal) and are the exclusive nesting population present in these areas (Balazs 1976; Craig and Balazs 1995; Keuper-Bennett and Bennet 2002; P. Dutton, NMFS, pers. comm. 2013), both in the southern foraging areas of the main islands (Balazs et al. 1996) as well as at northern foraging areas at Midway Atoll (Balazs et al. 2005).

Rice and Balazs (2008) studied the diving behavior of an adult female and two adult male green turtles during their roundtrip breeding migration from Laniakea, Oahu, to FFS. All three turtles exhibited a biphasic diving behavior. During the daylight hours, diving depth was shallow (1–4 m) and duration was short (1–18 min.). It was assumed that the turtles were moving deliberately toward their destination during this time. Between 1900 and 1930 h daily, the turtles began a diving pattern consisting of deep dives with a mean maximum dive depth of 35–55 m and a mean duration of 35 to 44 min. The shallow diurnal diving began between 0600 and 0700 h, after the nocturnal deep-diving pattern ended. The adult female made two dives in excess of 135 m and one male made several dives in excess of 100 m. These are the deepest dives ever recorded for a naturally diving green turtle. It took an average of 36 days for the turtles to make the trip to FFS and an average of 30 days to make the return trip. The deep nocturnal diving was unexpected and this behavior is in need of further investigation.

If foraging and diving behavior relevant to a specific DPS is present, additional information will be provided in their applicable discussions.

Juvenile cohorts - a comprehensive evaluation

The following is general information that is likely applicable to all green turtle DPSs. There is a lack of DPS-specific information for juvenile green turtles. Green turtle hatchlings pip and escape from their eggs and then move upward and out of the nest over a period of several days (Hendrickson, 1958; Carr and Ogren, 1960). Hatchlings emerge from their nests *en masse* almost exclusively at night and presumably use decreasing sand temperature (i.e. nighttime) as a cue (Hendrickson, 1958; Mrosovsky 1968). Glen et al. (2006) concluded the most likely cue for green turtle hatchling emergence was subsurface sand temperatures (10–20 cm), with emergence inhibited when temperatures were increasing, and most emergences occurring during nighttime hours. After an initial emergence, there may be secondary emergences on subsequent nights (Carr and Ogren 1960; Witherington 1986). Immediately after hatchlings emerge from the nest, they begin a period of frenzied activity. During this active period, hatchlings crawl to the surf, swim, and are swept through the surf zone (Carr and Ogren 1960; Carr 1961; Wyneken and Salmon 1992). Hatchlings first use visual cues, orienting to the brightest horizon, which is over the ocean on natural beaches without artificial lighting (Daniel and Smith 1947; Limpus 1971; Salmon et al. 1992; Witherington and Martin 1996; Witherington 1997; Stewart and Wyneken 2004). After reaching the surf, hatchlings frenzy swim away from the beach and are swept through the surf zone, after which wave orientation occurs in the nearshore area and later magnetic field orientation as they proceed further toward open water (Lohmann and Lohmann 2003).

Upon leaving the nesting beach and entering the marine environment post-hatchling green turtles begin an oceanic juvenile phase during which time they are presumed to primarily inhabit areas where surface waters converge to form local downwellings, resulting in linear accumulations of floating material, especially *Sargassum sp*. This association with downwellings is well documented for loggerheads, as well as for some post-hatchling green turtles (Witherington et al. 2006, 2012). The smallest of oceanic green turtles associating with these areas are relatively active, moving both within *Sargassum sp.* mats and in nearby open water, which may limit the ability of researchers to detect their presence as compared to relatively immobile loggerheads (Smith and Salmon 2009; Witherington et al. 2012).

The neritic juvenile stage begins when green turtles exit the oceanic zone and enter the neritic zone (Bolten 2003). The age at recruitment to the neritic zone likely varies with individuals leaving the oceanic zone over a wide size range (summarized in Avens and Snover 2013). Using skeletochronology, Goshe et al. (2010) estimated the duration of the oceanic juvenile stage to be between 1 and 7 years (mean=3, SD=1.6) in the northwestern Atlantic, with juveniles recruiting to neritic habitats over a size range of 19–30 cm SCL (Mendonça 1981; Goshe et al. 2010). Age and size at recruitment have been estimated at 5-6 years and 35-40 cm CCL, in the Eastern Pacific Ocean (Seminoff et al. 2003).

After migrating to the neritic zone, juvenile green turtles continue maturing until they reach adulthood, and some may periodically move between the neritic and oceanic zones (NMFS and USFWS 2007; Parker et al. 2011).

Food items documented for a limited number of stranded post-hatchling green turtles have included predominantly *Sargassum* spp. and associated hydroids, bryozoans, polychaetes, gastropods, as well as cnidarians and other pelagic invertebrates, fish eggs, and debris (Witherington et al. 2006; Boyle and Limpus 2008; Jones and Seminoff 2013). In the Eastern Pacific Ocean, green turtles reportedly forage on a greater proportion of invertebrate foods, with omnivorous diets reported in turtles throughout the region (Seminoff et al. 2003; López-Mendilaharsu et al. 2005; Amorocho and Reina 2007; Carrión-Cortez et al. 2010; Lemons et al. 2011).

During their pelagic phase, juvenile green turtles feed omnivorously on a range of planktonic material including crustaceans, jellyfish and ctenophores. Sub-adults and adults take tuna hooks baited with squid or fish, as demonstrated by bycatch of green turtles in several tuna longline fisheries in the Pacific (Beverly and Chapman, 2007). Sub-adult Eastern Pacific green turtles have been caught in the HI SSLL and DSLL fisheries which fish at depths less and 100 m and greater than 100 m respectively. It is not known if the turtles are caught at depth during the soak of the gear or when the gear is being set or hauled. Very little is known of juvenile or adult green turtle pelagic foraging behavior, such as foraging depth. The deepest dives recorded for green turtles are from adults migrating from the main Hawaiian Islands to the NWHI. 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 night time 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.

4.2.3.1 Central North Pacific DPS

Differential Distribution

The Central North Pacific green turtle DPS occurs in the Hawaiian Archipelago and Johnston Atoll (Figure 19). The Hawaiian Archipelago is the most geographically isolated island group and, therefore, green turtles in this DPS are geographically discrete in their range and movements, as evidenced by mark-recapture studies using flipper tags, PIT tags, satellite-linked transmitter tracking, and genetic analyses (Seminoff et al. 2015). After hatching, juveniles spend at least several years in pelagic areas where they feed primarily on small invertebrates. Between six to 10 years of age, at approximately 40 cm CCL, most green turtles recruit to coastal habitats.

Average size at recruitment to these neritic habitats for Pacific green turtles ranges from 35-50 cm Curved Carapace Length (CCL) (Balazs 1980; Limpus et al. 2003).

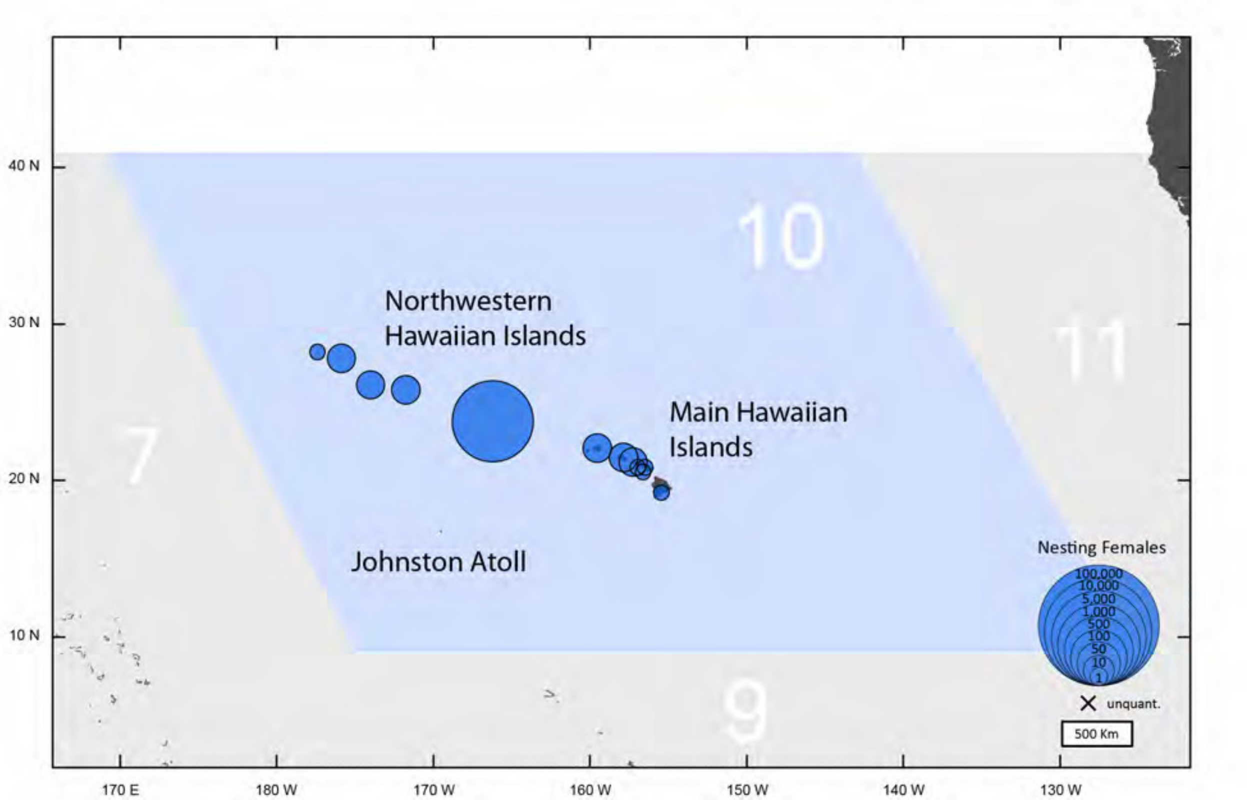

Figure 19. Geographic area of the Central North Pacific DPS. Size of circles indicates estimated nester abundance (see Section 14.2.1). DPS encompasses the entire Archipelago of Hawai'i and Johnston Atoll.

While the proposed action includes waters of the Hawaiian Islands, longline fishing does not occur within 75 nm from the main Hawaiian Islands. Adults migrate directly between the main Hawaiian Islands and FFS (Balazs 1994), but the proposed action is unlikely to encounter many migrating adult green turtles from the Central North Pacific DPS (Figure 19).

Population Dynamics

Structure

The primary demographic features of green turtles that are relevant for interpreting population structure and long term trends include age-to-maturity (often via growth studies), reproductive longevity, sex ratio, reproductive output (i.e. egg production, clutch frequency, hatching success, internesting interval), and annual survivorship.

Consistent with slow growth, age-to-maturity for the green turtles appears to be the longest of any sea turtle species (Chaloupka and Musick 1997; Hirth 1997). Hawaii green turtles are known to mature at a size of 80+ cm SCL. USFWS and NMFS (1998) reported an overall average annual rate of growth of about two cm/year for all size classes ranging from 35-82 cm and the minimum size observed for nesting females in Hawaii was 81 cm SCL (Balazs 1982b). Based on these data, it is estimated that an average of at least 25 years would be needed to achieve sexual maturity (i.e. first breeding migration to FFS) (USFWS and NMFS 1998).

Flipper tag returns and satellite tracking studies demonstrate that post-nesting females in the NWHI return to their foraging grounds in the MHI, and that foraging remains exclusively within geographic boundaries of this DPS. Demographic studies of green turtles do not reveal any structuring of traits within the DPS, although variable ecosystem productivity has produced differences in body conditions of nearshore foraging turtles (Balazs and Chaloupka 2004b; Wabnitz et al. 2010). Nesters at the primary nesting area of FFS average 92.2 cm SCL, have an internesting interval of 13.2 days, clutch size of 92.4 eggs, and a clutch frequency of 4 nests (Balazs 1980; Niethammer et al. 1997; Tiwari et al. 2010). Peak nesting in this DPS occurs from May through August (1980), and nesters return to breed at an interval of 4 years (G. Balazs, NMFS, pers. comm. 2013). Nest-level inventories are incomplete for this DPS, however, Balazs (1980) estimated hatchling emergence at 71%. Though previous estimates of age at first reproduction were as high as 35-50 years (Balazs 1980; Zug et al. 2002), a recent study examining capture histories, skeletochronology, and the observed nesting time series estimated green turtles in this DPS begin breeding on average at 23 years (K. Van Houtan NMFS pers. comm. 2013). As a result of the unusual nesting concentration at one site, typically studied population variables such as mean nesting size, nesting season, inter-nesting interval, clutch size, hatching success, nesting season, and clutch frequency have not been compared among multiple nesting sites within this DPS.

Genetic sampling in the Central North Pacific DPS has been extensive and representative given that there are few nesting populations in this region. Results of mitochondrial Deoxyribonucleic Acid (mtDNA) analysis indicate a low level of spatial structure with regard to minor nesting around the MHI and the NWHI although the same haplotypes occur throughout the DPS. Within the NWHI, studies show no significant differentiation (based on mtDNA haplotype frequency) between FFS and Laysan Island (P. Dutton, NMFS, pers. comm. 2013). Frey et al.'s (2013) analysis of low level of scattered nesting on the MHI (Molokai, Maui, Oahu, Lanai and Kauai; mtDNA and nDNA) showed that nesting in the MHI might be attributed to a relatively small number of females that appear to be related to each other, and demographically isolated from FFS. Frey et al. (2013) suggest that the nesting population at the MHI may be the result of a few recent founders that originated from the FFS breeding population. This regional range expansion may buffer against the loss of current nesting sites at FFS due to sea level rise. The current nesting distribution represents a curtailment of nesting activities within the Central North Pacific DPS's historic range.

There is little gene flow between Hawaii and other populations throughout the Pacific Ocean. The vast majority of nesting takes place in the NWHI, and the largest rookery is at East Island, FFS, where approximately 50% of Hawaiian nesting occurs, and where continuous monitoring has occurred since 1973 (Balazs 1980; Balazs and Chaloupka 2004b). East Island is a small sandy atoll, ~0.036 km2, surrounded by a shallow lagoon on the western side and a fairly steep shelf on the eastern side. Generally, nesting site fidelity at FFS is high (> 95%; Balazs 1980; Dizon and Balazs 1982; Bowen et al. 1992). Females generally will not nest in the year immediately following a nesting year (i.e. obligate skip-nesting) and vary in the number of years between nesting migrations (Piacenza et al. 2016). The length of the remigration interval depends on energetics, physiology, and environmental conditions (Chaloupka and Limpus 1996; Limpus and Chaloupka 1997; Miller 1997; Solow et al. 2002). Within a nesting season, Hawaiian green turtles will return several times to lay an average of 4 clutches at approximately 2 week intervals (Balazs 1980; Tiwari et al. 2010), so the probability of sighting an individual turtle at least once during a season in which she breeds is high (Piacenza et al. 2016).

Balazs et al. (2015) estimated remigration interval of Hawaiian green turtles, summing data from 1973 to 2013, to be 4 yr. (breeding probability = 0.25), in contrast to an earlier estimate of 2.47 yr. (breeding probability = 0.404; Balazs 1980). Balazs et al. (2015) also found the mean body length for adult females in Hawaii to be 90.7 cm SCL, reduced from the original 1980 estimate of 92.2 cm.

Piacenza et al. (2016) estimated the demographic indicators (DIs) of size at maturity, nester carapace length, breeding probability, and adult female survival using 3677 tagged nesting green turtles from 1973 to 2010 in Hawaii and reported that mean nester carapace length (range: 89.21−91.69 cm) and breeding probability (range: 0.0766−0.444 yr−1) showed directional changes over time, suggesting shifts in age structure that could be due to recruitment. Breeding probability, or the probability of transitioning from foraging to nesting, was estimated annually and ranged from 0.0766 (95% CI: 0.0574− 0.102) in 1998 to 0.444 (95% CI: 0.398−0.490) in 1997. If these probabilities were constant for individuals, they would correspond to remigration intervals of 3.3 to 14.1 yr. The top-ranking model also included a constant estimate of transition probability of nesters moving to the foraging state, 0.989 (95% CI: 0.925−0.998). This result corroborates a wide body of previous research that nearly all individual females will skip-nest the year immediately following nesting, and take at least 2 years to return to the nesting beach, and likely much longer on average (Miller 1997, and references therein).

Piacenza et al. (2016) did not find evidence that survival rates varied over time from 1980 to 2009 (S = 0.929, 95% CI: 0.924− 0.933). Following ESA protection, survival rates were expected to increase, especially as adults were targeted for harvest towards the end of the fishery. However, survival rates had likely already increased by 1980 (the first year included in the MSORD model), as there had been a campaign to protect green turtles in Hawaii in the 1970s; the state of Hawaii passed regulations restricting harvest starting in 1974 (Balazs 1980), and ESA protection formally occurred in 1978. Piacenza et al. (2016) results are comparable to the findings of Van Houtan et al. (2014), who used matrix models to deterministically estimate survival rates in Hawaiian green turtles. Adult female survival rates in Hawaii are relatively high compared to other populations. For example, in Costa Rica, where nesting turtles are impacted by fisheries in Nicaraguan waters, adult female survival is estimated to be 0.85 (95% CI: 0.83−0.87) and in Venezuela, where anthropogenic impacts are high, adult female survival is estimated to be 0.79 (95% CI: 0.73−0.84), but in Australia, where anthropogenic impacts are minimal, adult female survival is estimated to be 0.9485 (95% CI: 0.92−0.98)(Chaloupka and Limpus 2005; Troëng and Chaloupka 2007; Garcia-Cruz et al. 2015).

Diversity

Diversity is a critical element considered under the status review (Seminoff et al. 2015) for green turtles and includes consideration of the overall nesting spatial range, diversity in nesting season, diversity of nesting site structure and orientation, and the genetic diversity within the DPS. With nesting in the Central North Pacific DPS is unusually concentrated at one site there is little diversity in nesting areas. Remnant nesting aggregations may have existed in the MHIs as recently as the 1930s, but were no longer present in the 1970s (Balazs 1976). For example, an important green turtle nesting colony once nested on Lanai, and several select beaches on Molokai, Oahu, and Kauai were also used by green turtles (Balazs 1975b). Balazs (1980) reported that the distribution of green turtles in the Hawaiian Archipelago has been reduced within historical times. A more recent study (Kittinger et al. 2013) supports this finding and further suggests that there has been a significant constriction in the spatial distribution of important reproduction sites, presenting a challenge to the population's future and making this DPS highly vulnerable. As much as 80% of historically major nesting populations could be extirpated or have heavily reduced nesting abundances, and what was once geographically distributed nesting is now concentrated at a single site (Kittinger et al. 2013). The one nesting site, FFS, is a low-lying coral atoll that is susceptible to erosion, geomorphological changes and sea level rise, and has already lost significant nesting area (Baker et al. 2006). However, aside from sea level rise, FFS is relatively isolated from anthropogenic threats. The entire NWHI, which represents more than 98% of nesting in this DPS, lies within the Papahanaumokuakea Marine National Monument, a remote Monument that has controlled access for activities that occur within it. mtDNA studies indicate that there is a low level of stock substructuring among nesting sites in the Central North Pacific DPS and a relatively low level of diversity based on a total of four closely related mtDNA haplotypes identified (P. Dutton, NMFS, pers. comm. 2013).

Principal nesting sites for green turtles in the Central North Pacific DPS since 1960 is French Frigate Shoals (FFS) (Balazs 1980; Lipman and Balazs 1983); where 96% of the population currently nests. Approximately 50 of that population nests on East Island which was approximately 11 acres in size. In late October of 2018, Hurricane Walaka decimated East Island. At the time of this writing, NMFS is unclear what impacts will be seen in the population from this event. No green sea turtles were utilizing the island for nesting or basking when the hurricane struck, although it is estimated 19% of the nests at East Island and 20% of nests at nearby Tern Island had yet to emerge and were destroyed in the storm (C. Littnan pers. comm. 2018). There is a potential that sand will redeposit over the winter. Additionally, a large amount of sand were deposited on Tern Island. It is unclear whether the turtles will shift nesting locations to Tern Island. However, nesting was historically abundant at various sites across the archipelago as recently as 1920 (Kittinger et al. 2013). The other islets of FFS—Tern, Trig, Gin, and Little Gin—account for the remainder. Whale-Skate, joined by sand deposition between the former islets of Whale and Skate in the 1950s, eroded and became submerged in 1997 (Baker et al. 2006). Nesting by green turtles occurs in low numbers throughout the NWHI at Laysan, Lisianski, Pearl and Hermes Reef, and very uncommonly at Midway and Kure Atoll. Since 2000, green turtle nesting on the MHI has emerged in low numbers on 7 islands (Frey et al. 2013; Kittinger et al. 2013; PIFSC, unpublished data 2013). NMFS will be monitoring the population closely in FFS in 2019 to assess the turtle's response and subsequent impacts from the loss of East Island.

From 1965 to 2013, 17,536 green turtles have been tagged involving all post-pelagic size classes from juveniles to adults. With only three exceptions, the 7,360 recaptures of these tagged turtles have been made within the Hawaiian Archipelago. The three outliers involved a recovery in Japan, one in the Marshall Islands and one in the Philippines. FFS, located in the NWHI, represents the prominent focal point of green turtle nesting and hatchling production in the Hawaiian Archipelago (Figure 20). Information from tagging at FFS, other areas in the NWHI, areas in the Main Hawaiian Islands (MHI), and Johnston Atoll show that the vast majority of reproductive females and males periodically migrate to FFS for seasonal breeding from these distant locations. At the end of the season, they return to their respective foraging areas. Conventional tagging using PIT and metal flipper tags have documented 164 turtles making reproductive movements from or to FFS and foraging pastures in the MHI, and 58 turtles from or to FFS and the foraging pastures in the NWHI.

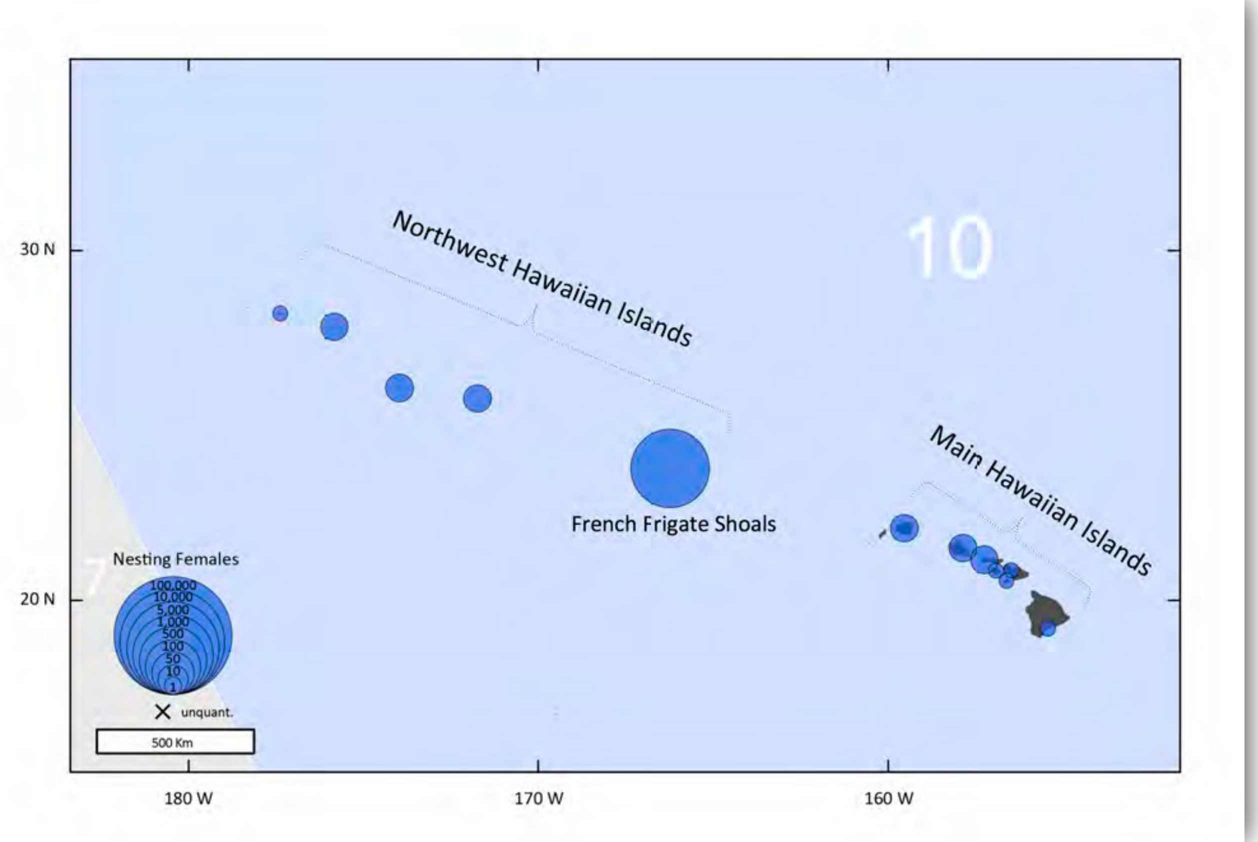

Figure 20. Close up of nesting green turtles in the Central North Pacific DPS. Size of circles indicates estimated nester abundance.

Status

Since initial nesting surveys at the FFS index beach in 1973, there has been a marked increase in annual green turtle nesting (Balazs and Chaloupka 2004). IUCN Redlist categorizes their status 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,with 61,000 in coastal habitats around Hawaii (IUCN 2012; Chaloupka and Balazs 2007). The historic population size is estimated to be approximately 320,000 individuals (IUCN 2012)

This increase over the last 40 years corresponds to an annual increase of 5.4% (Balazs et al. 2015). Between 1973 and 2015, nesting activity has been variable, as is typical of green turtle nesting dynamics, ranging between a low of 67 in 1973 and an all-time high of 808 nesting females observed during the 2011 six-week sampling period at East Island, FFS (with a total estimate of 843 nesters for the season) (NMFS-PIFSC unpubl. data). Surveys in 2013 were not be possible due to a December 2012 storm that destroyed the FFS field station making residence for biologists unsafe. Monitoring resumed in 2014. In-water abundance of green turtles is consistent with the increase in nesting trends (Balazs 1996; Balazs and Chaloupka 2004; Chaloupka et al. 2007). In addition, there has been a dramatic increase in the number of basking turtles in the main Hawaiian Islands and throughout the NWHI (Balazs 1996; Balazs and Whittow 1982; Parker and Balazs 2010). Long-term monitoring of the population indicates a strong degree of island fidelity exists within the rookery, and tagging studies have shown that turtles nesting at FFS come from numerous foraging areas where they reside throughout the Hawaiian Archipelago (Balazs 1976; Balazs 1980, 1983; Dutton et al. 2008). This linkage has been firmly established through genetics, satellite telemetry, flipper tagging and direct observation (Balazs 1983, 1994; Leroux et al. 2003; Dutton et al. 2008). The increase of the long-term nester trend can be attributed to increased survivorship (since harvesting of turtles in foraging grounds was prohibited in the mid-1970s) and cessation of habitat damage at the FFS rookery since the early 1950s (Balazs and Chaloupka 2004).

While the nesting population trajectory is positive and encouraging, more than 96% of nesting occurs at one site in the NWHI and it is highly vulnerable to threats. Results of mtDNA analysis indicate a low level of spatial structure and low genetic diversity within the DPS. Survival of this DPS is currently highly dependent on successful nesting at FFS (Niethammer et al. 1997). There has been a significant constriction in the spatial distribution of important reproduction sites, presenting a challenge to the population’s future and making this DPS highly vulnerable. As much as 80% of historically major nesting populations could be extirpated or have heavily reduced nesting abundances, and what was once geographically distributed nesting is now concentrated at a single site (Kittinger et al. 2013). The one nesting site, FFS, is a low-lying coral atoll that is susceptible to erosion, geomorphological changes and sea level rise, and has already lost significant nesting area (Baker et al. 2006). As previously discussed, in October of 2018, East Island was demolished by Hurricane Walaka. Impacts to the population are unknown at this time and warrant further research moving forward.

In summary, the concentrated nature and relatively small size of the DPS make it vulnerable to random variation and stochasticities in the biological and physical environment, including natural catastrophes, as well as changes in climate and resulting effects such as sea level rise. This increases its risk of extinction, even though it may have positive population growth (Seminoff et al. 2015). Both non-stochastic as well as stochastic events are significant current and future threats to this small, isolated, concentrated population.

There are many ongoing conservation efforts for green turtles in the Central North Pacific DPS by numerous Federal and State agencies and other non-governmental organizations. Green turtles in this DPS are protected by the ESA and in Hawaii, they are also protected by the Hawaii Revised Statutes, Chapter 195D (Hawaii State Legislature, accessed 9/10/2010) and Hawaii Administrative Rules, 13-124 (Hawaii Administrative Rules, accessed 9/10/2010), which adopt the same definitions, status designations, and prohibitions as the ESA and carry additional penalties for violations at the State government level. These two statutes have been, and currently are, key tools in efforts to recover and protect this DPS, and both have been effective in improving the status of sea turtles in Hawaii. Non-governmental organizations assist in the conservation of Hawaii’s green turtles by conducting public outreach programs, protecting basking green turtles, conducting beach monitoring of turtles, and conducting in-water surveys. Intensive monitoring and protective efforts are ongoing in the NWHI, in the main Hawaiian

Islands, and in nearshore waters. Debris “clean up” efforts are also conducted in Hawaii by the NOAA Marine Debris Program and non-government organizations (Friedlander et al. 2008).

The State of Hawaii’s Department of Land and Natural Resources efforts to conserve green turtles include wildlife regulations, coordination of stranding response and specimen storage on some islands; issuance and management of special activity permits; statewide outreach and education activities; and nest monitoring on Maui (Department of Land and Natural Resources, 2013). The Department of Land and Natural Resources Division of Conservation and Resources Enforcement investigates reports of illegal poaching, provides support and security at some nest sites and strandings, and addresses complaints from the public regarding turtle disturbances. Through ESA Section 6 (Species Recovery Grant) funding, the Department of Land and Natural Resources is working cooperatively with NMFS to minimize threats to green turtles in the main Hawaiian Islands.

To raise awareness among fishermen to reduce impacts to sea turtles from coastal nearshore hook and line fisheries around the Main Hawaiian Islands, a multiagency partnership that includes NMFS, the State of Hawaii, the Western Pacific Fisheries Management Council, local experts, and fishers has developed a "Fishing Around Sea Turtles" program. The program was developed in 2010 (and has been refined and improved since) is designed to promote "Turtle Friendly" fishing gear, such as barbless circle hooks, and provide best-practice guidelines for fisherman to reduce injury or mortality following an accidental interaction. Including practical fishing tips suggested by fishermen that may reduce the potential for interactions, and encourages reporting injured or dead turtles to NMFS’ sea turtle stranding program.

The Papahanaumokuakea Marine National Monument in the NWHI is a conservation area that encompasses coral reefs, islands, and shallow water environments that are important habitats for green turtles. The Monument is working to reduce threats through an ecosystem approach to management. This includes the development of an effective regulatory framework and permitting process, education and outreach, preventative measures to minimize risk, response, and restoration to damaged or degraded natural resources. The Pacific Remote Islands Marine National Monument was established in January 2009. The areas extend 50 nautical miles from the mean low water lines and include green turtle habitat. The protected area provides some protection to sea turtles and their habitat (through permitted access) and its remoteness. On August 26, 2016, President Obama issued Proclamation 9478 establishing the Papahanaumokuakea Marine National Monument Expansion (81 FR 60227). The Expansion area includes waters and submerged lands in the U.S. EEZ west of 163 W Longitude adjacent to the PMNM. The Monument Expansion consists of approximately 442,781 square miles. The Proclamation directs the Secretaries of Commerce and Interior to prohibit various activities, including commercial fishing, while allowing for sustainable non-commercial fishing and Native Hawaiian practices. The Western Pacific Fisheries Management Council is currently evaluating options for developing regulations to implement the commercial and non-commercial fishing provisions of the Proclamation.

At least 16 international treaties and/or regulatory mechanisms that apply to green turtles regionally or globally apply to green turtles within the Central North Pacific DPS. This includes: Convention on Biological Diversity, Convention on International Trade in Endangered Species, Indian Ocean-South-East Asian Marine Turtle Memorandum of Understanding, Inter-American Convention for the Protection and Conservation of Sea Turtles, and Secretariat of the Pacific Regional Environment Programme. Regulatory mechanisms in U.S. jurisdiction are in place through the ESA, Magnuson Stevens Act, and the State of Hawaii that currently address direct and incidental take of Central North Pacific green turtles, and these regulatory mechanisms have been an important factor in the increasing trend in this DPS.

In 2008, the Western and Central Pacific Fisheries Commission issued a Conservation and Management Measure (2008–03; https://www.wcpfc.int/doc/cmm-2008- 03/conservation-and-management-sea- turtles) to reduce sea turtle mortality during fishing operations, collect and report information on fisheries interactions with turtles, and encourage safe handling and resuscitation of turtles. This measure requires purse seine vessels to avoid encircling turtles and to release entangled turtles. It also requires longline vessels to use line cutters and de-hookers to release turtles.

Threats to the Species

Natural Threats

Central North Pacific green turtles are exposed to a variety of natural threats both at their nesting beaches and in the open ocean. The beaches where turtles nest and the nest themselves are threatened by hurricanes, tropical storms, storm surges, sand accretion, and rainfall associated with storms. Predators such as sea birds, and ghost crabs (*Ocypode* spp.) hunt the hatchlings. Sharks and other fish will also eat the hatchlings when they enter the marine environment. Large grouper (*Epinephelus tauvina*) are documented predators of post-hatchling green turtles in Hawaii; however, the extent of grouper depredation is unknown (Balazs 1995). Adult turtles are also killed by sharks and other marine predators (Seminoff et al. 2015).

Fibropapillomatosis affects green turtles from this DPS (Francke et al. 2013). This disease results in internal and/or external tumors (fibropapillomas) that may grow large enough to hamper swimming, vision, feeding, and potential escape from predators. The prevalence of the disease has declined since 1990s but persists in the population at varying levels throughout the Hawaiian archipelago (Chaloupka et al. 2009; Van Houtan et al. 2010). Strandings attributed to fibropapillomatosis as the primary cause of stranding peaked in 2005 (110 turtles) but has since declined over time to between 40 and 60 strandings per year between 2012 and 2017 (PIFSC 2018 unpublished). Due to limitations of stranding data, the extent of the disease or exact numbers of turtles with the disease are unknown as recovered strandings (sick, injured or dead turtles) are an unknown fraction of green turtles with fibropapillomatosis. Environmental factors may be significant in promoting fibropapillomatosis, and eutrophication (increase in nutrients) of coastal marine ecosystems may promote this disease (Van Houtan et al. 2010). Fibropapillomatosis remains an important concern. This is particularly true given the continued, and possibly future increasing, human impacts to, and eutrophication of, coastal marine ecosystems that may promote this disease. Spirorchid (blood fluke) infections are reported for Central North Pacific green turtles (Greenblatt et al. 2005; Work et al. 2005).

In June of 2018 volcanic eruptions on the Big Island of Hawaii caused lava flows to fill in tidal pools around the island. We have several anecdotal reports of turtles that were trapped in these pools and died. We do not have an estimate of the total numbers of turtles that were killed due to the lava flows.

Anthropogenic Threats

Coastal development and construction, vehicular and pedestrian traffic, beach pollution, tourism, and other human related activities is an increasing threat to the basking and nesting population in the main Hawaiian Islands (currently very limited) and may negatively affect hatchling and nesting turtles on these beaches. Climatic changes in the NWHI pose threats through reduction in area of nesting beaches critical to this DPS (Baker et al. 2006). The primary nesting area for the Central North Pacific DPS is threatened by sea level rise. For example, Whale-Skate Island at FFS was formerly a primary green turtle nesting site for this DPS but the island has subsided and is no longer available for nesting (Kittinger et al. 2013).

Threats to green turtle habitat in neritic and/or oceanic zones of the Central North Pacific DPS include contamination and degradation of foraging areas due to nearshore development, land based sources of marine pollution and increased human activity, contamination due to past military practices, vessel groundings, and fishing practices. Impacts to the quality of coastal habitats in the main Hawaiian Islands are a threat to this DPS and are expected to continue and possibly increase with an increasing human population and annual influx of millions of tourists. Loss of foraging habitat or reduction in habitat quality in the main Hawaiian Islands due to nearshore development is a threat to this DPS. Marina construction, beach development, siltation of forage areas, contamination of forage areas from anthropogenic activities, resort development or activities, increased vessel traffic, and other activities are all considered threats to this DPS and its habitat (Seminoff et al. 2015). In general, main Hawaiian Islands coral reefs have suffered from land-based sources of pollution, overfishing, recreational overuse, invasive species, and are threatened by climate change and increased temperatures resulting in coral bleaching events, coral disease, coastal development and runoff, and waste water (point and non-point source pollution) (Friedlander et al. 2008). Climate change influences on water temperatures, ocean acidification, sea level and related changes in coral reef habitat, wave climate and coastal shorelines are expected to continue.

Incidental bycatch in fishing gear, marine pollution, and interactions with recreational and commercial vessels, all negatively affect green turtles in the Central North Pacific DPS. Three of the most common reasons for sea turtle strandings in Hawaii are entanglement in fishing gear, interactions with fishing hooks, and interaction with marine debris (usually entanglement in nets). Human disturbance (e.g., by tourism) of foraging and basking sea turtles can occur in Hawaii, however it is unclear what level of threat this disturbance presents. Interactions between Central North Pacific green turtles and nearshore fisheries in the main Hawaiian Islands can result in entanglement, injury, and mortality. The number of reported strandings are expected to be a smaller subset of the actual level of interaction with this gear. Nearshore fishery interactions have increased over time and are currently the primary cause of stranding of sea turtles in Hawaii with over 100 turtles stranded dead or injured each year due to fishery interactions in hook and line gear, with an additional 10 to 20 turtles caught in gillnets per year (PIFSC MTBAP unpublished). NMFS and its partners are cooperating to reduce the impact on green turtles from hook-and-line fishing (see below).

NMFS estimates that between 20-64% of the turtles caught in the HI SSLL fishery could be from the Central North Pacific DPS (P. Dutton pers. comm. November 20, 2017). This means that 2-6 turtles are estimated to have been captured from this DPS by the HI SSLL fishery since 2004. NMFS estimates that between 4-43% of the turtles caught in the deep-set longline fishery could be from the Central North Pacific DPS (P. Dutton pers. comm. June 29, 2018). This means that 3-29 turtles are estimated to have been captured from this DPS by the deep-set longline fishery since 2004. The American Samoa longline fishery does not interact with this DPS based on the genetic analysis and what we know about the migrations and foraging behavior of this DPS.

4.2.3.2 East Pacific DPS

Differential Distribution

The East Pacific DPS extends from the California/Oregon border, USA (42°N) southward along the Pacific coast of the Americas to central Chile (40°S). The northern and southern boundaries of this DPS extend from the aforementioned locations in the USA and Chile to 143°W and 96°W, respectively. The offshore boundary of this DPS is a straight line between these two coordinates. This DPS encompasses the Revillagigedos Archipelago (Mexico) and Galapagos Archipelago (Ecuador).

Adult Foraging and Diving

See *Adult Foraging and Diving* above for information on green turtle general feeding and diving behavior. Blanco (2010) observed that the Costa Rica population did not extensively use the 20 m depth that was suggested as neutral buoyancy for green turtles (Hays et al. 2000) and the average depth where green turtles may rest (Hays et al. 2001). However, buoyancy control may not be a unique strategy for resting sea turtles (Seminoff et al. 2006b). Blanco (2010) did not find differences between depths and durations during day and night during migration. The significant proportion of shallow and short dives during the night indicated that these turtles were not achieving neutral buoyancy to rest but were resting very close to or at the surface.

Blanco (2010) reported that East Pacific green turtles moved between 5 and 1091 km after nesting. Some turtles were local residents in the Gulf of Papagayo, some moved an intermediate distance, and others moved to Panama and El Salvador. It appeared that Nombre de Jesús was an important nesting beach for turtles residing in waters throughout Central America. Turtles in this population did not undertake oceanic migrations reported for other sea turtles (Green 1984; Carr 1975; Plotkin et al. 1995; Seminoff et al. 2008). Rather they stayed within coastal areas even when migrating over 1000 km. Even though it may not be the shortest distance to the preferred foraging area, it appears that these green turtles prefer to migrate along the coast in shallow waters.

The mean daily speed of travel during migration for East Pacific turtles Blanco (2010) studied was approximately 37 km/day. East Pacific turtles travelled slower than green turtles in the Caribbean of Costa Rica which travelled an average of 58 km/day to reach the foraging grounds (from 400 to 1090 km) north of the nesting beaches (Troeng et al. 2005). Blanco's (2010) findings suggest that the East Pacific turtles off Costa Rica spend less energy than other sea turtles in migration, reflected in the short distance traveled and short distance covered in a day, but they used broader areas while foraging than other green turtle DPSs and other turtle species.

Population Dynamics

Structure

The primary demographic features of green turtles that are relevant for interpreting population structure and long term trends include age-to-maturity (often via growth studies), reproductive longevity, sex ratio, reproductive output (i.e. egg production, clutch frequency, hatching success, internesting interval), and annual survivorship.

Size and age at first reproduction has been estimated for green turtles using several methods, including mark recapture, skeletochronology, and marked, known-aged individuals. Estimates vary widely among studies and populations, and methods continue to be developed and refined (Avens and Snover 2013). East Pacific green turtles are known to mature at smaller sizes (60–77 cm SCL) (Seminoff et al. 2002) than conspecifics in the Northwestern Atlantic (85–100+ cm SCL), Hawaii (80+ cm SCL), and Australia (95 cm CCL) (Avens and Snover 2013).

Age at first reproduction is known for two East Pacific turtle nesting sites: 12-26 years at Tortuguero, Costal Rica (Frazer and Ladner 1986) and 12-20 years with an average of 16 years at Quintana Roo, Mexico (Richards et al. 2011). Seminoff et al. (2002) reports 9–21 years to reach sexual maturity for East Pacific green turtles after they settle into neritic foraging areas on northwest Mexico. However, a study in San Diego Bay, found very high growth rates (median growth rate was 1.03 cm/yr (1.6 to 11.4 cm/yr) for all turtles, and was 4.9 cm/yr for turtles ≤90 cm) (Eguchi et al. 2012). These growth rates were one of the fastest for the species in temperate areas and comparable to those reported for tropical regions. (McDonald Dutton and Dutton 1998; Eguchi et al. 2012).

Within region variation for any of the demographic features may suggest a level of spatial structure for the East Pacific DPS. Among all nesting assemblages in the East Pacific DPS, the Revillagigedos Islands stands out as uniquely different from the remaining areas. Females nesting in Michoacán are substantially smaller than those nesting in the Revillagigedos (82 cm vs. 94 cm mean CCL) (Alvarado-Diaz and Figueroa 1990; Juarez-Ceron et al. 2003). The estimated age-to-maturity is 9–47 years. In-water survivorship is known for very few areas. 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).

Size of nesters ranges from 101.7 cm CCL (Campeche, Mexico) to 109.3 cm CCL (Isla Aguada, Mexico (Guzman-Hernandez 2001, 2006). The internesting interval ranges from 9 to 18 days (Witherington and Ehrhart 1989b; Johnson and Ehrhart 1996; Troeng et al. 2005; Hart et al. 2013) and on average, females lay 3 clutches per season (range estimated from 2.8 to 4.6 nests per season; Carr et al. 1978; Johnson and Ehrhart 1996; Guzman-Hernandez and Garcia-Alvarado 2009, 2010, 2011, 2013a, 2013b). Remigration intervals have been reported between 2 and 3 years (Troeng and Chaloupka 2007; Zurita et al. 1994; Witherington and Ehrhart 1989).

Mean clutch size varies greatly among green turtle populations, but on average is approximately 100 eggs per clutch (Hirth 1997). The mean clutch size of East Pacific turtles in Isla Aguada, Campeche, Mexico was 112.25 (Guzman -Hernández 2001, Hernández et al. 2008; Guzman -Hernández and G. Alvarad Tortuguero, Costa Rica was 108 (Tiwari et al. 2006). Hatching success rate was 92% in Mexico

(Xavier et al. 2006), although the high of 92% is an overestimate since nests that did not show signs of hatching were excluded from the analysis.

Diversity

Diversity is a critical element considered under the status review (Seminoff et al. 2015) for green turtles and includes consideration of the over-all nesting spatial range, diversity in nesting seasons, diversity in nesting site structure and orientation (e.g. high vs. low beach face, insular Pacific (U.S. territories) vs. continental nesting sites), and genetic diversity within the DPS. With respect to the spatial range of nesting East Pacific green turtles, this DPS has a very broad nesting range, with nesting occurring from the tip of the Baja California Peninsula to Northern Peru (Figure 21). Such a broad latitudinal range may be advantageous to green turtles in this DPS in the face of global climate change, sea level rise, increased storm events and loss of nesting habitat in vulnerable areas. The East Pacific DPS exhibits year round nesting at several sites and non-overlapping nesting seasons at others. Thus, it appears that this DPS my benefit from nesting season temporal diversity in relation to population resilience. This DPS also has a diversity of nesting site structure and orientation, with nesting at both continental and insular sites—the latter of which apparently has much lesser human threats. Thus, there are at least some relatively threat-free nesting refugia within this DPS's range. With respect to genetic diversity within the DPS, there is significant genetic substructuring within this DPS, perhaps suggesting a level of resilience to population genetic bottlenecks (Seminoff et al. 2015).

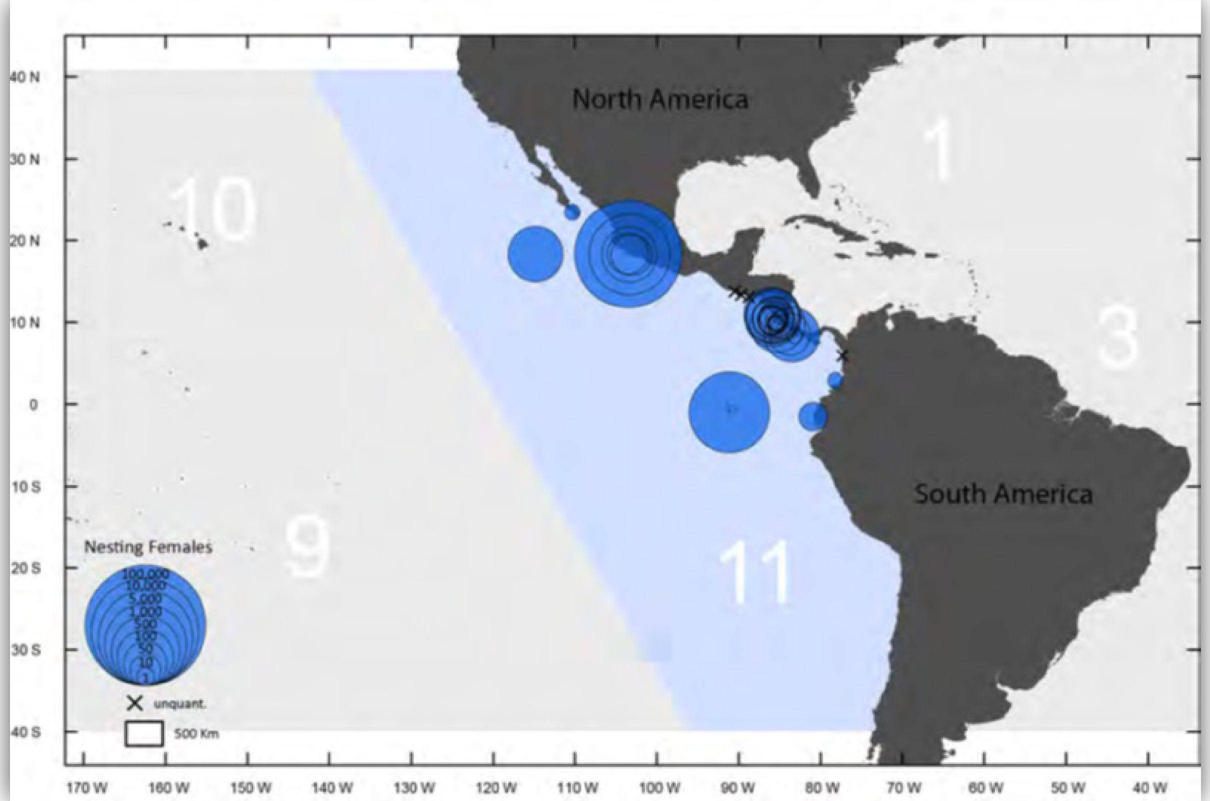

Figure 21. Nesting distribution of green turtles in the East Pacific DPS (blue-shaded area marked with '11'). Size of circles indicates estimated nester abundance.

Green turtle nesting is widely dispersed in the Eastern Pacific Ocean. The two largest nesting aggregations are found in Michoacán, Mexico and in the Galapagos Islands, Ecuador (Zarate et al. 2003; Delgado-Trejo and Alvarado-Figueroa 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).

The East Pacific population occurs within the action area. A total of 19 green turtle samples were collected by observers from 1996-2017 in the Hawaii-based shallow set longline fishery and were sequenced for assignment to DPSs (Dutton et al. 2017). Eleven green turtle samples were assigned to the Eastern Pacific DPS.

Seminoff et al. (2015) identified 39 total nesting sites for which abundance information is available. There are sporadic nesting events in many other areas in the East Pacific DPS, such as Guatemala and Peru, but nesting abundance is undocumented. Of these sites, there are two primary nesting concentrations (Michoacán, Mexico, and the Galapagos Islands, Ecuador) and a complex of beaches in Costa Rica that, although lesser in magnitude than Mexico and Galapagos, bears mention due to the apparently large numbers of green turtles that nest each year (M. Heidermeyer, Univ. Costa Rica, pers. comm. 2013). Galapagos nesters from the East Pacific

DPS showed multiple behavior patterns, including migration to Central American foraging areas, resident foraging areas within the Galapagos, and open ocean foraging areas where they foraged on soft-bodied invertebrates and surface dwelling prey that aggregate in frontal zones (Seminoff et al. 2008). Results of satellite telemetry work (Seminoff et al. 2008) with at-sea observations (IATTC 2012) indicate that many East Pacific green turtles live their lives in the high-seas of the Eastern Pacific likely because food is abundant in surface waters where currents converge and frontal zones exist.

Green turtles nest on sandy, ocean-facing mainland and island beaches (Hirth 1997). Although specific characteristics vary between rookeries, green turtle nesting beaches tend to have intact dune structures and native vegetation (Ackerman 1997). Sea turtle eggs require a high humidity substrate that allows for sufficient gas exchange and temperatures conducive to embryo development (Miller et al. 1997, 2003). Egg development is directly affected by temperature in the surrounding environment (Sato et al. 1998). For East Pacific sea turtles, Standora et al. (1982) reported water temperatures at Tortuguero between 27.5 and 28.5 ºC and Hays et al. (2002b) reported water temperatures at Ascension Island close to 28ºC. Blanco (2010) reported that mean water temperatures varied between 27.5 and 27.9 ºC off of her Costa Rica study sites. The similarity in the water temperatures of the three areas indicated that the shorter internesting interval of the East Pacific green turtles in Costa Rica was likely not a result of higher temperatures. Blanco (2010) postulated that the difference in the duration of internesting interval may be related to differences in size of the turtles and number of eggs per clutch (Bjorndal and Carr 1989; Broderick et al. 2003; Wallace et al. 2006).

During a single reproductive season turtles nest repeatedly with variable internesting times between consecutive clutches (Miller 1997). The internesting period is the time that it takes for the turtle to develop the next clutch, thus it is a direct representation of the ovulation, fertilization and shelling of the following clutch (Rostal 2007). Ovulation, fertilization and albumin deposition occur within the first 3 days after nesting (Wibbels et al. 1992) and in the remaining time the eggs are shelled in the oviduct. In general, during the internesting period sea turtles congregate in areas along the coast relatively close to the nesting beaches, moving back to the nesting beaches just a few days prior to the nesting event (Fossette et al. 2007; Schofield et al. 2009; Shillinger et al. 2010). The behavior of female turtles during internesting is driven by energy optimization due to the high cost of crawling onto the beach and laying eggs (Wallace et al. 2005); as a result, they spend most of the time resting on the seabed (Hays et al. 1999).

Blanco (2010) reported that female East Pacific green turtles spent the internesting period close to nesting beaches in northwestern Costa Rica and used the complete water column, which was very shallow. Males were also present in the waters off the nesting beaches and were often observed mating in the waters off the beach. Therefore, several parts of the life cycle of the Costa Rica population were occurring in a small area, highlighting the importance of these areas for conservation. Satellite tracking data collected by Blanco (2010) off Costa Rica indicated that the areas of importance during the internesting period of green turtles that nested on Nombre de Jesús and Zapotillal beaches were the waters off the nesting beaches in nearby Brasilito Bay (4 km north of the nesting beach). Turtles mainly remained in a particularly small area near the nesting beaches. Contrary to Blanco’s (2010) findings, green turtles on the Caribbean coast of Costa Rica travel from 30 to approximately 100 km off shore during internesting (Tröeng et al. 2005). Blanco’s (2010) results differ from the behavior described for leatherback turtles that nest at nearby Parque Nacional Marino Las Baulas (PNMB) that travel hundreds of kilometers during the reproductive season (Shillinger et al. 2010). The mean daily distance traveled by the green turtle was 4.6 km. There was little variation between turtles, and these differences apparently were not related to turtle sizes as was reported for leatherback turtles (Eckert 2002). In contrast to overall movements, distances covered in a day for these green turtles were similar to the daily distance traveled by leatherback turtles at PNMB (Shillinger et al. 2010).

Because of the temperate nature of many green turtle foraging areas at the northern and southern extents of their range in the East Pacific, green turtles may experience colder waters in this region than anyplace else in the world. In northwestern Mexico and California, green turtles become inactive during the cold months of December to March (Seminoff 2000). During this period, green turtles may enter a torpid state during which they may lay motionless on the sea floor for days to weeks. This behavior is poorly understood, although green turtle overwintering is the focus of increasing study and has also been documented in the Mediterranean and Gulf of Mexico (Broderick et al. 2007; Hochscheid et al. 2007). A common behavioral trait that appears to characterize green turtles in the Galapagos Islands, Hawaii, and Australia is basking, where turtles haul out on beaches or sand dunes during the daytime to apparently warm in the sunlight. This behavior is rare in other parts of the world and possibly is an adaptive response to the cooler thermal environment and ocean predation pressure in those regions (Whittow and Balazs 1982; Green 1998; Limpus 2008).

Status

The green turtle was 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 turtle is composed of 11 DPSs that qualify as a “species” for listing. The Services removed the current range-wide listing and, in its place, listed eight DPSs as threatened and three as endangered.

Although males and juvenile life stages are critical to the survival and recovery of the species, the population dynamics of green turtles and all of the other sea turtles we consider in this opinion are described based on the distribution and habit of nesting females, rather than their male and juvenile life stage counterparts. The spatial structure of male sea turtles and their fidelity to specific coastal areas is unknown; however, we describe 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 the number of nests laid per year.

Nesting abundance was ranked with a low risk of extinction. There were three primary regions considered under the critical assessment element of absolute abundance, with Mexico having the largest number of nesting females (13,664 nesters among seven nesting sites), followed by Ecuador (3,603 females in the Galapagos, 15 on mainland), and Costa Rica (2,826 females distributed among 26 nesting sites; see Table 15.1 in Seminoff et al. 2015.

Although trend information is lacking for the vast majority of sites, based on 25-year trend line for Michoacán, Mexico—the largest nesting aggregation in the East Pacific DPS—it is clear that green turtle nesting has 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. In addition to Mexico, data from the Galapagos Archipelago does not suggest a declining trend, and the largest-ever nesting numbers reported in Costa Rica suggest this site may be on the increase as well.

The SRT examined four lines of evidence relating to spatial structure of the population, including genetic data, flipper and satellite tagging, and demographic data. The genetic data indicate that there are regional genetic stocks, including Revillagigedos Archipelago (Mexico), Michoacán (Mexico), Central America (Costa Rica) and the Galapagos Islands. To a lesser extent, there was also a level of substructure evident in flipper tag returns, with a clear separation between the northern nesting beaches in Mexico, where most returns occurred north of El Salvador, and the nesting beaches in the Galapagos, where the vast majority of tag returns came from Nicaragua south to Peru. There is a relative paucity of satellite tracking data for green turtles in the East Pacific DPS, both those tracks available for our examination similarly depict separation between northern and southern portions of this DPS. With respect to demographics, the data available for the nesting sites within this DPS (Michoacán, Revillagigedos Islands, and Galapagos) indicate that these relatively well-studied rookeries differ substantially in key demographic parameters such as mean nesting size, hatching success, and nest size.

Green turtles from this DPS are listed as threatened under the ESA. An estimated 3,319–3,479 Eastern Pacific females nested annually (NMFS and FWS 2007b), and nesting has been steadily increasing at the primary nesting sites in Michoacán, Mexico, and in the Galapagos Islands since the 1990s (Delgado and Nichols 2005; Senko et al. 2011). Nesting trends at Colola have continued to increase since 2000 with the overall Eastern Pacific green turtle population also increasing at other nesting beaches in the Galapagos and Costa Rica (Wallace et al. 2010, NMFS and FWS 2007b). Based on nesting beach data, the current adult female nester population for Colola, Michoacán is 11,588 females, which makes this the largest nesting aggregation in the East Pacific DPS, comprising nearly 58% of the total adult female population. The total for the entire Eastern Pacific DPS is estimated at 20,112 nesting females (Seminoff et al. 2015).

Protection of green turtles is provided by local marine reserves throughout the region. In addition, sea turtles may benefit from the following broader regional efforts:

1) The Eastern Tropical Pacific Marine Corridor Initiative supported by the governments of Costa Rica, Panama, Colombia, and Ecuador, which is a voluntary agreement to work towards sustainable use and conservation of marine resources in these countries' waters;
2) the Eastern Tropical Pacific Seascape Program managed by Conservation International that supports cooperative marine management in the Eastern Tropical Pacific, including implementation of the Marine Corridor Initiative;
3) the Inter-American Tropical Tuna Commission and its bycatch reduction efforts that are among the world's finest for regional fisheries management organizations;
4) the Inter-American Convention for the Protection and Conservation of Sea Turtles, which is designed to lessen impacts on sea turtles from fisheries and other human impacts; and
5) the Permanent Commission of the South Pacific (Lima convention), which has developed an Action Plan for Sea Turtles in the Southeast Pacific.

There are indications that wildlife enforcement branches of local and national governments are stepping up their efforts to enforce existing laws, although successes in stemming sea turtle exploitation through legal channels are few and far between.

The following countries have laws to protect green turtles: Chile, Colombia, Costa Rica, Ecuador, El Salvador, Guatemala, Honduras, Mexico, Nicaragua, Panama, Peru, and the U.S.. In addition, at least 10 international treaties and/or regulatory mechanisms apply to the conservation of green turtles in the East Pacific DPS.

Since 1996, all countries from Mexico to Ecuador declared the use of TEDs as mandatory for all industrial trawling fleets to meet the requirements to export shrimp to the U.S. under the U.S. Magnuson- Stevens Fishery Conservation and Management Act (Helvey and Fahy 2012).

In 2008, the Western and Central Pacific Fisheries Commission issued a Conservation and Management Measure (2008–03; https://www.wcpfc.int/doc/cmm-2008- 03/conservation-and-management-sea- turtles) to reduce sea turtle mortality during fishing operations, collect and report information on fisheries interactions with turtles, and encourage safe handling and resuscitation of turtles. This measure requires purse seine vessels to avoid encircling turtles and to release entangled turtles. It also requires longline vessels to use line cutters and de-hookers to release turtles.

Threats to the Species

Natural Threats

Eastern Pacific green turtles are exposed to a variety of natural threats both at their nesting beaches and in the open ocean. The beaches where turtles nest and the nest themselves are threatened by hurricanes, tropical storms, storm surges, sand accretion, and rainfall associated with storms. Predation occurs at low levels in the East Pacific DPS. In the Galapagos Islands there is depredation on eggs and hatchlings by feral pigs (*Sus* spp.) and beetles (order Coleoptera) (Zarate et al. 2013). Predation levels are not reported (Zarate et al. 2003, 2006). There are accounts of jaguars (*Panthera onca*) killing adult female green turtles (L. Fonseca, National University of Costa Rica, unpubl. data 2009) at beaches in Costa Rica, but this is not a major problem for the DPS.

As discussed above, Fibropapillomatosis is the most commonly identified disease in green turtles and is characterized by the presence of internal and/or external tumors (fibropapillomas) that may grow large enough to hamper swimming, vision, feeding, and potential escape from predators (Herbst 1994). However, the extent to which this is a threat to the population is unknown. Disease, specifically fibropapillomatosis, was not a factor that contributed to the historical decline of this DPS.

The best available data suggest that fibropapillomatosis does not pose a current threat to the persistence of this DPS. Fibropapillomatosis is virtually non-existent in green turtles within the East Pacific DPS (Koch et al. 2007), although a variant of fibropapillomatosis has been found in one green turtle from San Diego Bay, USA (Greenblatt et al. 2005) that shared DNA affinities with the Mexican green turtle stock (P. Dutton, NMFS, unpubl. data). In addition, a few other turtles in San Diego Bay were believed to have the precursor to fibropapillomatosis based on eye anomalies (McDonald and Dutton 1990).

Anthropogenic Threats

The largest threat on nesting beaches to the East Pacific green turtle DPS 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 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.

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

The fishing industry in Peru is the second largest economic activity in the country. Over the last few decades, evidence of marine vertebrate bycatch has been collected for a range of industrial fisheries. It has recently been acknowledged that large impacts may also result from similar interactions with small-scale fisheries (SSF) due largely to their diffuse effort and large number of vessels in operation. From 2000 to 2007, shore-based and onboard observer programs from three SSF ports in Peru were used to assess the impact on marine turtles of small-scale longline, bottom set nets and driftnet fisheries (Alfaro- Shigueto et al. 2011). From this study, a total of 807 sea turtles were captured, 91.8% of which were released alive. It was estimated that 5,900 turtles were captured annually (3,200 loggerhead turtles, 2,400 green turtles, 240 olive ridleys and 70 leatherback turtles). SSFs in Peru are widespread and numerous (>100 ports, >9500 vessels, >37 000 fishers), and the observed effort in this study constituted 1% of longline and net deployments. Alfaro-Shigueto et al. (2011) suggest that the number of turtles captured per year is likely to be in the tens of thousands. Thus, the impacts of Peruvian SSF have the potential to severely impact sea turtles in the Pacific especially green, loggerhead, and leatherback turtles.

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

The HI SSLL fishery interacted with a total of ten green turtles (one estimated mortality) between 2004 and June 30, 2018 (NMFS 2018a). Nineteen turtles have been sampled since the mid-1990s and based on the genetics, NMFS estimates that between 32-77% of the turtles caught in the shallow-set longline fishery could be from the East Pacific DPS (P. Dutton pers. comm. November 20, 2017). This means that 3-8 turtles have been captured from this DPS by the shallow-set longline fishery since 2004. The HI DSLL fishery interacted with approximately 67 green turtles (62 estimated mortalities) between 2004 and June 30, 2018 (NMFS 2018b). Thirty-one of these turtles were sampled and based on the genetics, NMFS estimates that between 31-80% of the turtles caught in the HI DSLL fishery could be from the East Pacific DPS (P. Dutton pers. comm. June 29, 2018). This means that 21-54 turtles were captured from this DPS by the HI DSLL fishery since 2004. The American Samoa longline fishery interacted with approximately 270 green turtles (248 estimated mortalities) between 2006 and June 30, 2018 (NMFS 2018c). Thirty-one of these turtles were sampled and based on the genetics, NMFS estimates that between 2.0-23.2% of the turtles caught in the American Samoa longline fishery could be from the East Pacific DPS (P. Dutton pers. comm. July 5, 2018). This means that between 5-63 turtles were captured from this DPS by the American Samoa longline fishery since 2006.

In some countries and localities within the range of the East Pacific DPS, harvest of green turtle eggs is legal, while in others it is illegal but persistent due to lack of enforcement. The impact of egg harvest is exacerbated by the high monetary value of eggs, consistent market demand, and severe poverty in many of the countries in the Eastern Pacific Region. Egg harvest is a major conservation challenge at several sites in Costa Rica, including Nombre de Jesus and Zapotillal Beaches, where 90% of the eggs were taken by egg collectors during one particular study (Blanco 2010). Egg harvest is also expected to occur at unprotected nesting sites in Mexico, Guatemala, El Salvador, and Nicaragua (NMFS and FWS 2007b). 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.

Effects of climate change include, among other things, sea surface temperature increases, the alteration of thermal sand characteristics of beaches (from warming temperatures), which could result in the reduction or cessation of male hatchling production (Hawkes et al. 2009; Poloczanska et al. 2009), and a significant rise in sea level, which could significantly restrict green turtle nesting habitat. While sea turtles have survived past eras that have included significant temperature fluctuations, future climate change is expected to happen at unprecedented rates, and if sea turtles cannot adapt quickly they may face local to widespread extirpations (Hawkes et al. 2009). Impacts from global climate change induced by human activities are likely to become more apparent in future years (IPCC 2007). However, at the primary nesting beach in Michoacán, Mexico (Colola), the beach slope aspect is extremely steep and the dune surface at which the vast majority of nests are laid is well-elevated. This site is likely buffered against short-term sea level rise 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.

4.2.3.3 Central South Pacific DPS

Differential Distribution

The Central South Pacific DPS extends north from northern New Zealand to Fiji, Tuvalu, and Kiribati and east to include French Polynesia. 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. This DPS includes a longitudinal expanse of 7,500 km—from Easter Island, Chile in the east to Fiji in the west, and encompasses American Samoa, French Polynesia, Cook Islands, Fiji, Kiribati, Tokelau, Tonga, and Tuvalu.

Adult Foraging and Diving

See *Adult Foraging and Diving* above for general information on green turtle feeding and diving behavior.

Green turtles departing nesting grounds in this DPS travel throughout the South Pacific Ocean. Post-nesting green turtles tagged in the early 1990s from Rose Atoll returned to foraging grounds in Fiji and French Polynesia (Craig et al. 2004). Nesters tagged in French Polynesia migrated west after nesting to various sites in the western South Pacific (Tuato'o-Bartley et al. 1993). In addition to nesting beaches, green turtles are found in coastal waters (White 2013; White and Galbraith 2013), but in-water information in this population is particularly limited. Foraging areas are mostly coral reef ecosystems, with seagrass beds in Tonga and Fiji being a notable exception.

Craig et al. (2004) tracked migrations of seven post-nesting green turtles at Rose Atoll (American Samoa) in 1993–1995 and reported that most turtles migrated 1600 km to foraging areas in Fiji and occupied home ranges averaging 27 km^2. In total, 26 recaptures of primarily post-nesting turtles from French Polynesia, American Samoa, and Cook Islands showed a similar course of direction and destination: 96% migrated westward after nesting, with 58% going specifically to Fiji. Craig et al. (2003) proposed that this pattern reflects the lower availability of turtle food east of Fiji where most islands are small, steep and have limited areas suitable for seagrass or algal growth. In contrast, Fiji's extensive pastures of seagrass and algae appear to be a significant resource for many green turtles in the region. These turtles apparently spend most of their adult life in Fijian waters, taking only brief migrations to other islands to nest.

Population Dynamics

Structure

The primary demographic features of green turtles that are relevant for interpreting population structure and long term trends include age-to-maturity (often via growth studies), reproductive longevity, sex ratio, reproductive output (i.e. egg production, clutch frequency, hatching success, internesting interval), and annual survivorship.

Green turtle nesting in the Central South Pacific DPS is geographically widespread at low levels. The most abundant nesting area is Scilly Atoll, French Polynesia, which in the early 1990s hosted 300–400 nesters annually (Balazs et al. 1995). The current estimate is 1,050 breeding females (Seminoff et al. 2015). The most recent information is for American Samoa, with the majority of nesting at Rose Atoll and sporadic nesting on Tutuila and Swains Islands; sub-adult and adult turtles occur in low abundance in nearshore waters around Tutuila, Ofu, Olosega, Ta'u, and Swains islands (NMFS and USFWS 1998; Maison et al. 2010). Historically, 100–500 females nested annually at Canton Island, Kiribati (Balazs 1975a). Historical baseline nesting information in general is not widely available in this region, but exploitation and trade of green turtles throughout the region is well-known (Groombridge and Luxmoore 1989).

Flipper tag returns and satellite tracking studies demonstrate that post-nesting females travel the complete geographic breadth of this DPS, from French Polynesia in the east to Fiji in the west, and sometimes even slightly beyond (Tuato'o-Bartley et al. 1993; Craig et al. 2004; Maison et al. 2010; White, 2012a), as far as the Philippines (Trevor 2009). The complete extent of migratory movements is unknown.

Based on available data, Seminoff et al. (2015) estimated there are nearly 3,000 nesters in this DPS. However, the largest nesting site, Scilly Atoll, which comprises roughly one third of the entire nesting abundance, was last monitored in the early 1990's (Balazs et al. 1995) and has reportedly significantly declined in the past 30 years as a result of commercial exploitation (Conservation International Pacific Islands Program 2013). No sites have long-term monitoring programs, and no single site has had standardized surveys for even 5 continuous years. Most nesting areas are in remote, low-lying atolls that are logistically difficult to access. Unsurprisingly, many nesting areas (21 of 59, or 36%) only have qualitative information that nesting is present, indicating that there is still much to learn about green turtle nesting in this region. As these unquantified rookeries most likely each have a female abundance in the 1–100 range, their collective sum is probably fewer than 700 nesters. When added to our 2,902 total, this DPS likely has fewer than 3,600 nesters. Green turtle temporal population trends in the Central South Pacific DPS are poorly understood, with not even a single nesting site having five contiguous years of standardized monitoring that span entire nesting seasons. Partial and inconsistent monitoring from the largest nesting site in this DPS, Scilly Atoll, suggests significant nesting declines from persistent and illegal commercial harvesting (Petit 2013). Nesting abundance is reported to be stable to increasing at Rose Atoll, Swains Atoll, Tetiaroa, Tikehau, and Maiao. However, these sites are of moderate to low abundance and in sum represent less than 16% of the population abundance at Scilly Atoll alone. Nesting abundance is reported to be stable to increasing at Tongareva Atoll (White and Galbraith 2013). The uncertainty surrounding the above trends, and the general dearth of long-term monitoring and data from this DPS, presents significant challenges to any formal quantitative trend analyses.

Genetic sampling in the Central South Pacific DPS has been limited and many of the small isolated nesting sites that characterize this region have not been covered. Genetic sampling is currently underway at Tongareva Atoll, Cook Islands (M. White, unpubl. data). Based on limited sampling, there is evidence of significant spatial structuring. Within the DPS, there is significant population substructuring (F_{st}[4] 0.53, $p<0.005$) between American Samoa and French Polynesia (P. Dutton, NMFS unpubl. data). The samples from American Samoa were collected across four locations (Swains Island, Tutuila, Ofu and Rose Atoll) that had both low sample sizes ($n = 1–8$) and were a great distance from each other (160–500 km). However, these were pooled to

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[4] The FST is a measure of population differentiation as a result of genetic structure.

represent American Samoa as they shared haplotypes and were significantly distinct from French Polynesia, represented by one sampled nesting site (n = 9) at Mopelia (P. Dutton, NMFS unpubl. data). Flipper tag returns and satellite tracking studies demonstrate that post-nesting females travel the complete geographic breadth of this DPS, from French Polynesia in the east to Fiji in the west, and sometimes even slightly beyond (Tuato'o-Bartley et al. 1993; Craig et al. 2004; Maison et al. 2010; White 2012a), as far as the Philippines (Trevor 2009). The complete extent of migrations is unknown.

Demographic studies do not reveal any structuring of traits within this DPS. Limited demographic information is available for green turtles in the Central South Pacific DPS. Nesters at Scilly Atoll, French Polynesia in one study of five females averaged 95.6 cm CCL (Hirth 1980) and in another study of 51 females averaged 103 cm SCL (Balazs et al. 1995). Nesters at Rose Atoll, American Samoa averaged 94.7 cm CCL (n=68) (Van Houtan, NMFS unpubl. data 2013). Five nesters in Tokelau ranged from 102–104 cm CCL (Balazs 1983) and had a 14-day interval between clutches. Peak nesting occurs from August to November at Rose Atoll (Craig et al. 2004), occurs in November in American Samoa (Tuato'o-Bartley et al. 1993), occurs in January to February at Pitcairn Island (Brooke 1995), and occurs from June to December in Tokelau (Balazs 1983). Demographic information from nest-level inventories is not available for this population (Seminoff et al. 2015).

Diversity

Diversity is a critical element considered under the status review (Seminoff et al. 2015) for green turtles and includes consideration of the overall nesting spatial range, diversity in nesting season, diversity of nesting site structure and orientation, and the genetic diversity within the DPS.

The Central South Pacific has a broad geographical area, but the nesting sites themselves exhibit little diversity. Most nesting sites are located in low-lying coral atolls or oceanic islands as the region has no true continental land mass. Local nesting density is sparse spatially, typically spread over > 10 km stretches of beach and is also low in terms of abundance. Only one nesting site (Scilly Atoll with 1,050 females) has a nester abundance exceeding 250. Foraging areas are mostly coral reef ecosystems, with seagrass beds in Tonga and Fiji being a notable exception. In summary, most nesting sites in this DPS are in remote low-lying atolls, have low abundance, and nesting is at low spatial densities. Mitochondrial DNA studies based on very limited sampling indicate there are at least two genetic stocks in the Central South Pacific DPS, with a moderate level of diversity and presence of unique haplotypes (P. Dutton, NMFS unpubl. data).

Nesting occurs sporadically throughout the geographic distribution of the population, with isolated locations having relatively low to moderate nesting activity (Figure 22). Green turtles departing nesting grounds in this DPS travel throughout the South Pacific Ocean. Post-nesting green turtles tagged in the early 1990s from Rose Atoll returned to foraging grounds in Fiji and French Polynesia (Craig et al. 2004). Nesters tagged in French Polynesia migrated west after nesting to various sites in the western South Pacific (Tuato'o-Bartley et al. 1993). In addition to nesting beaches, green turtles are found in coastal waters (White 2013; White and Galbraith 2013), but in-water information in this population is particularly limited.

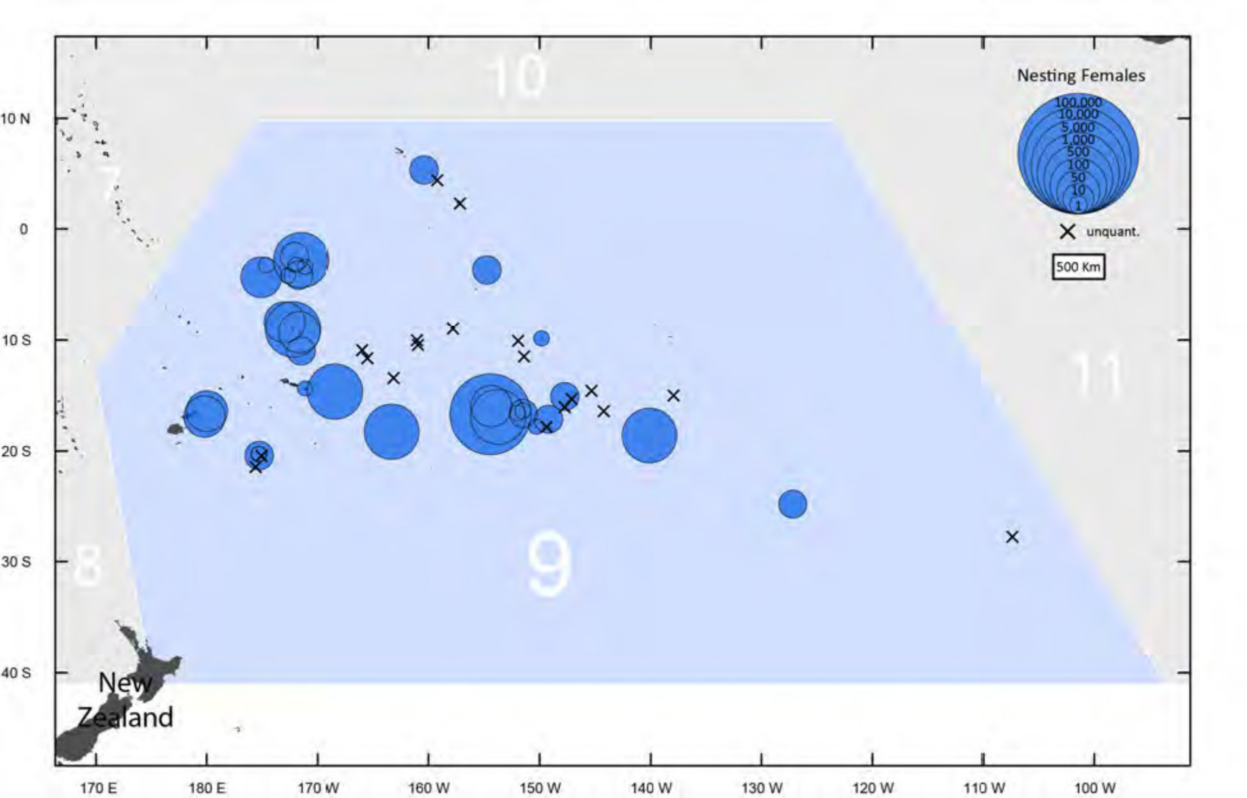

Figure 22. Nesting distribution of green turtles in the Central South Pacific DPS. Size of circles indicates estimated nester abundance (see Section 13.2.1). Locations marked with 'X·' indicate nesting sites lacking abundance information.

Status

Green turtle temporal population trends in the Central South Pacific DPS are poorly understood. As previously discussed, no nesting sites have standardized monitoring that span entire nesting seasons. However, some monitoring data from the largest nesting site (Scilly Atoll) in this DPS, suggests significant nesting declines occur from persistent and illegal commercial harvesting (Petit 2013). Conversely, nesting abundance is reported to be stable to increasing at Rose and Swains Atoll, Tetiaroa, Tikehau, and Maiao. However, these sites are of moderate to low abundance and in sum represent less than 16% of the population abundance at Scilly Atoll alone. Nesting abundance is reported to be stable to increasing at Tongareva Atoll (White and Galbraith 2013). The uncertainty surrounding the above trends, and the lack of long-term monitoring and data from this DPS, presents significant challenges to any trend analyses (Seminoff et al. 2015).

Threats to the Species

Natural Threats

We do not know the extent and level of threat from disease for the South Central Pacific DPS. Depredation may have been a factor that contributed to the historical decline of this DPS. The best available data suggest that current nest and hatchling predation on several Central South Pacific DPS nesting beaches and in water habitats is a potential threat to this DPS. Identified predators that may constitute a terrestrial threat to turtles include hermit crabs, ghost crabs, Polynesian rats, frigate birds (*Fregata ariel, F. minor*), and reef herons (*Egretta sacra;* Balazs, 1983). In the marine environment, sharks and other carnivorous fish (e.g., groupers) prey on sea turtles.

Anthropogenic Threats

Directed take in the marine environment has been a significant source of mortality in American Samoa, and turtle populations have seriously declined (Tuato'o-Bartley et al. 1993; NMFS and USFWS 1998), even though the take is illegal under the ESA. Grant et al. (1997) reported that even though there had been educational efforts relating to sea turtles, some turtles and eggs were still illegally taken. NMFS and USFWS (1998) noted directed take as a significant source of mortality in American Samoa. The extent of current illegal take may be less than in the past (A. Tagarino, American Samoa DMWR, pers. comm., 2013 as cited in Seminoff et al. 2015); however, actual levels are unknown. Turtles from American Samoa migrate to other countries (e.g., Fiji, Samoa, and French Polynesia) where turtle consumption is legal or occurs illegally. For example, there is a documented instance of two turtles which were tagged at Rose Atoll being captured and eaten in Fiji (Craig 1993; Tuato'o-Bartley et al. 1993). This illustrates the complexity of threats affecting green turtles of the Central South Pacific DPS.

Animals are protected in some countries, but when they migrate to other countries they face the threat of harvest. Turtles have been traditionally harvested for food and shells in the country of Samoa and have cultural significance (Craig 1993). Overexploitation of turtles has negatively affected local populations (Government of Samoa 1998). Unsustainable harvest (direct take for meat) remains a major threat to green turtles in Samoa (J. Ward, Government of Samoa, pers. comm. 2013 from Seminoff at el. 2015). In Fiji, Weaver (1996) identified the contemporary harvest and consumption of turtles by humans for eggs, meat, and shells as a significant threat for sea turtles. Commercial harvest (a major threat), as well as subsistence and ceremonial harvest, are all contributing factors. Rupeni et al. (2002) report that green turtles are threatened from traditional harvesting for ceremonial purposes, as well as from subsistence and commercial harvesting for meat, eggs, and shell (turtles captured for general consumption and sale in local markets). Although a thorough assessment of these threats is not available, Batibasaga et al. (2006) note that sea turtles in Fiji have been substantially overfished since the 1980s, and report that an estimated 400–500 green turtles are killed in Fiji each year. Laveti and MacKay (2009) found that open sale of turtles in markets no longer occurs, but report that anecdotal information suggests substantial catch of turtles for subsistence, traditional use, and possible black market commercial sales. Their market research on the island of Viti Levu from April 2006 to 2007 found 29 green turtle carapaces (average price of US $42). Illegal harvest of sea turtles by villages in Fiji for household consumption still occurs, and the rules that allow traditional take are poorly understood, with low compliance (Laveti and MacKay 2009). Jit (2007) notes that the green turtle nesting beaches of Heemskereq Reefs and Ringgold Isles are vulnerable to illegal harvest by fishing vessels.

In Kiribati (e.g., Phoenix Islands), an unknown number of turtles are caught as bycatch on longlines and eaten (Obura and Stone 2002). Poaching has been reported for Caroline Atoll, but to what extent it currently occurs is unknown (Teeb'aki, 1992). In Tonga, Bell et al. (1994)

report that collection of eggs for subsistence occurs, and Prescott et al. (2004) and Havea and MacKay (2009), also note that it is still a practice on islands where turtles nest. Bell et al. (2009) report that in Tonga sea turtles are harvested and live turtles are often seen transported from outer islands to the main island, Tongatapu. In 2007, Havea and MacKay (2009) conducted a survey in the three islands of Ha'apai to determine how many turtles were captured. They found that fishermen captured 56 turtles on O'ua, 23 on Ha'afeva, and 119 on Tungua. It is not clear how representative these three villages are for Ha'apai (another 7 islands or villages in Ha'apai were identified as hunting turtles in 1972). It is likely that this number is the minimum number of turtles captured in the Ha'apai Group (Havea and MacKay, 2009). No other data was reported on turtle hunting for other islands. Turtles were primarily captured by diving (hand), spear, and net, and used for consumption at home, local sales or barter, traditional occasions, and in some cases for a commercial market in the main island of Tongatapu (Havea and MacKay 2009). It is unclear if this harvest is sustainable, especially given the increased catch rates in Tungua for the commercial market (Havea and MacKay 2009).

In Tuvalu, harvest of sea turtles for their meat has been cited as a major threat (Alefaio and Alefaio 2006). In Tokelau, Balazs (1983) reported human take of both sea turtle eggs from nests and males and females while copulating, while nesting, or by harpoon. Apparent reductions in sea turtle numbers brought into question the sustainability of harvest in Tokelau and elicited discussion regarding conservation measures for the sea turtle population (Balazs 1983). However, it appears sea turtles are still consumed in Tokelau (Ono and Addison 2009). In the Cook Islands, turtles are sometimes killed during nesting at Palmerston and Rakahanga, while nesting and via fishing on Nassau, and while nesting at Manihiki and Tongareva, and probably at other atolls; the exact level of take overall is unclear (White 2012a). At Tongareva (2011-2014) four females were taken while nesting, two juveniles and one adult female by net, one sub-adult speared, and four sub-adults (3 females, 1 male) were taken in water by hand (White 2012; M. White unpubl. data). Turtles are occasionally speared underwater at Rakahanga (White and Galbraith, 2013). Only one clutch of eggs has been harvested at Tongareva Atoll between (2010-2014, M. White, unpubl. data). Take of turtles and eggs has been identified at Mauke, however the number taken is unknown (Bradshaw and Bradshaw 2012).

Incidental capture in artisanal and commercial fisheries is a significant threat to the survival of green turtles throughout the Central South Pacific DPS. The primary gear types involved in these interactions include longlines and nets. Incidental capture in line, trap, or net fisheries presents a threat to sea turtles in American Samoa (Tagarino 2011). Subsistence gill nets have been known to occasionally catch green turtles. Additionally, longline fishing is considered a threat to Central South Pacific green turtles. The American Samoa longline fishery is closely regulated and monitored, and has the most reliable data in the area for bycatch. NMFS estimates that between 30.7-72.9% of the turtles caught in the American Samoa longline fishery could be from the Central South Pacific DPS (P. Dutton pers. comm. July 5, 2018). This means that between 83-197 turtles were estimated to have been captured from this DPS by the American Samoa longline fishery since 2006. It is unclear exactly how many Central South Pacific green turtles in the South Pacific Ocean are taken in other longline fisheries, however it is estimated that over 200 green turtles could be killed annually by longline fishing in just the part of the South Pacific around American Samoa bounded by 180° and 155° W longitude, and 3° S – 32° S latitude (Maison et al. 2010).

In Fiji, green turtles are killed in commercial fishing nets, however the exact extent and intensity of this threat is unknown (Rupeni et al. 2002). Jit (2007) suggests that sea turtle bycatch is occurring in tuna fisheries in Fiji, but no information is provided on possible extent of sea turtle take or the species that are possibly taken. However, McCoy (2008) reports that green turtle bycatch is occurring in longline tuna fisheries in Fiji. Unfortunately, fishing trips do not appear to properly represent spatial and temporal distribution of fishing effort throughout the year, and the level of observer coverage is low, so the exact level of interactions with green turtles is unclear. The most recent data, provided by Fiji to the WCPFC (2016, 2018), reports 79 green sea turtles have been captured between 2011 and 2017. Of those 79 total turtles, 44 were deceased (56%; WCPFC 2016, 2018).

In the Cook Islands, longline fishery regulations require fishers to adopt the use of circle hooks and to follow "releasing hooked turtles" guidelines (Cook Islands Marine Resources Longline Fishery Regulations 2008), although it is unclear how effective these regulations are. McCoy (2008) suggests that sea turtle bycatch is occurring in tuna fisheries in the Cook Islands; however, no information is provided on possible extent of sea turtle take or the species that are possibly taken. This was further substantiated after reviewing the WCPFC annual reports provided by the Cook Islands. White (2012a) reports that Cook Islands territorial waters are fished by other countries; however, the extent of sea turtle bycatch has not been fully analyzed and is unclear.

NMFS estimates that between 0-5% of the turtles caught in the HI SSLL fishery could be from the Central South Pacific DPS (P. Dutton pers. comm. November 20, 2017). This means that one turtle may have been captured from this DPS by the shallow-set longline fishery since 2004. NMFS estimates that between 0-24% of the turtles caught in the HI DSLL fishery could be from the Central South Pacific DPS (P. Dutton pers. comm. June 29, 2018). This means that 0-16 turtles were estimated to have been captured from this DPS by the HI DSLL fishery since 2004.

4.2.3.4 Central West Pacific DPS

Differential Distribution

The Central West Pacific DPS has as its 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.

Adult Foraging and Diving

See *Adult Foraging and Diving* above for general information on green turtle feeding and diving behavior.

In many areas, green turtle foraging areas coincide with fishing zones and urban development in this DPS. In the CNMI, seagrass beds used by green turtles as foraging habitat have been identified on Saipan (Kolinski et al. 2001), Tinian (Kolinski et al. 2004), and Rota (Kolinski et al. 2006) Islands. Seagrasses around Tinian and Rota Islands are foraging areas as well as seagrass beds around Saipan. Guam's coral reefs are also presumably food sources for turtles

(NMFS and USFWS 1998). Coral reefs and seagrass beds off of Pohnpei, Yap, Chuuk, Kosrae and off the lagoon shoreline of the Kwajalein Atoll islands and Majuro Atoll are foraging habitat.

Population Dynamics

Structure

Demographic data availability is limited and somewhat variable for nesting sites in this DPS. Variability in parameters such as remigration interval, clutch size, hatching success, and clutch frequency is not separated out regionally within the DPS and, therefore, does not necessarily suggest a high level of population structuring. Hatching success varies widely from 44.1-73.8% for areas with available information (Suganuma 1985). Clutch size range varies widely from 59 to 139 eggs/nest (Palacios 2012a, 2012b). Clutch frequency ranges from 4 to 6 nests per season (Suganuma 1985). Remigration interval varies from 3 to 4 years by nesting site (Cruce, 2009). The known mean nester sizes range from 102 cm CCL in Palau to 104.5 cm CCL in Tinian, CNMI (Pultz et al. 1999).

Stock structure analysis indicated that rookeries separated by more than 1,000 km were significantly differentiated from each other (FST values from 0.06 – 0.9, $p<0.001$) while neighboring rookeries within 500 km showed no genetic differentiation. Dutton et al. (2014) suggest that there are at least seven independent stocks in the region based on mtDNA analyses.

Satellite telemetry shows that nesting females migrate to areas within and outside of the Central West Pacific DPS. For example, satellite tracks show animals moving from the Mariana Islands to the Philippines and Japan, and others moving from the Chichijima Islands of Ogasawara to the main islands of Japan (Japan Fisheries Resource Conservation Association 1999; Hatase et al. 2006). Green turtles have also been shown to move from the FSM to the Philippines and to the west (G. Balazs, NMFS, unpublished data; Kolinski et al. unpublished data.).

Diversity

Diversity is a critical element considered under the status review (Seminoff et al. 2015) for green turtles and includes consideration of the overall nesting spatial range, diversity in nesting season, diversity of nesting site structure and orientation (e.g., insular vs. continental nesting sites), and the genetic diversity within the DPS. Aspects such as these are important considerations for assessing the potential impact of catastrophic events such as storms, sea level rise, and disease.

The overall range of the DPS is relatively widespread. Green turtles in this DPS are found from the Marshall Islands in the east to Palau in the west, and from Japan in the north to the Solomon Islands in the south. Nesting occurs on various islands and atolls throughout the DPS, however at what appears to be low numbers (except for a few locations). Nesting information is limited for some areas, but occurs from November to August in Palau; from March through September in the FSM; and May to August in Ogasawara, Japan. Some animals are traveling outside the bounds of this DPS, into the East Indian/West Pacific DPS.

While nesting and foraging areas are not concentrated in one area and this provides a level of habitat use diversity and population resilience, the contribution of this characteristic to such diversity and resilience is reduced by the threats faced in each of the nesting and foraging areas.

Green turtle nesting occurs at least at low levels throughout the geographic distribution of the population, with isolated locations having high nesting activity (Figure 23). The highest numbers of females nesting in this DPS are located in Gielop and Iar Island, Ulithi Atoll, Yap, FSM (1,412); Chichijima (1,301) and Hahajima (394), Ogasawara, Japan; Bikar Atoll, Marshall Islands (300); and Merir Island, Palau (441) (NMFS and USFWS 1998; Bureau of Marine Resources 2005; Barr 2006; Palau Bureau of Marine Resources 2008; Maison et al. 2010; H. Suganuma, Everlasting Nature of Asia, pers. comm., 2012; J. Cruce, Ocean Society, pers. comm. 2013 from Seminoff et al. 2015). Green turtles departing nesting grounds in this DPS travel throughout the Western Pacific Ocean. Results of three post-nesting green turtles from Palau in 2006 showed they remained nearby or traveled to the Aru Islands in Indonesia – roughly 1,100 km away (Klain et al. 2007). Five postnesting green turtles leaving Erikub Atoll in the Marshall Islands in 2007 traveled to the Philippines, Kiribati, FSM, or remained in the Marshallese EEZ (Kabua et al. 2012). Turtles tagged in Yap (FSM) were recaptured in the Philippines, Marshall Islands, Papua New Guinea, Palau, and Yap (Palau BMR 2008; Cruce 2009). A turtle tagged on Gielop Island, Yap in 1991 was recaptured in Muroto Kochi prefecture, Japan in 1999 (Miyawaki et al. 2000). A nesting female tagged on Merir Island, 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 from Seminoff et al. 2015; Ogasawara Marine Station, Everlasting Nature of Asia. unpublished data). A turtle tagged in Japan was recorded nesting in Yap, FSM (Cruce 2009). In addition to nesting beaches, green turtles are found in coastal waters in low to moderate densities at foraging areas throughout the DPS. Aerial sea turtle surveys show that an in-water population exists around Guam (DAWR, 2011). In-water green turtle density in the Marianas Archipelago is low and mostly restricted to juveniles (Pultz et al. 1999; Kolinski et al. 2005, 2006; Palacios, 2012a). In-water information in this DPS overall is particularly limited.

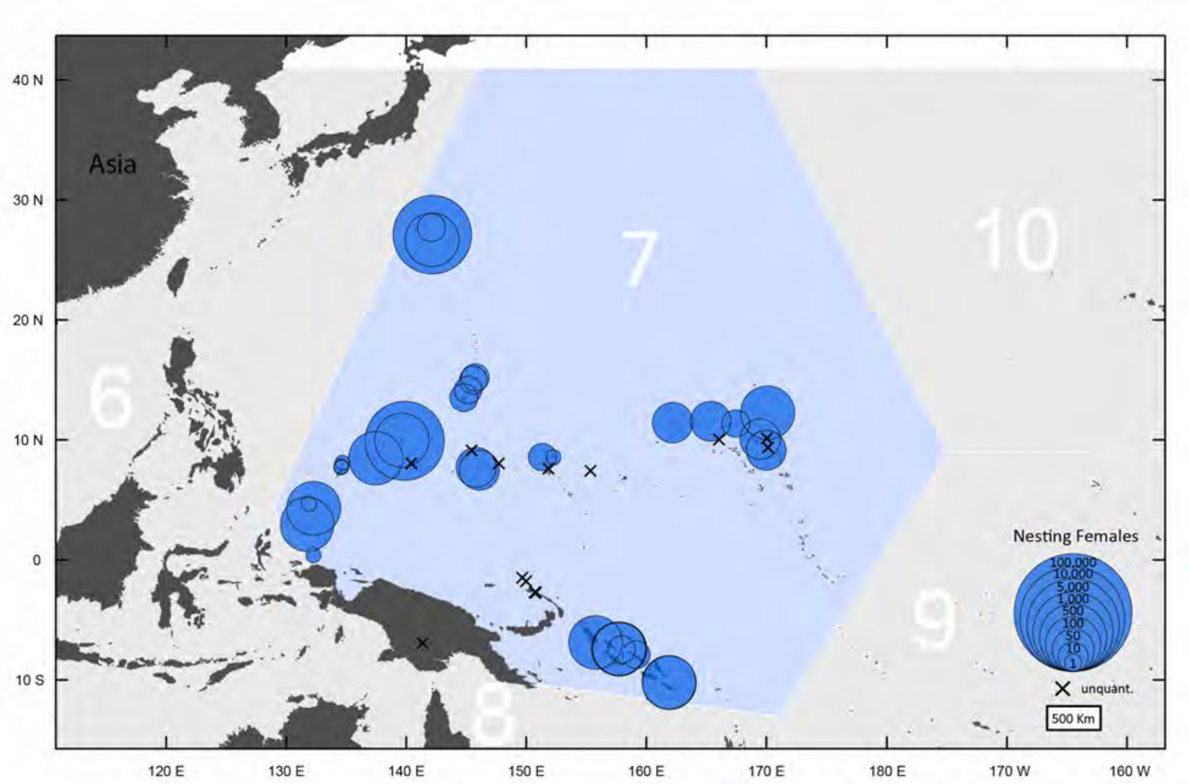

Figure 23**.** Nesting distribution of green turtles in the Central West Pacific DPS (blue-shaded region). Size of circles indicates estimate nester abundance.

Currently, there are approximately 51 nesting sites and 6,518 nesting females in the Central West Pacific (Seminoff et al. 2015). There are a number of unquantified nesting sites, possibly with small numbers, 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 for this DPS.

Status

There is insufficient long-term and standardized monitoring information to adequately describe abundance and population trends for many areas of the Central West Pacific DPS. The limited available information suggests a nesting population decrease in some portions of the DPS 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 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 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. (2008) reports an estimated annual population growth rate of 6.8% per year for the Chichijima nesting site.

Threats to the Species

Natural Threats

As discussed above, fibropapillomatosis is the most commonly identified disease in green turtles. In the FSM, disease has unknown impact. Twelve of 702 (1.7%) female green turtles tagged at Gielop Island between 1990 and 1993 had carapace lesions that were diagnosed as fibropapilloma (Kolinski 1994). Lesions of this type have also been reported on turtles foraging around Yap proper, as well as turtles in the Elato and Lamotrek regions (Kolinski, 1994). Cruce (2008) reported carapace lesions on four (5.8%) of 69 turtles encountered on Loosiep Island. She reported that the lesions were similar to those observed on Gielop Island during the 2005–2007 nesting seasons, the majority of which were suspected to be burrowing barnacle infestations and one was reported to be a papilloma.

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

Natural environmental events, such as cyclones and hurricanes, may affect green turtles in the Central West Pacific DPS. 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 Niño–Southern Oscillation, and currents (NMFS and USFWS 1998). Some erosion of nesting beaches at Oroluk was reported in 1990 after passage of Typhoon Owen (NMFS and USFWS 1998).

Anthropogenic Threats

Directed take 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; Guilbeaux 2001; Hitipeuw and Maturbongs 2002; Philip 2002; Humber et al. 2014). In addition to the collection of eggs from nesting beaches, the killing of nesting females continues to threaten the stability of green turtle populations. Ongoing harvest of nesting adults has been documented in the CNMI (Palacios 2012b), FSM (Cruce 2009), Guam (Cummings 2002), Papua (Hitipeuw and Maturbongs 2002), Papua New Guinea (Maison et al. 2010), and Republic of Palau (Guilbeaux 2001). The primary threat to turtles in the CNMI is human induced, with 79% of strandings attributed to directed take (Summers et al. in prep). Mortality of turtles in foraging habitats is also problematic for recovery efforts. Ongoing intentional capture of green turtles in their marine habitats has been documented in southern and eastern Papua New Guinea (Limpus et al. 2002) and the Solomon Islands (Broderick 1998; Pita and Broderick 2005). Sea turtles are considered a traditional delicacy for most ethnic groups in the CNMI, and turtles and eggs are readily taken on nesting beaches or in coastal waters (McCoy 1997; NMFS and USFWS 1998). Knowledge of existing regulations does not inhibit many people from eating turtles or their eggs. During March-August 2009, 16 green turtle nests (estimated to have been laid by five adult nesting females) were documented during intensive monitoring of seven beaches on Saipan; three (60%) of the five potential nesting turtles, as well as three of the nests, were illegally harvested (CNMI Division of Fish and Wildlife 2009), suggesting that poaching remains a significant threat to turtles on Saipan (Maison et al. 2010).

Incidental capture in artisanal and commercial fisheries is a threat to the survival of green turtles in the Central West Pacific. Sea turtles may be caught in longline, pole and line, and purse seine fisheries. In the Republic of the Marshall Islands, a purse-seine fishery for tuna and a significant longline fishery operate in the EEZ, and sea turtles have been captured in both fisheries with mortality sometimes occurring (Hay and Sablan-Zebedy 2005). McCoy (2007a) presented a summary of sea turtle interactions with longline vessels based in Majuro from observer data from 2005 to 2007. A total of 33 sea turtle interactions were documented during this period, of which six (18%) were identified as green turtles. The mortality rates recorded for these 33 interactions were high, with only five turtles identified as alive upon release (McCoy 2007a). Reviewing the data available in the WCPFC annual reports provided by the Marshall Islands reveals 5 green sea turtle interactions have occurred with their purse seine fisheries between 2010 and 2017, in which all turtles were released alive (WCPFC 2013, 2014, 2015, 2016, 2017, 2018).

In Palau, a total of 18 sea turtles were captured on shallow-set longline vessels during 12 trips with observer coverage from April–December 2007. Out of the 18 interactions, two (11%) were green turtles (McCoy 2007b). One was landed onboard alive and released, the other was dead at the time of landing. The catch per unit effort of the 18 interactions was 0.26 turtles per 1,000 hooks, with an average of 1,442 hooks deployed per 47 sets observed during the 12 trips. Taking into consideration that in February 2007, approximately 100 longline vessels were licensed to fish in the Palau EEZ, with about 50 to 80 actually actively engaged in the fishery in Palau, the potential for interactions with green turtles is relatively high. Recent Palau annual reports to the WCPFC reveal 10 turtles were caught from 2000-2014 in longline fisheries (WCPFC 2015b). However, only one permanent and three seasonal observers were available for coverage, resulting in a maximum coverage of 0.6% (WCPFC 2015b). One additional turtle was caught in 2016 and currently no observer coverage is available in the longline fleet with no reported species of special interest described for 2017 (WCPFC 2017b).

In the FSM EEZ and surrounding areas, an Oceanic Fisheries Programme (2001) review determined that 83 sea turtles were captured in 2,143 observed longline sets from 1990–2000 in an area described as the western tropical Pacific from 10°N to 10°S. McCoy (2003) estimated that the percentage of overall longline effort represented by these 2,143 observed sets was likely less than 2 to 5%. The condition of the 83 turtles captured in these sets was identified as 58% alive and healthy, 8% alive but injured or stressed, 6% barely alive, and 27% dead (Oceanic Fisheries Programme 2001). Although green and olive ridley turtles made up the majority of sea turtles that could be identified to the species level, a large number of the turtles encountered could not actually be identified, so the actual species composition of sea turtle interactions in the longline fisheries could not be determined.

In the Solomon Islands, domestic and foreign purse seine and pole and line fisheries, as well as a foreign longline fishery, participated in the commercial tuna fishery in 2007 (WCPFC 2008). In the CNMI, numerous subsistence and small scale commercial fishing operations occur along Saipan's western coast and along both the Rota and Tinian coasts (CNMI Coastal Resources

Management Office 2011). Incidental catch of turtles in Guam coastal waters by commercial fishing vessels probably also occurs (NMFS and USFWS 1998). However, no bycatch studies have been undertaken to quantify the level of incidental capture by commercial fishing operations in the Solomon Islands (Project GloBAL 2009e), the CNMI (Project GloBAL 2009b), or Guam (Project GloBAL 2009a). In 2007, 222 fishing vessels (200 purse-seiners and 22 longliners) had access to Papua New Guinea waters (Kumoru 2008). Although no official reports have been released on sea turtle bycatch within these fisheries (Project GloBAL 2009d), sea turtles interactions with both fisheries have been commonly observed (Kumoru 2008). However, the level of mortality is unknown.

NMFS estimates that between 0-8% of the turtles caught in the shallow-set longline fishery could be from the Central West Pacific DPS (P. Dutton pers. comm. November 20, 2017). This means that one turtle may have been captured from this DPS by the shallow-set longline fishery since 2004. NMFS estimates that between 0-18% of the turtles caught in the HI DSLL fishery could be from the Central West Pacific DPS (P. Dutton pers. comm. June 29, 2018). This means that 0-12 turtles were captured from this DPS by the HI DSLL fishery since 2004. NMFS estimates that between 0-24.7% of the turtles caught in the American Samoa longline fishery could be from the Central West Pacific DPS (P. Dutton pers. comm. July 5, 2018). This means that between 0-67 turtles were captured from this DPS by the American Samoa longline fishery.

4.2.3.5 Southwest Pacific DPS

Differential Distribution

The Southwest Pacific DPS 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.

Adult Foraging and Diving

See *Adult Foraging and Diving* above for the description of feeding and diving behavior.

Population Dynamics

Structure

For the Southwest Pacific DPS, Seminoff et al. (2015) identified 12 total nesting sites, although it should be noted that perhaps more so than in other DPSs, proximate nesting beaches were grouped. It would be possible to split the nesting aggregations into more than 100 different sites, but because many of the most recent estimates (Limpus 2009) are aggregated, we followed this tendency and aggregated nesting within broad regional areas. Nesting occurs at moderate to high levels within the Southwest Pacific DPS. Some isolated locations have extremely high nesting activity. The highest nesting assemblages in this DPS, and perhaps the entire species (Chaloupka et al. 2008), are located in the nGBR. Roughly 90% of the nesting activity here occurs at Raine Island and Moulter Cay, with appreciable nesting also occurring at Number Seven and Number Eight Sandbanks and Bramble Cay (Limpus 2009). Estimates of annual nesters at Raine Island vary from 4,000 – 89,000 (Seminoff et al. 2004; NMFS and USFWS 2007; Chaloupka et al.

2008; Limpus 2009). Female nesting abundance in the nGBR is not directly counted throughout the nesting season. This is largely because of the remoteness of the site and the sheer numbers of turtles that may nest on any given night, which makes accurate counting very difficult. A mark-recapture approach (Limpus et al. 2003) is used at Raine Island to estimate the number of adult female green turtles in the waters surrounding Raine Island during the sampling period. Females are painted during nightly tally counts, and then marked and unmarked adult female turtles are counted in the surrounding internesting habitats the following day using a structured survey protocol.

Jensen et al. (2018) combined genetic markers and a mixed stock analysis (MSA) with sex determined through laparoscopy and endocrinology, to link male and female green turtles foraging in the Great Barrier Reef (GBR) to the nesting beach from which they hatched. Results show a moderate female sex bias (65%–69% female) in turtles originating from the cooler southern GBR nesting beaches, while turtles originating from warmer northern GBR 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). They concluded that the northern GBR green turtle rookeries have been producing primarily females for more than two decades and that the complete feminization of this population is possible in the near future. This information suggests the population will likely crash due to the lack of males without management intervention.

The number of turtles nesting in the GBR area of Australia differs widely from year to year and is well correlated with an index of the Southern Oscillation (Limpus and Nicholls 2000). For example, the estimate of annual nesters at Raine Island during a medium density nesting season is about 25,000 (Limpus 2009), while in a high density season (1999–2000) the estimate of nesters at Raine Island increases to 78,672 ± 10,586. Heron Island is the index nesting beach for the sGBR, and nearly every nesting female on Heron Island has been tagged since 1974 (Limpus and Nicholls 2000). The mean annual nester abundance varied between 26 and 1,801 during 1999–2004 (Limpus 2009).

In comparison to Australia, fewer turtles nest in New Caledonia and Vanuatu. In New Caledonia, Pritchard (1994, cited in Maison et al. 2010) described turtles to be abundant or near saturation levels on the following islands: Surprise, LeLeixour, Fabre, and Huon. A 2006 and 2007 survey of over 6,000 km of nesting habitat identified nesting locations hosting an estimated 1,000 – 2,000 green turtles females nesting annually (Maison et al. 2010 citing Limpus et al. 2009). In Vanuatu, hundreds of nesting green turtles have been observed on Malekula Island, Southern Epi Island, Santo and Thion Islands, Tegua and Hiu Islands (Maison et al. 2010).

Nesting occurs in many islands throughout the Southwest Pacific DPS, but there are only two nesting areas (Raine Island and Heron Island, described in more detail below) with long-term (>15 years) annual indices of nesting abundance.

Genetic sampling in the Southwest Pacific DPS has been extensive for larger nesting sites along the GBR, the Coral Sea, and New Caledonia; however, there are several smaller nesting sites in this region that still need to be sampled (e.g. Solomon Islands, Vanuatu, Tuvalu, and Papua New Guinea). Within this DPS there is significant population substructuring (FST= 0.09–0.79, $p<0.05$). Of the ten nesting sites studied, four regional genetic stocks have been identified in the Southwest Pacific Ocean; nGBR, southern Great Barrier Reef (sGBR), Coral Sea (Dethmers et al. 2006; Jensen 2010), and New Caledonia (Dethmers et al. 2006; P. Dutton NMFS, unpubl. data). Mixed stock analysis of foraging grounds show that green turtles from multiple nesting beach origins commonly mix in foraging grounds along the GBR and Torres Strait regions (Jensen 2010), but with the vast majority originating from nesting sites within the GBR. There is evidence of low frequency contribution from nesting sites outside the DPS at some foraging areas.

Nesting beach monitoring along with flipper and satellite tagging show the spatial structure of this DPS is largely consistent with viable populations. Foraging is widely dispersed throughout this DPS and also into other DPS's (Limpus 2009). Nesting is widely dispersed throughout the region; there is more than one major nesting site; there is evidence of some connectivity between nesting sites within each of the four regional stocks but no connectivity among regional stocks, and there is nesting on the continental and on islands. The habitat which hosts most of the documented nesting in this DPS is protected (Limpus 2009).

Demographic information for nesting turtles is widely available for nesting beaches in the Australian component of the DPS. The following demographic data are provided by Limpus (2009). For the nGBR stock, nesters at Raine Island average 106 cm CCL (n=20,947) in length, have a 12-day re-nesting interval (n=16), 5.3-year remigration interval (n=2,094), and at nearby Bramble Cay [same stock] nesters on average lay 6.2 clutches per season (n=684). Furthermore, green turtle clutches at Raine Island average 104 eggs (n=501) and have an emergence success of 78.2%. For the southern Great Barrier Reef (sGBR) stock nesters at Heron Island average 107 cm CCL (n=1,942) in length, have a 14-day re-nesting interval (n=264), 5.8-year remigration interval (n=518), and on average lay 5.1 clutches per season (n=878). Green turtle clutches at Heron Island average 114 eggs (n=85) and have an emergence success of 89%.

Growth rates obtained from nearshore capture-mark-recapture studies suggest the sGBR subpopulation attains maturity at 30–40 years (Limpus and Chaloupka 1997; Chaloupka et al. 2004). No similar studies are available for other regions in this population. Stage-based survivorship rates are also available from nearshore studies in sGBR foraging areas. Annual survival was 88% for juveniles, 85% for subadults, and 95% for adults (Chaloupka and Limpus 2005). The high estimate of adult survival should be viewed with caution given a long-term decline in average nester size and increase in remigration interval (Limpus 2009) which could be caused by disproportionally high mortality in adult stage classes.

Diversity

Diversity is a critical element considered under the status review (Seminoff et al. 2015) for green turtles and includes consideration of the overall nesting spatial range, diversity in nesting season, diversity of nesting site structure and orientation (e.g. high vs. low beach face, insular vs. continental nesting sites), and the genetic diversity within the DPS. These are important considerations for assessing the potential impact of catastrophic events such as storms, sea level rise, and disease.

This region has high genetic diversity. It is characterized by high nucleotide diversity resulting from a mix of highly divergent lineages found at nesting sites, some of which are among the oldest lineages found in *C. mydas*.

Nesting and foraging in this region are relatively diverse for green turtles. Nesting is widely dispersed throughout the region, there is more than one major nesting site, and nesting is not completely limited to islands. Nesting, however, is not evenly distributed throughout the DPS, and some of the densest nesting occurs on Raine Island, which has important habitat-based threats. The pivotal temperature for hatchling sex ratio varies within this DPS, with some nesting sites producing primarily females and some producing primarily males (Limpus 2009; Fuentes et al. 2009). Nesting can occur year-round in the most northerly rookeries, but a distinct peak occurs in late December to early January for all Australian rookeries. Foraging occurs year-round and in diverse areas geographically and ecologically (coral and rocky reefs, seagrass meadows and algal turfs on sand and mud flats). In a study of the nGBR nesting assemblages, none were found to pass a threshold for being vulnerable to cyclonic activity (which overlaps with the main nesting season); two were vulnerable to sea level rise, and almost all sites in the study were expected to be vulnerable to increased temperatures by 2070 (Fuentes et al. 2011).

Nesting areas are aggregated into 12 different sites for this DPS. Nesting occurs at moderate to high levels within the Southwest Pacific DPS. Some isolated locations have extremely high nesting activity. The highest nesting assemblages in this DPS, and perhaps of all green turtle DPSs (Chaloupka et al. 2008), are located in the nGBR. Roughly 90% of the nesting activity here occurs at Raine Island and Moulter Cay, with appreciable nesting also occurring at Number Seven and Number Eight Sandbanks and Bramble Cay (Limpus 2009). Estimates of annual nesters at Raine Island vary from 4,000 – 89,000 (Seminoff et al. 2004; NMFS and USFWS 2007; Chaloupka et al. 2008; Limpus 2009).

Green turtle nesting is widely dispersed throughout the Southwest Pacific Ocean (Figure 24). The bulk of this DPS nests within Australia’s Great Barrier Reef (GBR) World Heritage Area and eastern Torres Strait. The northern Great Barrier Reef (nGBR) and Torres Strait support some of the world’s highest concentrations of nesting (Chaloupka et al. 2008). Nesting sites also occur on the Coral Sea Islands, New Caledonia, and Vanuatu. The largest known nesting area for green turtles in New Caledonia is the d’Entrecasteaux atolls, which are located 258 km north of Grande Terre and include Surprise, LeLeixour, Fabre, and Huon Islands (Maison et al. 2010). Vanuatu hosts over 189 nesting sites on 33 islands (Maison et al. 2010).

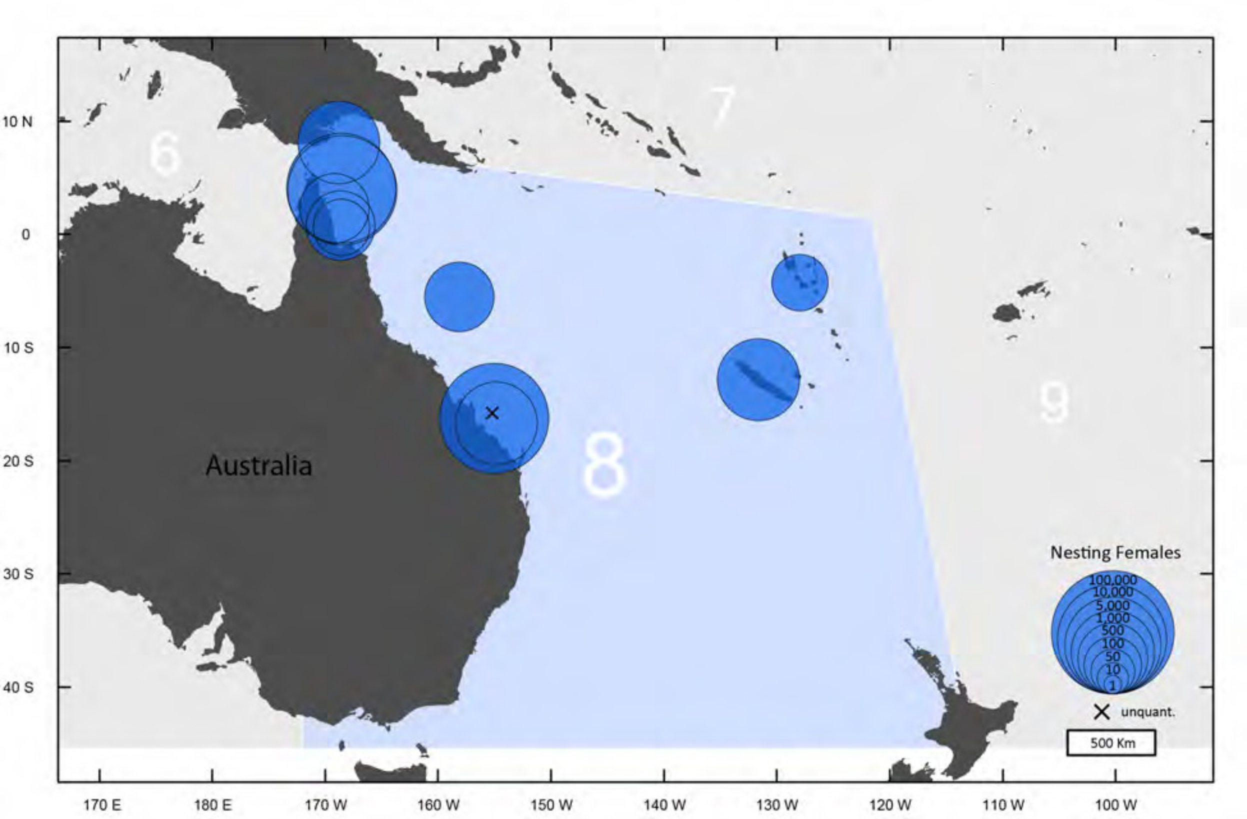

Figure 24. Nesting distribution of green turtles in the Southwest Pacific DPS. Size of circles indicates nesting estimated nester abundance. Locations marked with '·' indicate nesting sites lacking abundance information.

Status

Nesting occurs in many islands throughout the Southwest Pacific DPS, but there are only two nesting areas (Raine Island and Heron Island, described in more detail below) with long-term (>15 years) annual indices of nesting abundance. The Raine Island, Australia index count (1994–2004, intermittent) has high inter-annual variability and a slightly increasing linear trend. Heron Island, Australia, index count (1967– 2004, intermittent) also has high inter-annual variability and a slightly increasing linear trend. Although long robust time series are not available for New Caledonia, recent and historic accounts do not suggest a significant decline in abundance of green turtles nesting in New Caledonia (Maison et al. 2010). The trend at Vanuatu has not been documented (Maison et al. 2010). The Raine Island (nGBR) nesting index is the mean number of females ashore for nesting (during the first 2 weeks of December) that are counted during one survey of the nesting habitat per night (Limpus 2009). The number of nesters observed on nightly tally counts was relatively low from 1975 through the early 1980s, then had higher peaks starting in 1984 (Limpus 2009). From the mid-1990s to the mid-2000s, there has been a leveling off of the rate of increase (Chaloupka et al. 2008). The Heron Island, Australia, index count is derived from a tagging census of the total annual nesting population. There was a 3% per year increase in annual nesting abundance in the subset of data from 1974–1998 (Chaloupka and Limpus 2001) and a similar 3.8% per year increase from the 1974–2002 subset (Chaloupka et al. 2008). When including all years from 1967–2004 there is an increasing linear trend in the annual nesting population size, but the relationship was not significant (Limpus 2009). The increase in annual nesting females at Heron Island is concurrent with an estimated increase of 11% per year from 1985–1992 for the green turtle foraging population (immature and mature females and males) in Heron Reef/Wistari Reef complex (Chaloupka and Limpus 2001).

Threats to the Species

Natural Threats

The potential effects of diseases and endoparasites, as described for other DPSs, also exist for green turtles found in the Southwest Pacific DPS. Low levels of fibropapilloma-associated herpesvirus are common in green turtles in some but not all semi-enclosed waters like Moreton Bay and Repulse Bay in Australia, more infrequent in nearshore open waters and rare in offshore coral reef habitats (Limpus 2009). Mortality and recovery rates from this virus are not quantified but stranded, infected turtles are regularly encountered in south Queensland (Limpus 2009).

Other health conditions such as coccidiosis, parasites, and fungal infections also occur (Limpus, 2009). In late 1991, at least 70 green turtles died from coccidiosis infection in southeastern Queensland, but coccidiosis does not appear to be a static threat given that comparable studies in 1992 and 1993 did not detect the disease (Limpus 2009). Mortality from parasitic worms is not well quantified, but stranded turtles that are heavily infected with blood flukes are regularly encountered in south and north Queensland; and blood flukes and spirochiid trematodes likely cause some green turtle mortality (Limpus 2009). Fungi have been identified in association with green turtles (in cloaca and near nests) and are thought to cause the death of some eggs within the nest (Limpus 2009).

Primary hatchling and egg predators include crabs, birds, fish, and mammals. The magnitude of egg predation is not well documented, but within Australia the highest levels of vertebrate predation on eggs appears occur within other DPSs or for other species (primarily loggerheads) (Environment Australia, 2003). In Vanuatu, nest predation by feral dogs is a primary threat (Maison et al. 2010). Survivorship of hatchlings in southern Great Barrier Reef during the transition from nest to sea (accounting for crab and bird predation) may be quite high (0.98) (Limpus 1971), but survivorship of hatchlings as they (0.4) (Gyuris 1994 as cited in Limpus 2009) transition across the reef flat from the water's edge to deep water is likely considerably lower. Similar survivorship estimates are not available for the northern Great Barrier Reef, but survival during the nest to sea transition are expected to be low and variable, depending on the predator assemblage. Although many birds co-occur with sea turtle hatchlings in the northern Great Barrier Reef, only some birds like the rufous night heron are important predators (Limpus et al. 2003). Terrestrial crabs which occur throughout the northern Great Barrier Reef have been observed feeding on turtle hatchlings and eggs, but the crabs are generally of low density (Limpus et al. 2003). Shark predation on hatchlings as well as adults has been documented (Limpus et al. 2003).

Anthropogenic Threats

Southwest Pacific DPS turtles are 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 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).

Modeling suggests that the population trajectory for the southern Great Barrier Reef nesting aggregation is sensitive to removals of large turtles, consistent with subsistence harvesting (Chaloupka 2002). The nGBR nesting aggregation has less precise data and lacks comprehensive modeling of the population, but it is presumed that the nGBR nesting aggregation is more threatened by harvest than the sGBR nesting aggregation (Limpus 2009). Although there is currently no legal commercial harvest in Australia, there has been intense harvesting in some areas within the last century. The north Australian nesting sites historically had a low intensity of sporadic harvest, but in the summer of 1959 there was a harvest of approximately 1,200 nesting females from Raine Island and Moulter Cay (Limpus et al. 2003). Aside from this intense, short-lived commercial harvest, the nGBR nesting sites (which support the largest documented abundance within this DPS) appear to be relatively undisturbed by largescale sustained commercial harvest (Limpus et al. 2003). In contrast, intense green turtle commercial harvest (sometimes exceeding a thousand turtles per year) in the sGBR nesting sites was longer and occurred intermittently for the first 50 years of the twentieth century (Limpus 2009). Also, in Torres Strait, there is a treaty (the Torres Strait Treaty 1985) that allows people from Papua New Guinea to catch turtles within a shared fishing zone. In Papua New Guinea the take of turtles can be commercial (M. Hamann, James Cook University, pers. comm. 2014 as cited in Seminoff et al. 2015). Incidental capture in artisanal and commercial fisheries is a threat to the survival of green turtles in the Southwest Pacific Ocean. The primary gear types involved in these interactions include trawl fisheries, longlines, drift nets, and set nets. Artisanal and industrial fleets, target a wide variety of species including prawns, crabs, sardines, and large pelagic fish.

Turtles nesting in the Southwest Pacific DPS are vulnerable to the Queensland East Coast Trawl Fisheries and the Torres Strait Prawn Fishery, and to the extent they forage west of Torres Strait, they are also vulnerable to the northern prawn fishery. Total mortality of Australian green turtles in fisheries bycatch is not known because there is not reliable reporting of threatened species bycatch in Australian commercial fisheries (Limpus 2009). Australian trawl fisheries have increased the number of boats, the length of the shot-times and the number and size of nets towed since the 1960s, but the capture of green turtles has been less frequently reported in prawn trawls in Queensland than loggerheads (Limpus 2009). The total mortality from eastern and northern prawn fishery is estimated to be 50–100 green turtles per year from the late 1980s to the late 1990s. TEDs have been required in most of Australia's prawn fisheries since 2002 or earlier (northern prawn fishery since April 2000, East Coast Trawl Fisheries since December 2000, Torres Strait Prawn Fishery since March 2002, and Western Australian prawn and scallop trawl fisheries since 2002). TEDs are thought to reduce turtle captures in northern prawn fisheries by two orders of magnitude. The reported number of all species of turtles caught in the northern prawn fishery is 883 in 1999, 68 in 2000, 113 in 2001, 27 in 2002 and 2003 (Australian Government; http://www.environment.gov.au/soe/2006/publications/drs/indicator/133/index.html#issuesforwhichthisisanindicatorandwhy).

The use of TEDs in the northern prawn fishery became mandatory, due in part to several factors: (1) Objectives of the Australian Recovery Plan for Marine Turtles, (2) requirement of the Australian Environment Protection and Biodiversity Conservation Act for Commonwealth fisheries to become ecologically sustainable, and (3) the 1996 U.S. import embargo on wild-caught prawns taken in a fishery without adequate turtle bycatch management practices (Robins et al. 2002). Australian and international longline fisheries capture marine turtles. Precise estimates of international capture of Southwest Pacific Ocean DPS green turtles by the international longline fleet are not available, but they are thought to be larger than the Australian component (DEWHA 2010).

Turtle bycatch by the Eastern Tuna and Billfish Fishery of Australia has been dominated by green and leatherback sea turtles, the vast majority of which are released alive (81% in 2006, and 88% in 2007) (DEWHA 2010). According to the Australian government, the average annual bycatch of all species of turtles within Australia’s Eastern Tuna and Billfish fishery is 42 from 1997 through 2004 and 16.5 in 2006 and 2008; turtle interactions in the Western Tuna and Billfish Fishery is lower (see link above; DEWHA 2010).

To assess the impact of bycatch by integrating information on bycatch rates, mortality rates, and body sizes, Wallace et al. (2013) assigned a bycatch impact score to Regional Management Units for various fisheries. Longline fisheries were determined to be a low risk for this population (Wallace et al. 2013). In addition to threats from prawn trawls fisheries, green turtles may be threatened by other fishing gear (summarized from Limpus 2009). Although tunnel nets capture many green turtles, they do not appear to have substantial mortality rates. Gill nets (targeting barramundi, salmon, mackerel, and shark) in Queensland and the Northern Territory have been observed to catch green turtles, but the magnitude has not been quantified. Crab pots and float lines entangle green turtles and, although the magnitude of mortality is not quantified, it is presumed to be in the tens per year. Untended “ghost” fishing gear that has been intentionally discarded or lost due to weather conditions may entangle and kill many hundreds of green turtles annually.

NMFS estimates that between 0-8% of the turtles caught in the shallow-set longline fishery could be from the Southwest Pacific DPS (P. Dutton pers. comm. November 20, 2017). This means that one turtle may have been captured from this DPS by the shallow-set longline fishery since 2004. NMFS estimates that between 0-25% of the turtles caught in the HI DSLL fishery could be from the Southwest Pacific DPS (P. Dutton pers. comm. June 29, 2018). This means that 0-17 turtles were captured from this DPS by the HI DSLL fishery since 2004. NMFS estimates that between 11.6-46.5% of the turtles caught in the American Samoa longline fishery could be from the Southwest Pacific DPS (P. Dutton pers. comm. July 5, 2018). This means that between 31-126 turtles were captured from this DPS by the American Samoa longline fishery since 2006.

Green turtles are captured in shark control programs in Australia, but management considerations are being considered to reduce the impact. The average yearly capture of all species of sea turtles from 1962– 1995 was 119.4 turtles per year, with > 35% released alive (59% were released with undocumented condition) (Gribble et al. 1998). Observed green turtle annual mortality during 1998–2003 was 2.7 per year (Limpus 2009). Green turtles have been captured in the New South Wales shark meshing program since 1937, but total capture for all turtle species from 1950 through 1993 is roughly 5 or fewer turtles per year (Krogh and Reid 1996). Post release survival does not appear to have been monitored in any of the monitoring programs.

4.2.3.6 East Indian/West Pacific DPS

Differential Distribution

The western boundary for the East Indian–West Pacific DPS is 84°E longitude from 40°S to where it coincides with India near Odisha, northeast and into the West Pacific Ocean to include Taiwan extending east at 41°N to 146°E longitude, south west to 4.5°N, 129°E, then south and east to West Papua in Indonesia (at 135°E) and the Torres Straits in Australia (at 142°E longitude). The southern boundary is 40°S latitude, encompassing the Gulf of Carpentaria. Green turtle nesting is widely dispersed throughout the East Indian-West Pacific DPS, with important nesting sites occurring in Northern Australia, Indonesia, Malaysia (Sabah and Sarawak Turtle Islands), Peninsular Malaysia, and the Philippine Turtle Islands.

Adult Foraging and Diving

See *Adult Foraging and Diving* above for the description of feeding and diving behavior.

Population Dynamics

Structure

Genetic sampling in the East Indian-West Pacific DPS has occurred at more than 22 rookeries. There appears to be a complex population structure, even though there are gaps in sampling relative to distribution (e.g., in Thailand, Vietnam, parts of Indonesia, and the Philippines). Overall, this region is dominated by a few common and widespread haplotypes and has varying levels of spatial structure characterized by the presence of rare/unique haplotypes at most rookeries. Within the DPS, there is significant population substructuring (pairwise FST 0.10–0.95, $p<0.05$). Of 22 rookeries studied, 16 regional genetic stocks have been identified in the East Indian-West Pacific DPS: 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). Tagging and tracking studies have been geared to studying internesting migrations, and defining the range of internesting habitats and post-nesting migrations. Green turtles that were satellite tracked from Pulau Redang, Terengganu indicate migrations to the South China Sea and Sulu Sea areas (Liew 2002). Cheng (2000) reported movements of eight post-nesting green turtles from Wan-An Island, Taiwan using Argos-linked satellite transmitters.

The turtles distributed widely on the continental shelf to the east of mainland China. Destinations included southern Japan (Kyushu and Okinawa), Taiwan, and mainland China. Satellite telemetry studies conducted from 2000 to 2003 demonstrated that the green turtles nesting at Taipin Tao are a shared natural resource among the nations in the southern South China Sea. Green turtle females tracked in the same area travelled long distances commencing a post-nesting migration. Eleven green turtles tracked with satellite transmitters migrated in two general directions: the first route stretched eastward along the eastern coast of the Gulf of Thailand to the Vietnam peninsula then some crossed the South China Sea and entered Sulu Sea in the Philippines; and the second route went south across the Gulf of Thailand to the Malaysia peninsula travelling a distance that ranged from 456 to 2,823 km (Charuchinda et al. 2002) to the China Sea, and the remaining one migrated north to the coastal region of Japan (Wang 2006). Waayers and Fitzpatrick (2013) found that in the Kimberly region of Australia, the green turtle appears to have a broad migration distribution and numerous potential foraging areas.

Mixed stock analysis of foraging grounds shows that green turtles from multiple nesting beach origins commonly mix at feeding areas in foraging grounds across northern Australia (Dethmers et al. 2010) and Malaysia (Jensen 2010) with higher contributions from nearby large rookeries. There is evidence of low frequency contribution from rookeries outside the DPS at some foraging areas. The demography of green turtles in the East Indian-West Pacific DPS varies throughout the nesting assemblages. This variation in parameters such as mean nesting size, remigration interval, internesting interval, clutch size, hatching success, and clutch frequency suggests a high level of population structuring in this DPS. The size of nesters throughout the DPS range from 82.1 cm CCL to 103.6 cm CCL (Trono 1991; Hirth 1997; Charuchinda and Monanunsap 1998; Basintal 2002). Growth rates are 0.83 cm/yr. for nesting females according to Pilcher and Basintal (2000). Clutch size varies among rookeries from 87.2 to 115 eggs per nest.

Remigration interval also varies from 2 to 5 years, and clutch frequency from 1.67 to 8 nests per season. Hatching success ranges from 37 to 94% with some sites in incubation facilities (Hendrickson 1958; Suwelo 1971; Trono 1991; Leh 1994; Hirth 1997; Abe et al. 1998, 2003; Charuchinda and Monanunsap 1998; Pilcher and Basintal 2000; Tiwol and Cabanban 2000; Basintal 2002; Chan et al. 2007; Kobayashi et al. 2008; Adnyana et al. 2008; Zainudin et al. 2008; Lwin 2009a, 2009b; Cheng et al. 2009; Jensen 2010; Waayers 2010; Chen et al. 2010; Dethmers 2010; Muhara and Herlina 2012; Reischig et al. 2012).

Diversity

The components considered under diversity include the overall nesting spatial range, diversity in nesting season, diversity of nesting site structure and orientation (e.g., high vs. low beach face, insular vs. continental nesting sites), and the genetic diversity within the DPS. Components such as these are important considerations for assessing the potential impact of events and phenomena such as storms, sea level rise, and disease. Nesting and foraging areas are widespread within this DPS, providing a level of population resilience through habitat diversity. The nesting season varies throughout the DPS, with nesting from June to August in the inner Gulf of Thailand. Peak nesting occurs from March to July on Derawan Island (Charuchinda and Monanunsap 1998; Abe et al. 2003; Aureggi et al. 2004; Adnyana et al. 2008), and year round in Thameela Island, Myanmar and in Aru, Indonesia, with peak nesting from November to March in Aru (Lwin 2009a; Dethmers 2010). Peak nesting occurs from November to March in Sukamade, southeastern Java (Arinal 1997), Barrow Island, Australia and Western Australia (Pendoley 2005).

Nesting occurs on both insular and continental sites, yielding a degree of nesting diversity. Limited information also suggests that there are two types of nesters within the DPS, those with high site fidelity which nest regularly at one site, such as the Sabah Turtle Islands, and those with low site fidelity such as at Ishigaki Island (Basintal 2002; Abe et al. 2003).

The largest nesting site lies within Northern Australia, which supports approximately 25,000 nesting females, calculated from the 5,000 nesting female's order of magnitude (Figure 25) (Limpus 2009). Currently, the East Indian-West Pacific DPS hosts 58 reported nesting sites (in some cases nesting sites are made up of multiple beaches based on nesting survey information) with six of these sites supporting more than 5,000 nesting females each (including the 25,000 nesters in Northern Australia). Nonetheless, populations are substantially depleted from historical levels.

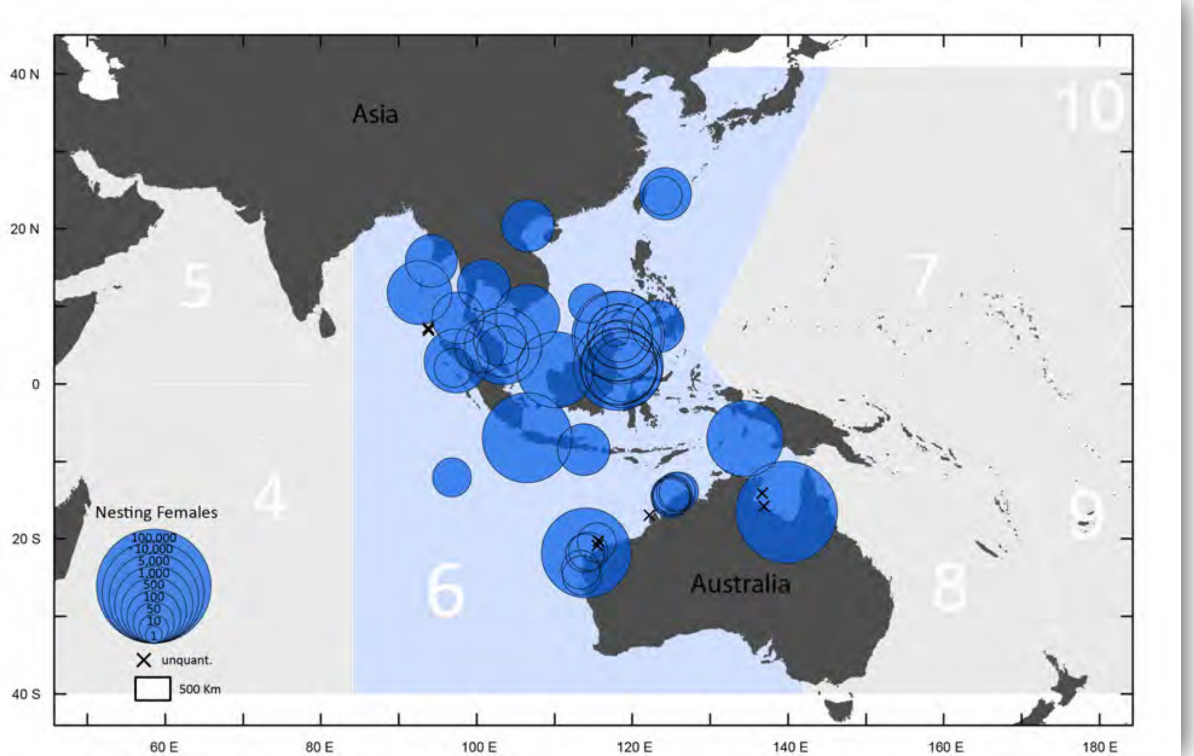

Figure 25**.** Nesting distribution of green turtles in East Indian-West Pacific DPS (blue-shaded area). Size of circles indicates estimated nester abundance. Locations marked with '·' indicate nesting sites lacking abundance information.

The in-water range of the East Indian-West Pacific DPS is similarly widespread with shared foraging sites throughout the DPS. Tagged green turtles that nest in Western Australia have been re-sighted in Arnhem Land and as far north as the Java Sea near Indonesia (Baldwin et al. 2003; Limpus et al. 2007). The extensive coastline and islands of Indonesia support a large range of nesting and foraging habitat for green turtles (Halim and Dermawan 1999). Waayers and Fitzpatrick (2013) found that in the Kimberly region of Australia, the green turtle appears to have a broad migration distribution and numerous potential foraging areas. A satellite-tagged female green turtle at Redang, Malaysia, travelled near Koh Samui, Thailand (Liew 2002). Green turtle foraging grounds are known around the Andaman and Nicobar Islands (Andrews et al. 2006a).

Status

There are four sites for which 15 or more years of recent data are available for annual nester abundance: Sabah Turtle Islands in Malaysia; Royal Navy Center in Khram Island, Thailand; Redang in Terrengganu, Myanmar; and Thameela Islands, Myanmar. Only Sabah Turtle Islands represent a sizable nesting population, estimated at 7,011 in 2011. The PVA indicates that the nesters from Sabah Turtle Islands in Malaysia, with an estimated 7,000 nesters, will likely continue to increase, while the nesters from the Royal Navy Center in Khram Island, Thailand (estimated 297 nesters), Redang in Terrengganu, Myanmar (estimated 278 nesters), and Thameela Islands, Myanmar (estimated 109 nesters) will likely continue to decline (Seminoff et al. 2015). In water surveys off the coast of Malaysia calculated the sex ratio to be 1M:4F for juveniles in the area (Pilcher 2010).

For the Sabah Turtle Islands in Malaysia, the probability that this population will fall below the trend reference point (50% decline) at the end of 100 years approaches zero. The probability that this population falls below the absolute abundance reference (100 females per year) at the end of 100 also approaches zero. This trend may be a result of effective conservation measures made by the Sabah Government in the 1970s when the Sabah Turtle Islands were acquired from private ownership to provide complete protection to the nesting turtles and their eggs (Chan 2006).

Annual nesting in the Khram Island, Sea Turtle Conservation Center of the Royal Thai Navy, Gulf of Thailand has decreased from a mean of approximately 405 nests per year between 1975–1983 to a mean of approximately 250 nests per year from 1992–2001 (Charuchinda and Monanunsap 1998; Charuchinda et al. 2002). For these beaches, there is a nearly 100% probability that this population will fall below the trend reference point (50% decline) within 100 years. There is also a nearly 100% probability that this population falls below the absolute abundance reference (100 females per year) at the end of 100 years. For Redang in Terengganu, Malaysia, there is a 72.9% probability that this population will fall below the trend reference point (50% decline) at the end of 100 years. There is an 89.8% probability that this population falls below the absolute abundance reference (100 females per year) at the end of 100 years. For Thameela in Myanmar, there is an 87.9% probability that this population will fall below the trend reference point (50% decline) at the end of 100 years. There is a 96.7% probability that this population falls below the absolute abundance reference (100 females per year) at the end of 100 years (Seminoff et al. 2015).

Threats to the Species

Natural Threats

Fibropapillomatosis has been found in green 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), 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.

The best available data suggest that current nest and hatchling predation on the East Indian-West Pacific DPS is prevalent. Depredation of nests by feral animals is also widespread in many South Asian areas (Sunderraj et al. 2001; Islam 2002). Nest predation by feral pigs and dogs is a major threat on the Andaman and Nicobar Islands of India (Fatima et al. 2011). Jackals, foxes, wild boars, and monitor lizards also predate green turtle nests and hatchlings along the beaches of Bangladesh (Andrews et al. 2006a). Lizards and ghost crabs are the natural predators of green turtle nests in several areas (Chantrapornsyl 1993). In Malaysia, ghost crabs predate green turtle eggs (Ali and Ibrahim, 2000), and gold-ringed cat snakes or mangrove snakes, (Asiatic) reticulated pythons, monitor lizards, and house mice predate hatchlings (Hendrickson, 1958). Monitor lizards, crabs, and ants predate eggs and hatchlings on the beaches of Vietnam (as cited in "Sea Turtle Migration-Tracking and Coastal Habitat Education Program– An Educator's Guide" http://www.ioseaturtles.org/Education/seaturtlebooklet.pdf). In Japan, raccoon dogs and weasels are a threat to nests (Kamezaki et al. 2003). In Taiwan, snakes predate the nests (Cheng et al. 2009). Hendrickson (1958) estimated that 4% of the adult females on Malaysian beaches showed signs of assumed shark attack-amputated flippers and missing shell. It has been speculated that sharks congregate in large numbers around the Sarawak Turtle Islands during the peak breeding season (Hirth 1997). On the North West Cape and the beaches of the Ningaloo coast of mainland Australia, a long established feral European red fox population historically preyed heavily on eggs and is thought to be responsible for the lower numbers of nesting turtles on the mainland beaches (Baldwin et al. 2003; Kelliher et al. 2011). During the 2010–2011 nesting season, foxes predated 23% of all the nests laid along the North West Cape and Cape Range Division of the Ningaloo Coast (Kelliher et al. 2011).

Anthropogenic Threats

Despite substantial declines in green turtle nesting numbers, egg harvest remains legal in several of the countries within this DPS (Humber et al. 2014). In Bangladesh, green turtle nesting was common on most of these beaches. In 1989, 35 green turtles were recorded nesting in one night on one beach in St. Martin, Bangladesh (Islam 2002). Egg collection is considered the most serious threat for the few green turtle nests in Bangladesh if not relocated to a hatchery (Islam et al. 1999 as cited in Islam 2001, 2002). Over-exploitation has brought the nesting turtles to near extinction (Hasan 2009). Turtle eggs were historically collected and sold to visitors from the mainland of Myanmar, with about 1.6 million green turtle eggs harvested annually in the early 1900s (Thorbjarnarson et al. 2000; Islam 2002). Prior to 1986, virtually all eggs were collected. From 1986 to 1996, one third of the eggs were required to be left to hatch. After 1997, the Myanmar Fisheries Department fully protected all beaches where turtle nesting still occurred (Thorbjarnarson et al. 2000), and collection of eggs and harvest of nesting females were banned. However, sea turtle eggs and nesting females continue to be harvested due to a lack of law enforcement (Islam 2001).

In Thailand, the major threat to sea turtles is the exploitation of eggs and turtles for meat and turtle products. Egg collection continues on remote beaches that are not regularly monitored (Charuchinda et al. 2002). In Myanmar and Thailand, hatcheries are set up to protect a portion of the eggs. However, these hatcheries retain hatchlings for several days for tourism purposes, thus reducing the likelihood of hatchling survival (Charuchinda et al. 2002). In the 1950s, the green turtle nesting population in Malaysia started showing a decline after decades of egg collection (Chan 2006). In the early 1970s, less than 10% of eggs were retained for incubation in hatcheries in peninsular Malaysia. Over 4,100,000 eggs were harvested in Sarawak between 1967 and 1978, of which only 2% were transplanted to hatcheries. Green turtle eggs were nearly completely harvested in Sarawak right up until the 1980s. In 2001, the%age of eggs protected in peninsular Malaysia increased to approximately 50%; the remainder was marketed (Siow and Moll 1982). Turtle nesting numbers began to decrease in peninsular Malaysia where the number of eggs laid in Terrengganu went from 928,900 in 1956 to between 107,135 and 417,981 annually from 1984 to 1989 (Mortimer 1992).

In Sabah, from 1965 to 1978, a total of over 6,000,000 eggs were collected, and approximately 2,700,000 were transplanted to hatcheries (Siow and Moll 1982). After 40 years of intense egg harvest in Sabah, the nesting population declined (de Silva 1982; Limpus, 1995). It was believed this decline could be attributed to egg harvesting, although turtles were also threatened from incidental capture in fishing gear (Mortimer 1991a). In order to provide some protection for turtles, all three Sabah Turtle Islands were acquired and protected by the Sabah State Government in the 1970s (de Silva 1982). Egg collection dropped to approximately 250,000 in the early 1980s, but had increased to nearly 1 million eggs by the late 1990s (E. Chan, Institute of Oceanography, Kolej Universeti Sains dan Teknogli, Malaysia, pers. comm. 2002 as cited in NMFS and USFWS 2007). Despite the protections on the three Sabah Turtle Islands, the nesting population continued to decline until 1987 when there were signs of some recovery for green turtles (Pilcher 2000). However, after more than 20 years of conservation efforts (1970–1990), the population had still not shown signs of recovery (Limpus et al. 2001).

At Pahgumbahan 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). 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 till 1999. Illegal egg harvesting continues, but there is an increased effort to fully protect green turtles from harvest on the islands of Bilang-Bilangan and Mataha in Indonesia (Reischig et al. 2012). There are a few beaches in Malaysia, Indonesia, and Thailand where eggs are protected in hatcheries. Malaysian hatcheries in Sabah were found to produce 100% females, which will skew green turtle sex ratios in the wild (Tiwol and Cabanban 2000). Pilcher (2010) found that 80% of green turtles foraging off the coast of Borneo, Malaysia were female likely resulting from long-term hatchery practices. In addition to the harvest for meat, eggs were also harvested throughout the Indonesian archipelago with many as 2 million eggs taken off the beaches every year (Limpus 1997).

In the Turtle Islands, owned by both the Philippines and Malaysia, an 88% decline in egg production between 1959 and 1992 can be attributed to the almost complete exploitation of all the eggs. However, the collection of eggs is now regulated and of an estimated egg production of 9,022,553 eggs between 1984 and 1992, 65% were conserved (Hirth 1997). From 1984 to 2000, 71% of the 21,678,109 eggs laid in the Tawi-Tawi province of the Philippines were conserved, while 21% of the eggs were collected for consumption (Cruz 2002).

Egg harvest remains a problem in Vietnam and was a principal factor in the decline of turtles nesting in that nation. Because nesting has declined so dramatically, egg harvest has also declined and become scattered and inconsistent. Although sea turtle nests are currently protected on Con Dao National Park and Nui Chua beaches in Vietnam, in unprotected areas nearly 100% of eggs are harvested (Ministry of Fisheries 2003). Green turtle nests (less than 10) laid on the

Vietnam beaches along the Gulf of Tonkin have been reported as being susceptible to collection (Hamann et al. 2006a). Because of the decline of turtles in Vietnam, the number of turtles caught for consumption has also decreased; however, captures have been reported to continue at a low rate in seven coastal communities where at least one family catches turtles (Hamann et al. 2005).

In Japan, egg collection was common in the coastal areas during times of hunger and later by those who acquired them on the black market (Kamezaki et al. 2003) but is no longer a problem (Abe et al. 2003; Kamezaki et al. 2003). Currently, egg poaching in Japan is illegal due in large part to research and conservation efforts throughout the country. Laws were enacted in 1973 to prohibit egg collection on Yakushima, and in 1988, the laws were extended to the entire Kagoshima Prefecture (Matsuzawa 2006).

Nesting females continue to be killed in countries within Southeast Asia and the Indian Ocean (Fleming 2001; Fretey 2001; Cruz 2002). In the 1800s, turtles and turtle eggs were an important food source for the indigenous people of the Andaman and Nicobar Islands in India (Shanker and Andrews 2004). Egg and turtle harvest remains at a subsistence level. In Bangladesh, since the 1980s, green turtle nesting populations have declined due to severe exploitation of eggs and illegal killing of adult turtles (Islam 2002). Indonesia has a lengthy history of exporting sea turtle products continuing to the 1990s (Milliken and Tokunaga 1987; Groombridge and Luxmoore 1989). Local islanders in Indonesia have traditionally considered turtles, especially green turtles, as part of their diet (Hitipeuw and Pet-Soede 2004 as cited in FAO 2004). About 25,000 green 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 turtle being the most common turtle.

Green turtles can be found in the waters and nesting along the beaches of the Kai islands in Indonesia. They are harvested whenever encountered to be used as meat. Suárez (2000, as cited in Limpus 2009) recorded 173 green turtles captured with nets or hooks in the water or taken on the nesting beach during a 6-month period. The green turtle populations that formerly nested on Bali have been extirpated (Schulz 1984; Groombridge and Luxmoore 1989), but thousands of green turtles were being brought into Bali each year (21,000 in 1990), where they were butchered for meat for personal consumption and for Balinese ceremonies and rituals (Barr 2001). In 1990 the Balinese government decreed that green turtle utilization would be limited to a maximum of 5,000 turtles per year, though actual numbers may be more than 50% higher (Halim et al. 2001). Available evidence indicates that egg and turtle harvests (on the nesting beach and at sea) are far in excess of sustainable levels (Groombridge and Luxmoore 1989; Barr 2001). Turtle fisheries continue around Aru primarily for trade in Bali.

On the main islands within the Aru archipelago, where green turtles come to nest, the inhabitants are dependent solely on marine resources (Dethmers 2010). Drift nets are set near the nesting beach and seagrass beds catching an average of 15 turtles per night (Dethmers 2010). Many nesting turtles are collected in the waters just off the Indonesian beaches where some fishermen collect as many as 300 turtles on a trip. This type of harvest extends out to Aru, Southeast Sulawesi, East Kalimantan, Irian Jaya, Madura, Timor and Flores. About 25% of the harvested turtles are males, which confirms that in addition to the harvest of nesting females on the beach, harvest also occurs at foraging and courtship grounds. Mostly in the remote areas of the Philippines, green turtles are still killed and sold for meat, and eggs continue to be exploited for consumption and trading. The Pawikan Conservation Project was created in 1979 to address the decline of sea turtles in the Philippines and has been effective in promoting conservation and scientific management of the sea turtle resources although much still needs to be done (Ramirez-de Veyra 1994).

Green turtles are being incidentally taken during fishing activities in the waters surrounding the Turtle Islands, and the number of turtles taken is increasing with the increasing number of fishing vessels, particularly during illegal fishing operations by Chinese vessels. In 2002 alone, four vessels from China were caught with more than 58 turtles onboard, mostly green turtles (Cruz 2002). Despite increased conservation efforts, the killing of turtles and selling of turtle meat still occurs in the Philippines, mostly in remote areas (Cruz 2002). At-sea poaching of turtles is a continuing problem in Southeast Asia, especially by Hainanese and Vietnamese vessels. The poaching occurs in a wide-ranging area of the region, and has moved as turtle populations have been depleted, with vessels being apprehended off Malaysia, Indonesia, and the Philippines (Pilcher et al. 2009 as cited in Lam et al. 2011). The apprehension of Chinese vessels with large numbers of sea turtles (tens to hundreds), including green turtles, throughout the eastern Indian Ocean and South China Sea (Lam et al. 2011) highlights the problem, though it likely represents only a small portion of the poaching that occurs. It is notable that many of the fishermen that have been apprehended are aware of the laws and associated penalties for harvesting marine turtles, but do so under the cover of darkness and other times when they are aware that enforcement is limited (Lam et al. 2011).

Licensed fisherman in Japan can legally catch sea turtles for local consumption (Horikoshi et al. 1994). The annual number of turtles caught is 150. Of these turtles the majority are immature green turtles caught in the Yaeyama Islands (Abe et al. 2003).

In Australia, green turtles are harvested by Aboriginal and Torres Strait Islanders for subsistence purposes. Tens of thousands of turtles were harvested by indigenous people in the Ningaloo Region of Australia from the 1950s to the early 1970s (Limpus 2002). The total annual harvest in Australian waters in the 1970s was estimated to be between 7,500 and 10,500 (Kowarsky 1982). The most common method of capturing turtles is by harpoon from a boat. However, today there is a widespread use of motorized aluminum boats in contrast to the traditional dugout canoes powered by paddles or sail. Daly (1990) reported an estimate of 10,000 adult green turtles being harvested in the Torres Strait with about 4,000 of these taken by Torres Strait islanders and about 6,000 by Papua New Guineans for sale in their coastal markets (Hirth and Rohovit 1992). In 2001, Morris and Lapwood recorded 96 green turtles were harvested on the Dampier Peninsula over a 4 month period, the majority adult-sized females, and estimated 500 green turtles harvested annually (K. Morris, pers. comm. as cited in Limpus 2009). The total harvest of green turtles by indigenous people across northern and Western Australia is probably several thousand annually (Kowarsky 1982; Henry and Lyle 2003 as cited in Limpus 2009). The indigenous harvest of eggs may be unsustainable in northeast Arnhem Land (Kennett et al. 1998).

NMFS estimates that between 0-10% of the turtles caught in the shallow-set longline fishery could be from the East Indian/West Pacific DPS (P. Dutton pers. comm. November 20, 2017). This means that one turtle may have been captured from this DPS by the shallow-set longline fishery since 2004. NMFS estimates that between 0-28% of the turtles caught in the HI DSLL fishery could be from the East Indian/West Pacific DPS (P. Dutton pers. comm. June 29, 2018). This means that 0-19 turtles were captured from this DPS by the HI DSLL fishery since 2004. NMFS estimates that between 0-11.4% of the turtles caught in the American Samoa longline fishery could be from the East Indian-West Pacific DPS (P. Dutton pers. comm. July 5, 2018).

This means that between 0-31 turtles were captured from this DPS by the American Samoa longline fishery since 2006.

4.2.3.7 Summary of the Status of Green Sea Turtles

In this section of this biological opinion, we explained that the Central North Pacific, Eastern Pacific, Southwest Pacific, and East Indian-West Pacific DPSs of green sea turtle is threatened; and Central West Pacific and Central South Pacific DPSs are endangered. 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 26).

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 26). 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 North Pacific and Eastern Pacific green sea turtle DPSs as the majority of interactions occur with these two DPSs. Additionally, we looked at the remaining DPSs as well for any threats which may be applicable. However, as previously noted, caveats regarding the populations' trend apply to various DPSs. The Central Western Pacific DPS has some nesting locations which are increasing in abundance while other sites have unknown trends; not enough data is present for the Central South Pacific DPS to conclude an overall trend; and the East-Indian West Pacific DPS has both increasing and decreasing abundance estimates depending on location. However, not enough information is available at this time to determine and overall trend for this particular DPS.

In our analysis of each DPS, we concluded most DPSs face the same general threats. For instance, hatchling predation is encountered by all DPS; however, the predatory species may change depending on geographic location. Those dynamics are consistent with the status and trend of the count data: green sea turtles 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 26). 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 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 (Figure 26), 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 both the Central North Pacific and Eastern Pacific green sea turtle DPSs.

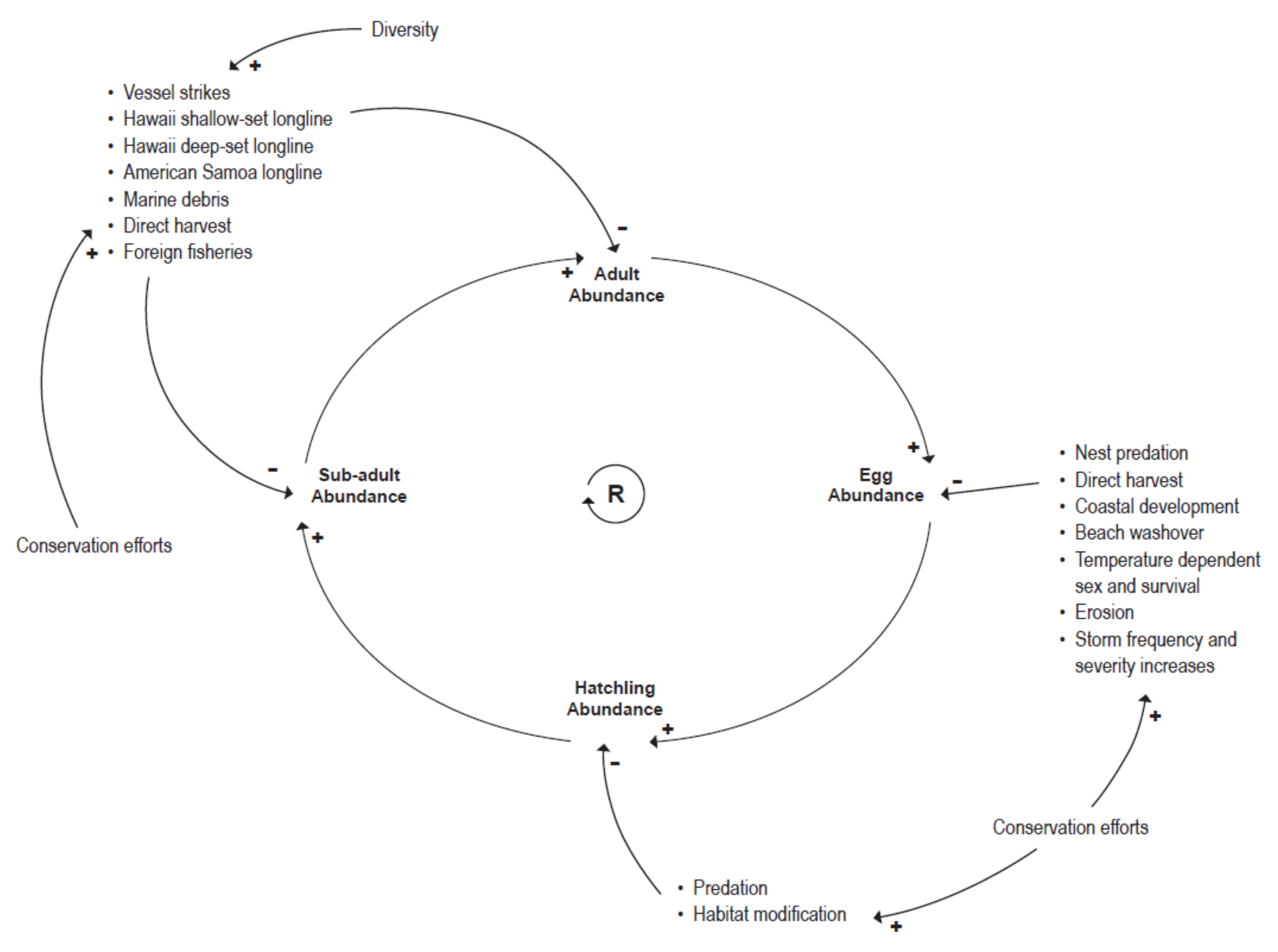

Figure 26. Green sea turtle casual loop diagram depicting various stressor-response relationships for each life cycle stage applicable to each DPS under consideration.

4.2.4 Olive Ridley Sea Turtle

4.2.4.1 Distribution

Olive ridley sea turtles are globally distributed in the tropical regions of the South Atlantic, Pacific, and Indian Oceans. In the South Atlantic Ocean, they are found along the Atlantic coasts of West Africa and South America. In the Eastern Pacific, they occur from Southern California to Northern Chile. They are found in coastal waters of over 80 countries (Abreu-Grobois and Plotkin 2008). There are two listed populations, one which is listed as threatened globally, and the other population which nests on the Pacific coast of Mexico is listed as endangered. At the time of listing (1978), there was not an option to list a DPS, however a substantial amount of information has become avialable on the population but the Services have not fully assembled or analyzed the new information (NMFS and FWS 2014).The data appears to indicate a possible separation as olive ridley sea turtles are not known to move between or among ocean basins (NMFS and FWS 2014).

Differential Distribution

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 1986; Arenas and Hall 1992; Pitman 1991, 1993; Plotkin 1994, 2010; Plotkin et al. 1994, 1995; Beavers and Cassano 1996). They often associate with the highly productive area called the Costa Rica Dome located between 8 to 10°N and 88 to 90°W, which is characterized by a shallow (within 10 m of the surface) thermocline and areas of upwelled waters rich in prey items (Swimmer et al. 2009). Olive ridley sea turtles appear to forage throughout the eastern tropical Pacific Ocean, often in large groups, or flotillas, and are occasionally associated with floating debris (Arenas and Hall 1992 in NMFS and FWS 2014). Olive ridley sea turtles occupy the neritic zone during the breeding season. Some reproductively active males and females migrate toward the coast and aggregate at nearshore breeding grounds located near nesting beaches (Hughes and Richard 1974; Pritchard 1969; Cornelius 1986; Plotkin et al. 1991, 1996, 1997; Kalb et al. 1995). A significant proportion of the breeding also takes place far from shore (Pitman 1991; Kopitsky et al. 2000), and some males and females may not migrate to nearshore breeding aggregations at all. Some males appear to remain in oceanic waters, are non-aggregated, and mate opportunistically as they intercept females en route to near shore breeding grounds and nesting beaches (Plotkin 1994; Plotkin, et al. 1994, 1996; Kopitsky et al. 2000; Parker et al. 2003).

The post-reproductive migrations of olive ridleys in the eastern Pacific Ocean are unique and complex. Their migratory pathways vary annually (Plotkin 1994, 2010), there is no spatial and temporal overlap in migratory pathways among groups or cohorts of turtles (Plotkin et al. 1994, 1995), and no apparent migration corridors exist (Plotkin 2010). Unlike other sea turtles that migrate from a breeding ground to a single feeding area, where they reside until the next breeding season, olive ridley sea turtles are nomadic migrants that swim hundreds to thousands of kilometers over vast oceanic areas (Plotkin 1994, 2010; Plotkin et al. 1994, 1995; Parker et al. 2003). This nomadic behavior may be unique to olive ridley sea turtles in the eastern Pacific Ocean as studies in other ocean basins indicate olive ridley sea turtles occupy neritic waters and do not make the extensive migrations observed in this region (Plotkin 2010).

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). Long-term satellite tracking data of 30 eastern Pacific post-nesting olive ridley sea turtles revealed that they were widely distributed in the pelagic zone from Mexico to Peru and lacked migratory corridors (Plotkin 2010). These turtles migrated long distances, swam continuously, displayed no fidelity to specific feeding habitats, and were nomadic.

Foraging and diving behavior of the olive ridley sea turtle is poorly understood when compared to other sea turtle species (Hochsheid 2014; NMFS and FWS 2014). Available data suggests post nesting females in the Western Pacific Ocean display plasticity by foraging in both coastal and oceanic habitats. One turtle spent 40 weeks in an area less than 150 km2, and multiple turtles overlapped in the areas they foraged (Whiting et al. 2007; NMFS and FWS 2014). McMahon et al. (2007) reported three of four females foraged in relatively deep water (> 100 m) with the maximum dive lasting 3.33 + 0.33 h, indicating long dives towards the seabed. Olive ridley sea turtles forage on a variety of marine organisms, including tunicates, gastropods, crustaceans, and fishes that tend to migrate with the deep scattering layer. As a result, olive ridley sea turtles typically forage in deep water far from shore (Polovina et al. 2003, 2004; NMFS 2006; McMahon et al. 2007).

4.2.4.2 Population Dynamics

Structure

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). Fine-scale population structure also was not found across 13 solitary and arribada nesting beaches along the Baja Peninsula and the main coast of Mexico (Rodríguez-Zárate et al. 2013). Genetic diversity was low among these nesting sites indicating a population collapse likely due to localized over exploitation (Rodríguez-Zárate et al. 2013). Lopez-Castro and Rocha-Olivares (2005) found genetic diversity in solitary nesting assemblages from the Baja California Peninsula to be significantly lower than arribada nesting populations along the east Pacific coast of Mexico and Costa Rica. They concluded that the genetic composition of the Baja population indicates reproductive isolation and genetic differentiation. They believed that the loss of genetic diversity and the differences in mating strategies distinguished the Baja population from the arribada beaches on the main continent, and recommended that the peninsular population be considered a distinct management unit (Lopez- Castro and Rocha-Olivares 2005). Wallace et al. (2010) 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.

Diversity

Large data gaps still exist in this species demography, including age and sex distribution; growth, birth, and death rates; immigration, and emigration (Zug et al. 2006; NMFS and USFWS 2007, 2014). Several studies can provide insight into the potential demographics for the species. However, after extensive literature review, morphometric measurements for male turtles were lacking overall. Two studies provide measurements of males, however they are only for breeding adults. Shanker et al. (2003) provided measurements (n=1,526) with mean SCL measurements ranging between 64.2 cm and 67.1 cm. It should be noted that turtles were measured over multiple mating seasons, signifying potential bias as only males mature enough for mating purposes were selected. The same occurred with Plotkin et al. (1996), whom measured 8 males over two years and found SCLs of 63.3-67.0 cm (x = 65.3 cm, SD = 1.2) coinciding closely with Shanker et al. (2003). Extensive review of the literature revealed that no data was present for immature males.

The following is a synopsis of the available literature for olive ridley SCL data. Work and Balazs (2002) necropsied seven olive ridley turtles from the HI DSLL fishery which all happened to be females. Their results include immature olive ridley sea turtles ranging from 43.7 to 54 cm SCL; one subadult which measured 57.5 cm SCL; and adults were greater than 62 cm SCL (Work and Balazs 2002). The authors identified and discussed the skewed sex ratios found in the data. Work and Balazs (2010) later determined 60 cm SCL was considered the cutoff for adults after necropsying 58 olive ridley sea turtles (45 female, 13 male). SCLs did not differ significantly amongst immature turtles and sex ratios were greater due to a larger sample size (Work and Balazs 2010). Additionally, Plotkin and Bernardo (2003) measured 48 female turtles and determined a size range of 58.5 to 69.5 cm SCL for adults, whereas Whiting et al. (2007) found a range of 63.8 cm to 70.8 cm SCL (n=12). Zug et al. (2006) on the other hand determined nesting females have a minimum SCL of 53 cm and range from 53-79 cm SCL. Finally, Shanker et al. (2003) assessed 2,110 mature females producing a mean SCL range of 64.3 cm to 67.1 cm over the same four year period as the males.

Figure 27 sourced from Zug et al. (2006) depicts the age estimation charts based on SCL measurements. Using this data, olive ridleys median age at sexual maturity would be approximately 13 years old with a SCL measurement of 60 cm (Zug et al. 2006). 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).

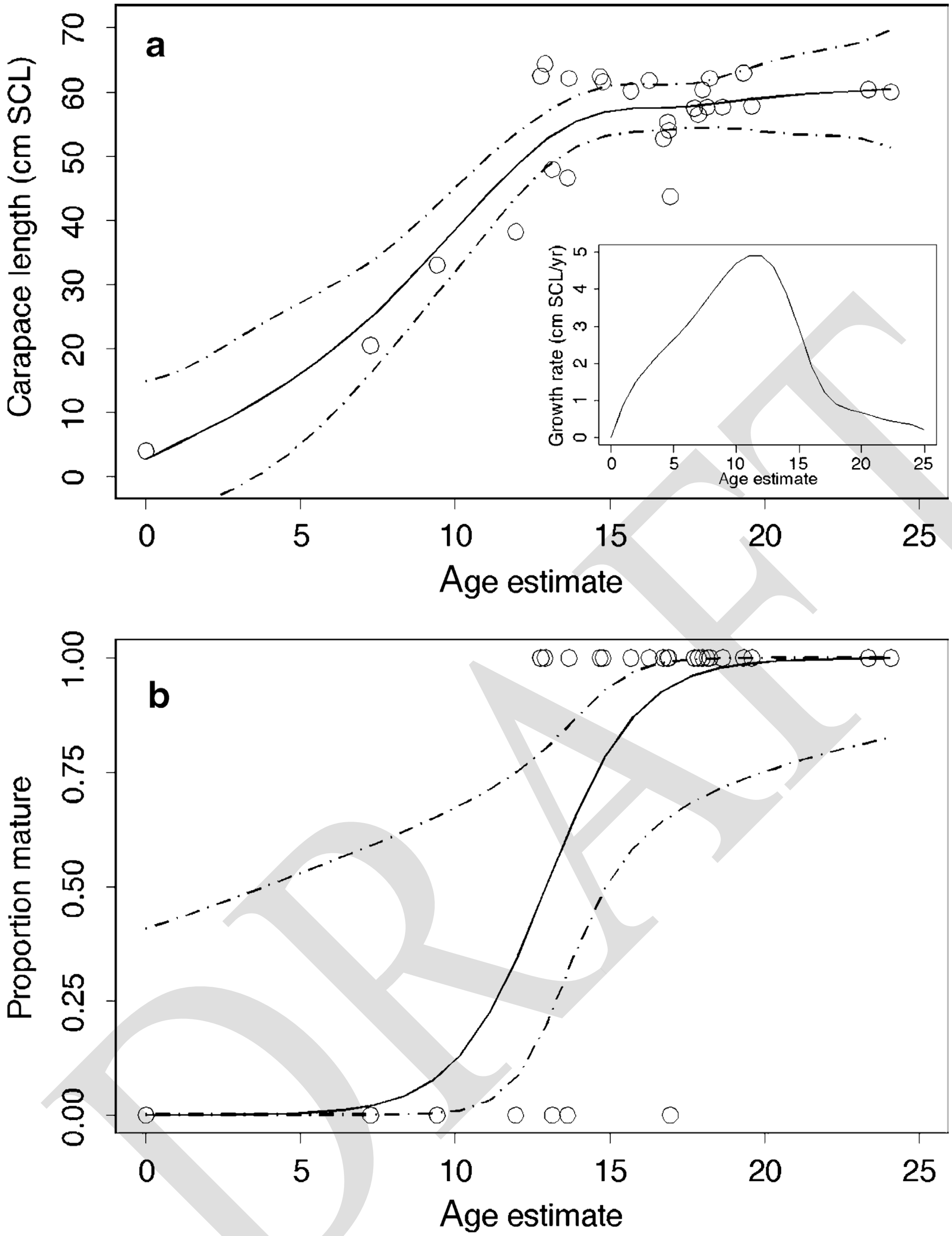

Figure 27: Panel (a) shows the expected size-at-age growth function (solid curve) fitted using a generalized smoothing spline model (Gu 2002) with 95% credible interval shown by the dashed curves. Age estimates shown by open dots and an estimate for expected olive rid- ley hatchling size is also included. Inset shows the age-specific growth rate function derived by numerical differentiation of the expected size-at-age function [solid curve in panel (a)]. Panel (b) shows the expected age-specific maturity function (solid curve) also fitted using a generalized smoothing spline model with 95% credible interval shown by the dashed curves, and open dots show the individual binary response variable (immature, mature; source: Zug et al. 2006).

Growth rate data for olive ridleys in the wild are unknown (Avens and Snover 2013; NMFS and FWS 1998). Female olive ridleys are believed to attain sexual maturity at an age similar to its congener, the Kemp's ridley (*Lepidochelys kempii*). Based on samples collected in the northcentral Pacific Ocean, Zug et al. (2006) estimated the median age of sexual maturity for the olive ridley is 13 years with a range of 10 to 18 years.

Individual olive ridleys 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). Olive ridleys commonly nest in successive years (Pritchard 1969; Cornelius 1986; Plotkin 1994), and the behavior may well be the norm for the species. In general, individual olive ridleys 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).

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). Smaller clutch sizes observed for solitary nesters might be due to energetic costs associated with undertaking internesting movements among multiple beaches (Plotkin and Bernardo 2003). Solitary nesters generally oviposit on 14-day cycles whereas arribada nesters oviposit approximately every 28 days (Pritchard 1969; Kalb and Owens 1994; Kalb 1999). However, this generality may not apply to all populations. Solitary nesters in Sergipe, Brazil, averaged 22.35 + 7.01-days internesting cycle (Matos et al. 2012). Within a nesting season, solitary nesters use multiple beaches for oviposition but arribada nesters display nest site fidelity (Kalb 1999). However, several studies indicate this, too, may not apply to all populations--some arribada nesters nest at different arribada beaches (Shanker et al. 2003; Fonseca et al. 2013), and some solitary nesters show strong site fidelity (Whiting et al. 2007). Tripathy and Pandav (2007) also found strong beach fidelity and determined remigration intervals varied between 1 and 8 years in Rushikulya, India. Pandav et al. (2000) previously found remigration intervals for males was 0.99 years (SD=0.06, range = 0.86 to 1.14 years, n = 28) and 1.01 years (SD=0.05, range = 0.96 to 1.09 years, n = 5) for females. Da Silva et al. (2007) found similar remigration results of between 1-6 years in Brazil, although the sample size was low (n= 5) and concluded more data was needed to establish these rates. Additionally, Whiting et al. (2007) satellite tagged 8 olive ridleys in Australian waters and found one turtle exhibited remigration fidelity to a nesting beach at an interval of approximately 1 year.

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 Hernández-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. Hernández -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 Hernández-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 (Castro 1986; Dornfeld and Paladino 2012; Gaos et al. 2006). Conversely, survivorship is low on high density arribada nesting beaches because of density-dependent mortality (Cornelius et al. 1991). The sheer number of turtles (1,000-500,000 turtles) nesting in spatially limited areas results in density dependent egg mortality during a single arribada. Moreover, turtles return approximately every month during a discrete nesting season (3-6 months) and nests that remained intact during the previous month are again at risk when new waves of turtles crawl ashore. For example, at La Escobilla, Mexico, approximately 6% of nests were destroyed in the first arribada, but increased to over 15% in the second arribada as nest density increased (Ocana et al. 2012). In addition to nest disturbance, the existence of high nest densities over time apparently alters the nutrient composition of sand, as well as the concentration of ammonia in the sand (McPherson and Kibler 2008). High ammonia concentrations, and/or high concentrations of fungal and bacterial pathogens, at beaches with high nest densities might also contribute to density-dependent nest loss. In controlled experiments at Playa La Flor, Nicaragua, and Playa Nancite and Ostional, Costa Rica, nest density affected hatching success with higher density resulting in lower hatching success (Bézy et al. 2013; Honarvar 2007; Honarvar et al. 2008). As nest density increased, gas exchange became limited during the latter part of the incubation period, likely due to the increased metabolic activity from developing embryos. CO2 levels increased and O2 levels decreased in higher density plots, which led to higher embryo death (Honarvar 2007; Honarvar et al. 2008). Bacterial (Honarvar et al. 2011) and small organism (Madden et al. 2008) diversity and richness were also greater in areas of high nest density and close to vegetation and away from tidal wash. During high-density arribadas, nesting females inadvertently break eggs, which provide nutrients for increased bacterial growth. Also, the high zone on the beach is less likely to be exposed to tidal overwash and accumulation of broken eggs in this area over time may contribute to bacterial diversity and richness (Honarvar et al. 2011).

4.2.4.3 Status

Mexico’s Pacific Coast Population - Endangered

The endangered olive ridley sea turtle thought to be increasing in many areas. The endangered population of olive ridley sea turtles nest primarily in large arribadas on the west coasts of Mexico with some solitary nesting throughout the region. Since reduction or cessation of egg and turtle harvest in Mexico in the early 1990s, annual nest totals have increased substantially, but have not returned to their pre-1960s abundance estimates. On the Mexican coast, three populations appear stable, two are increasing (Ixtapilla and La Excobilla), and one decreasing, with over one million nests laid annually (Table 21) (NMFS and FWS 2014).

Eguchi et al. (2007) estimated the density and abundance of the olive ridley sea turtle from shipboard line-transects which resulted in an estimate of 1,150,000 – 1,620,000 turtles in the eastern tropical Pacific in 1998-2006. During 2010, vessel surveys from the coast to 185 km offshore of the Mexican Central Pacific (Jalisco, Colima, and Michoacan waters) covered 3,506 km and recorded 749 sightings (Martín del Campo et al. 2014). The weighted average of the three periods (winter, spring, and autumn 2010) of olive ridley sea turtles was 177,617 (CI: 150,762-204,471, CV: 17.2%, 95%), with the highest abundance recorded in winter in the oceanic region of Jalisco (N: 181,150, CI: 117,150-280,110, CV:21.4%). Martín del Campo et al. (2014) conclude that olive ridley sea turtles are abundant in coastal and oceanic waters of the Mexican Central Pacific and their numbers are probably still increasing as a result of the protection programs that began in the 1990s.

Table 21. Endangered populations of olive ridley arribada and solitary nesting beaches in Mexico, and estimates of annual abundance at each site and current trends. Table has been adapted from Table 1 in NMFS and FWS 2014.

Location	Annual Number	Trend
ARRIBADA		
La Escobilla	1,013,034 females	increasing
Mismaloya	2,328 nests	stable
Ixtapilla	2,900 – 10,000 nests	increasing
Moro Ayuta	10,000 – 100,000 nests	stable
Tlacoyunque	608 nests	stable
Chacahua	2,042 nests	decreasing
SOLITARY		
El Verde	1,160 nests	stable
Platanitos	1,301 nests	increasing
Cuyutlán	1,257 nests	increasing
Maruata-Colola	4,198 nests	stable
Puerto Arista	707 nests	stable
Moro Ayuta	no estimate available	stable

Nuevo Vallarta	4,900 nests	unknown
San Cristobal	89 nests	unknown
El Suspiro	220 nests	unknown

Threatened Olive Ridley Sea Turtles of the Eastern Pacific

Threatened olive ridley sea turtles nest south of Mexico to Colombia. Within this range lie several beaches where arribadas reportedly occurred in the past but no longer do, as well as beaches where they still occur: five in Nicaragua, two in Costa Rica, and one in Panama. Current estimates for some of the beaches are either unavailable or are based on sporadic nesting beach surveys. In Costa Rica, the Ostional nesting assemblage is one of the largest in the world, second only to La Escobilla, Mexico (Valverde et al. 2012). As with other arribadas, a large variability in the magnitude of mass nesting events in Costa Rica can occur, with arribadas at Ostional ranging between 3,564 and 476,550 egg-laying females during the period 2006–2010 (Table 22) (Valverde et al. 2012). Valverde et al. (2012) estimated the nesting population size by dividing the estimated arribada abundance totals by estimated olive ridley nesting frequency of 2.21 (Van Buskirk and Crowder 1994 in Valverde et al. 2012). NMFS and U.S. FWS (2014) estimate that females may lay two clutches on average per arribada nesting season, with approximately 100-110 eggs laid per clutch. However, Ballestero et al. (2000) utilized a fixed quadrant method (vs. line transects) to estimate that the nesting population was approximately 588,500 fluctuating between 232,318 and 1,147,969 turtles per arribada between 1988 and 1997. If these estimates are correct, Valverde et al. (2012) concludes that the Ostional assemblage has decreased in abundance over the past two decades likely as a result of low hatching rates. In contrast to solitary nesting beaches, survivorship is low on high density arribada nesting beaches because of density-dependent mortality (NMFS and U.S. FWS 2014). This density-dependent effect negatively impacts nesting populations because in addition to nest disturbance and egg mortality, high nesting density alters the nutrient composition of sand, gas exchange, and ammonia concentration in the sand which results in high concentrations of fungal and bacterial pathogens resulting in lower hatch success thus affecting population growth (NMFS and FWS 2014).

Table 22. Threatened olive ridley arribada and solitary nesting beaches in the Eastern Pacific and estimates of abundance expressed as arribada size, nests, or females at each site and trends. Table has been adapted from Table in NMFS and FWS 2014.

Location	Annual Number	Trend
ARRIBADA		
Nancite, Costa Rica	256- 41,149 females	decreasing
Ostional, Costa Rica	3,564 to 476,550 females	increasing but declining recently
Nancite, Costa Rica	256-41,149 turtles per arribada	decreasing
Chacocente, Nicaragua	27,947 females	unknown
La Flor, Nicaragua	521,440 females	stable
Isla Canas, Panama	8,768 females	decreasing
SOLITARY		
Hawaii Beach, Guatemala	1,004 females	decreasing

Threatened Olive Ridley Sea Turtles of the Western Pacific

In the Western Pacific, olive ridley sea turtles are solitary nesters and typically occur in tropical and warm temperate waters from Australia through Southeast Asia (NMFS and FWS 2014). 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 (Table 23) (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 2008). Data are not available to analyze trends (NMFS 2005; NMFS and FWS 2014).

Table 23. Threatened olive ridley arribada and solitary nesting beaches in the Western Pacific and estimates of abundance expressed as arribada size, nests, or females at each site and trends. Table has been adapted from Table in NMFS and FWS 2014.

Location	**Annual Number**	**Trend**
ARRIBADA		
Gahirmatha, India	150 - 250,000 females	stable
SOLITARY		
Australia	3000 females	unknown
Alas Purwo, Indonesia	250 females	increasing
Terengganu, Malaysia	10 nest	decreasing

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). On Jamursba-Medi beach, 77 olive ridley 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). Extensive hunting and egg collection, in addition to rapid rural and urban development, have reduced nesting activities in Indonesia. In eastern Java, olive ridley 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 nesting is scattered throughout northern Australia, with a few thousand females nesting annually (Limpus 2008). The breeding population in northern Australia may be the largest population remaining in the Western Pacific region, although a full evaluation of their distribution and abundance is needed (Limpus 2008; NMFS and FWS 2014). 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 2008).

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 2008). 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 2008).

4.2.4.4 Threats to the Species

Natural Threats

Fibropapillomatosis is the only disease that has been found in olive ridley sea turtles. It is believed to occur at low levels (NMFS and U.S. FWS 2014). Beetles, reptiles, invertebrates, and birds, all prey on eggs and hatchlings on land. Sharks, billfish, mahi mahi, whales, and birds prey on adults and hatchlings in the ocean (Frazier et al. 1994, 1995; Pitman and Dutton 2004; Villasnor et al. 2010).

Anthropogenic Threats

Major anthropogenic threats to both threatened and endangered populations include impacts to nesting beaches resulting from development, direct harvest, and fishing bycatch, which are briefly described below. Climate change and marine debris may also be a growing threat to this species, as it is for other sea turtle species and is discussed below.

Impacts to nesting habitat and habitat loss resulting from development, construction, beach armoring, sea level rise, human encroachment, lighting pollution, etc. on the breeding populations in Mexico are lacking, although human-induced habitat impacts are expected to increase as Mexico's population expands and tourism increases (NMFS and FWS 2014). 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 (NMFS and FWS 2014).

Ostional beach in northwest Costa Rica is an arribada rookery that supports a large mass-nesting assemblage along with a legal community-based egg-harvest program (Campbell 1998, Campbell et al. 2007). The rationale that supports the Ostional egg harvest is based on data that showed a significant number of clutches are destroyed during arribadas by nesting turtles, that the hatching rate at this beach is very low, and that legalizing the harvest may help to limit the previously uncontrolled illegal take of eggs (Alvarado-Ulloa 1990 and Cornelius et al. 1991 in Valverde et al. 2012). The egg harvest functions much as it was suggested by the scientific community: the associates are allowed to harvest eggs for the first 2.5 days of each arribada (the first 2 days for commercialization and the last half a day for local consumption), while keeping the beach clean and reducing the impact of feral predators (Ordonez et al. 1994 in Valverde et al. 2012). Between 2006 and 2010, Valverde et al. (2012) estimated the mean egg harvest was 4,746 eggs, ranging between 1,527 to 8,138 total clutches. The estimated mean of clutches harvested was 21.2% (Valverde et al. 2012). It is not clear whether the Ostional arribadas underwent a significant change in abundance during the study period, and the number of years covered is too short to establish a long-term trend, however the population appears to have declined when compared with historical data given that the population appears to be suffering from low hatch success (18%), high clutch destruction rates, and low recruitment (Valverde et al. 2012).

A major threat to olive ridley sea turtles is bycatch in fisheries, including longline, drift gillnet, set gillnet, bottom trawling, dredge, and trap net fisheries that are operated either on the high seas or in coastal areas throughout the species' range. Fisheries operating near arribadas can take tens of thousands of adults as they congregate. For example, trawl and gillnet fisheries off the east coast of India drown so many olive ridley sea turtles that tens of thousands of dead adults wash up on the coast annually (NMFS and FWS 2014). Just recently, 600 Olive Ridleys washed up dead, from suspected trawler operations, at the Orissa rookery (Orissa Post 2019).

In the Eastern Pacific, fishery interactions are a major threat to the species, primarily because of development of a shrimp trawl fishery along the Pacific coasts of Central America starting in the 1950s, which is thought to kill tens of thousands of olive ridley sea turtles annually (NMFS and FWS 2014). Trawlers in Costa Rica are reported to catch over 15,000 sea turtles annually, and 90% of those are olive ridley sea turtles (Arauz et al. 1998). As a result of litigation brought about by six environmental NGOs, trawl fishing was banned in Costa Rica in September 2013 (Arias 2013). In addition, the growth in longline fisheries in the region over recent years represents a growing bycatch threat to the species, with the potential to interact with hundreds of thousands of turtles annually (Frazier et al. 2007, Dapp et al. 2013). From 1999 to 2010, an observer program collected data to assess the impact of the Costa Rican longline fishery and documented an estimated 699,600 olive ridley sea turtles caught, including 92,300 adult females and an additional 23,000 green turtles (Dapp et al. 2013). Artisanal gillnet and longline fisheries of Peru and Chile are known to interact with olive ridley sea turtles (Alfaro-Shigueto et al. 2011, Donoso and Dutton, 2010). Small scale fisheries operating in Peru using bottom set nets, driftnets, and longline fisheries were observed between 2000 and 2007. Approximately 6,000 sea turtles were captured annually; 240 were olive ridley sea turtles (Alfaro-Shigueto et al. 2011). Threats to olive ridley sea turtles in Australia include high bycatch in gillnet and trawl fisheries, ghost net entanglement, egg loss due to pig and dog predation, and significant egg harvest as a result of Indigenous practices (Limpus 2008).

Between 2005 and 2017 there were 610 olive ridley interactions in the HI DSLL fishery and from this the estimated mortality is 579 (NMFS 2018XX). 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 HI SSLL fishery rarely interacts with olive ridley sea turtles and since 2004, they have caught 10 and all were released alive, with one estimated mortality (NMFS 2018XX). 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).

As with the other species discussed above, no significant climate change-related impacts to olive ridley turtle populations have been observed to date. However, over the long-term, climate change-related impacts will likely influence biological trajectories in the future on a century scale (Paremsan and Yohe 2003). Only limited data are available on past trends and current scientific methods are not able to reliably predict the future magnitude of climate change and associated impacts or the adaptive capacity of this species. However, olive ridley sea turtles in the east Pacific Ocean are highly migratory, 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 2010 in NMFS and FWS 2014). As with leatherback turtles nesting in the Eastern Pacific, olive ridleys 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).

Marine debris is also a source of concern for olive ridley sea turtles. Olive ridley sea turtles can ingest small debris and larger debris can entangle animals leading to death. For olive ridley sea turtles the greatest risk is when they are in the pelagic environment but there are no data to quantify what the impacts are.

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 28).

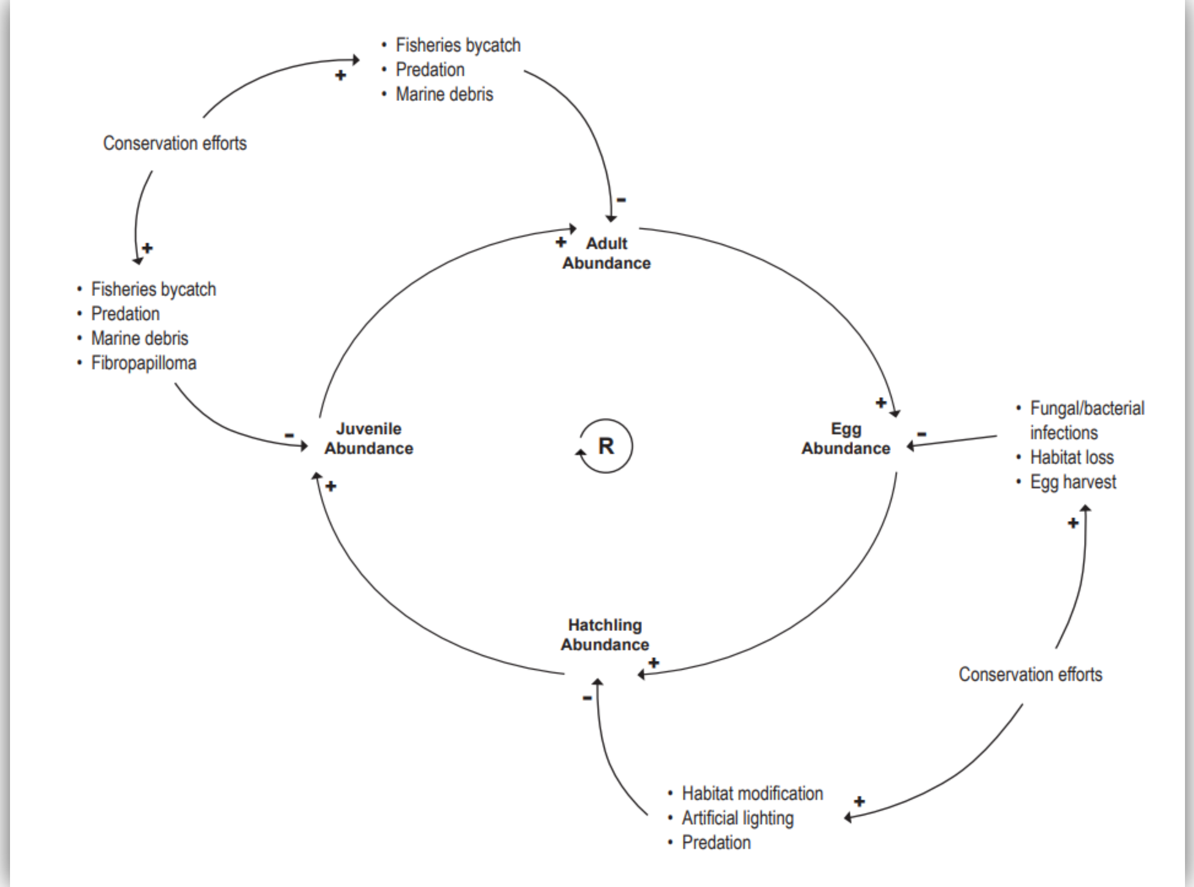

Figure 28. Olive ridley sea turtle causal loop diagram depicting various stressor-response relationships for each life cycle stage.

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. 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 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.

Our graphic illustrates the population behavior of olive ridley sea turtles (females), which is based on available nest count data. Nesting females serve as a surrogate for all of the life stages in the population due to the limited data availability of all other life stages and for males. Natural threats occur to the global population as a whole and therefore can potentially be relevant to the olive ridley sea turtle. 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 slightly positive.

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.

4.2.5 Oceanic Whitetip Shark

4.2.5.1 Distribution

Oceanic whitetip sharks *(Carcharhinus longimanus)* are worldwide circumtropical and subtropical pelagic apex predators found primarily between latitudes 30° North (N) and 35° South (S) (Compagno 1984; Baum et al. 2015; Young et al. 2016). Although, the species has been reported as far as 45°N and 40°S in the Western Atlantic (Lessa et al. 1999). A geographical representation of the species range was provided by Last and Stevens (Figure 29) (2009).

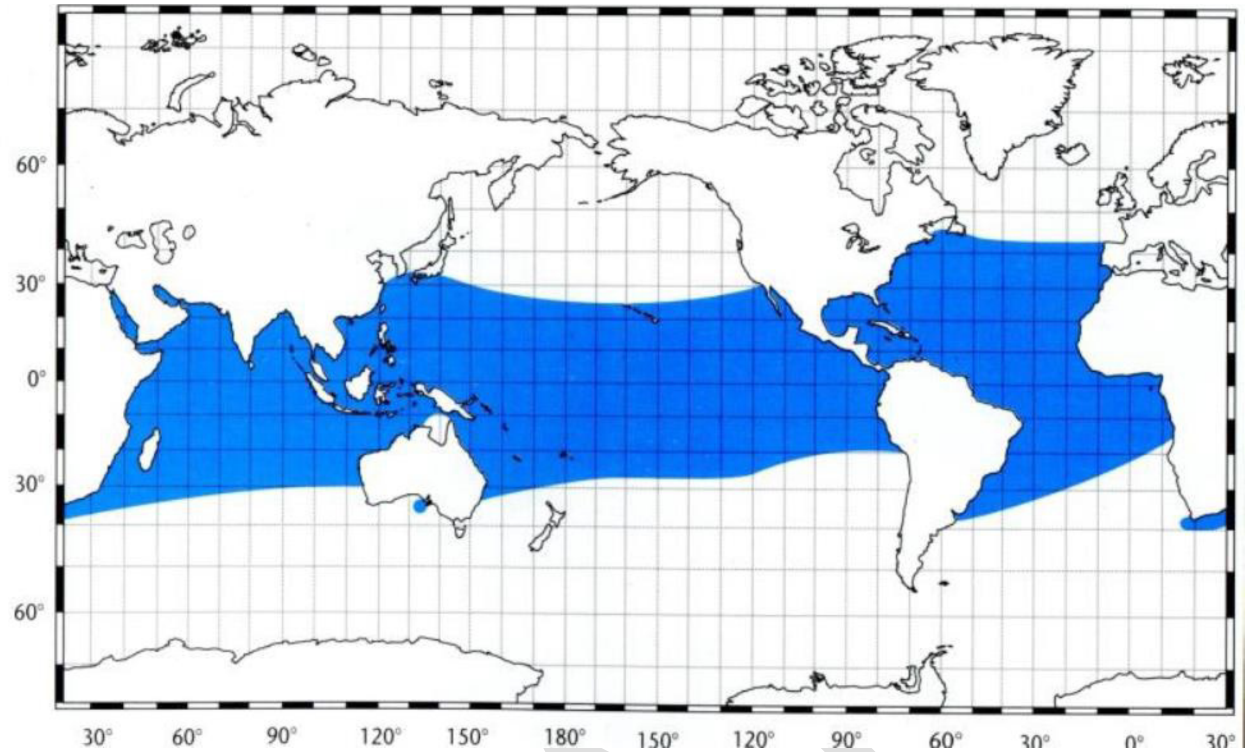

Figure 29. Geographical distribution of the oceanic whitetip shark (Last and Stevens 2009; Young et al. 2016).

Abundance of oceanic whitetips appears to be the greatest in pelagic waters 10° on either side of the equator with decreased concentrations as the distance from the equator increases and with increasing proximity to various continental shelves (Backus et al. 1956; Strasburg 1958; Compagno 1984; Nakato et al. 1997; Bonfil et al. 2008; Clarke et al. 2011; Hall and Roman 2013; Tolotti et al. 2013; Young et al. 2016).

Thermal preferences by oceanic whitetips suggest inter-ocean basin movements, such as around the southern tip of Africa or South America, are restricted due to thermal barriers (Bonfil et al. 2008; Musyl et al. 2011; Howey-Jordan et al. 2013; Gaither et al. 2015; Young et al. 2016). Oceanic whitetips are typically found in epipelagic waters between 15ºC and 28ºC with strong preferences to warmer surface layers greater than 20ºC (Howey-Jordan et al. 2013; Howey et al. 2016; Young et al. 2016; Andrezejaczek et al. 2018). Deep dives (>200m) through the thermocline into the mesopelagic zone have been documented by Howey-Jordan et al. (2013) and Howey et al. (2016) into waters as cold as 7.75 ºC for brief periods, most likely to forage (Young et al. 2016). Although Musyl et al. (2011), Tolotti et al. (2015), and Carlson and Gulak (2012), determined that exposure to such temperatures are not continuous with 95% of their time greater than 120 meters (m) in depth, above the thermocline (Young et al. 2016).

While little is known about oceanic whitetip migration patterns, several studies have provided valuable insights which are discussed in detail by Young et al. (2016). However, as a general overview, Musyl et al. (2011) showed complex movement patterns generally restricted to central tropical waters north of the North Equatorial Countercurrent (NEC) in the Pacific (Young et al. 2016). Whereas the NMFS Cooperative Shark Tagging Program (CSTP) studies in the Atlantic have discovered movements by juveniles ranging from the Lesser Antilles west into the central

Caribbean Sea, from east to west along the equator, from the northeastern Gulf of Mexico to the Atlantic Coast of Florida, from the Mid-Atlantic Bight to southern Cuba, and northeast tracks from southern Brazil and are represented in Figure 30 below (Kohler et al. 1998; Bonfil et al. 2008; Young et al. 2016).

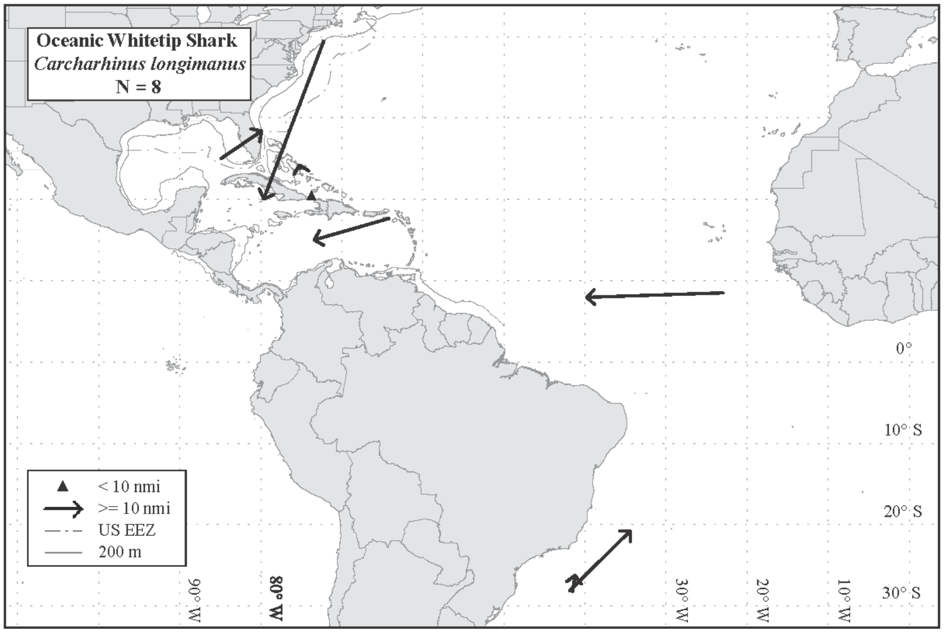

Figure 30. Movement distributions of oceanic whitetip sharks from mark recapture studies from the NMFS Co-operative Shark Tagging Program during 1962-93, NMFS unpublished data, and Young et al. 2016.

In the equatorial and southwestern Atlantic, fisheries independent data exhibited oceanic whitetip sharks which were tagged with pop-up satellite tags (PSATs) in the operational range of the Brazilian longline fleet, exhibit some degree of site fidelity, even after traveling several thousand kilometers (Tolotti et al. 2015; Young et al. 2016). Similarly, eleven mature oceanic whitetip sharks were tagged in the Bahamas and these individuals remained within 500 km of the tagging site for approximately 30 days before dispersing across 16,422 km^2 of the western North Atlantic, and subsequently returning to the Bahamas after 150 days (Howey-Jordan et al. 2013; Young et al. 2016). Additionally, Carlson and Gulak (2012) satellite tagged an oceanic whitetip shark nearby in the Gulf of Mexico which moved a straight-line distance of approximately 238 km from waters off southeast Louisiana to the edge of the continental shelf about 300 km north of the Yucatan Peninsula (Young et al. 2016).

Meanwhile observations from the Spanish longline fishery targeting swordfish from 1993-2011 in the Indian Ocean, indicate that the distribution of oceanic whitetips is primarily North of 25ºS, most likely influenced by the seasonal expansion or displacement of warm water masses (García-Cortes et al. 2012; Young et al. 2016). The distribution illustrated by Garcia-Cortes et al. (2012) are highly influenced by the effort of the fleet as the data is related to total catches rather than catch per unit effort (CPUE) (Young et al. 2016).

Vertical and horizontal behavior of oceanic whitetip sharks were studied by Filmalter et al. (2012) in the western Indian Ocean also using PSATs and mini-PSATs. The study results displayed the ability of these sharks to travel great distances in the pelagic environment (Young et al. 2016). Finally, 56 oceanic whitetip sharks were opportunistically tagged by the Spanish fishing fleet from 1985-2004, which discovered these sharks exhibit a trans-equatorial migration in the Indian Ocean (Mejuto et al. 2005; Young et al. 2016).

Feeding and Diving Behavior

Oceanic whitetip sharks are apex pelagic predators feeding primarily on cephalopods and numerous teleost species (Backus et al. 1956; Bullis 1961; Bonfil et al. 2008), while various studies have also reported depredation of sea birds, marine mammals, other sharks, rays, molluscs, crustaceans, and even garbage (Compagno 1984; Cortes 1999; Young et al. 2016). The evidence suggests these sharks are opportunistic in nature. Furthermore, aggregations of oceanic whitetips have been documented in the Bahamas due to the availability of large teleost prey like billfish or mahi, and due to supplemental feeding from recreational sport fishermen within the region causing alterations to the species distribution patterns by increasing local site fidelity (Madigan et al. 2015; Young et al. 2016). Cortes (1999) scored oceanic whitetips with a high trophic level with a score of with a score of 4.2 out of a maximum 5.0 based on the diets species composition (Young et al. 2016).

These sharks are can be found to a depth of approximately 150 m, with brief deep dives into the meso- and bathypelagic zones (>200 m) but are more well known to exhibit strong preferences for surface mixed layers where temperatures typically remain greater than 20°C (Howey-Jordan et al. 2013; Howey et al. 2016; Young et al. 2016; Tolotti et al. 2017). The maximum recorded dive of the species was to a depth of 1,082 m (Howey-Jordan et al. 2013). Similar studies have been conducted elsewhere providing supporting documentation which indicates the species may exhibit possible behavioral thermoregulation. Most shark species, like the oceanic whitetip shark, are poikilothermic, or more commonly referred to as being cold-blooded. This means their body temperature is regulated by the surrounding water temperatures. Various authors have studied vertical and horizontal movements in various oceans. Most studies show oceanic whitetip sharks spend the majority of their time in surface mixed waters within a 2°C gradient where sea surface temperatures were greater than 25°C, rarely venture less than 200 m in depth, although the maximum recorded depth was 256 m, and spend most of their time above the thermocline (Musyl et al. 2011; Carlson and Gulak 2012; Filtmater et al. 2012; Howey-Jordan et al. 2013; Tolotti et al. 2015; Howey et al. 2016; Young et al. 2016).

4.2.5.2 Population Structure

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 (Young et al. 2016). Camargo et al. *(*2016) compared the mitochondrial control region in 215 individuals from the Atlantic and Indian Oceans which identified 12 haplotypes (Figure 31) (Young et al. 2016). Of these individuals, 129 shared one haplotype while two additional haplotypes were found in all regions, with another two haplotypes found in just the eastern and western Atlantic Ocean populations and an additional seven haplotypes in one or two individuals (Carmargo et al. 2016; Young et al. 2016). Significant genetic differentiation was discovered between the eastern and western Atlantic Ocean based on haplotype frequencies

(ΦST = 0.1039, P <0.001) although some were not (FST = 0.02 – 0.03, P > 0.01), even after excluding populations comprised of less than 10 individuals and through pairwise comparisons among populations within the regions (Camargo et al. 2016; Young et al. 2016). Furthermore, Indian Ocean sample sizes may be too small to differentiate statistically significant genetic structure within the region and compared to others (Camargo et al. 2016; Young et al. 2016). Lastly, it should be noted that this study only used mitochondrial markers, meaning male-mediated gene flow is not reflected (Young et al. 2016) although other species in the Carcharhinus genus are known to exhibit male-mediated gene flow between populations (Portnoy et al. 2010).

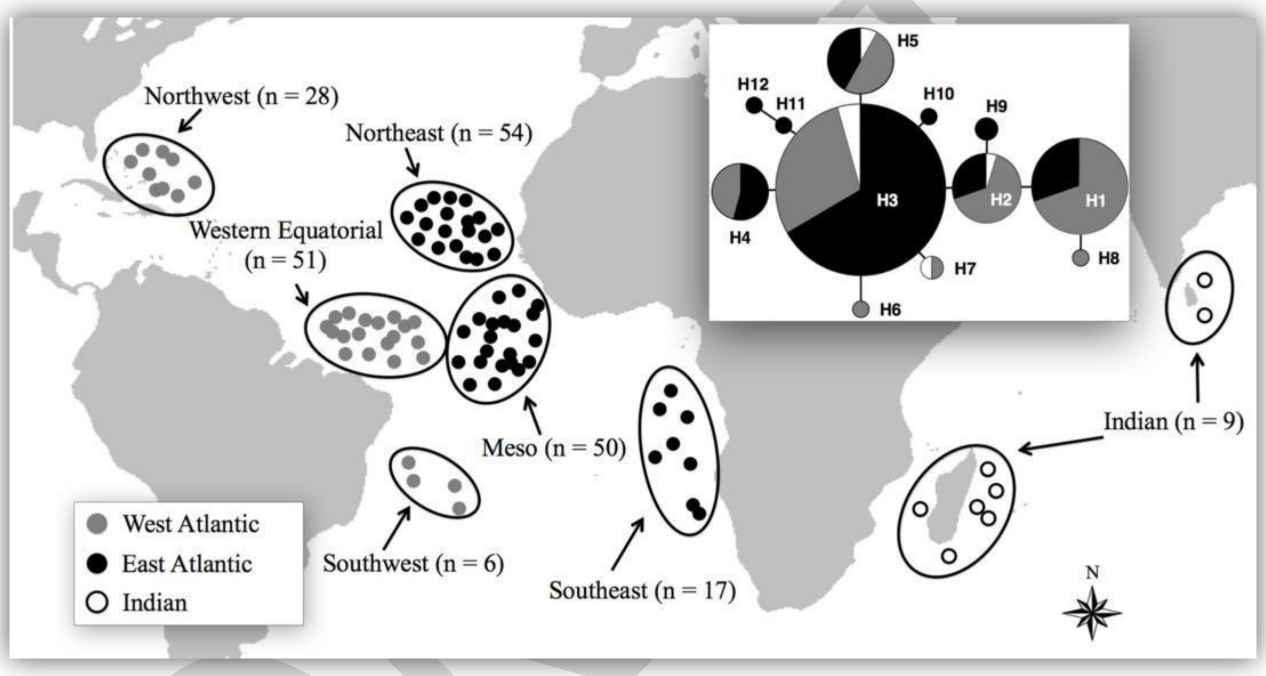

Figure 31. Geographic distribution of C. longimanus samples relaying analyzed network haplotypes and compilation of the mitochondrial DNA control regions. Source: Camargo et al. 2016; Young et al. 2016.

Meanwhile, 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 (Young et al. 2016). No fine-scale matrilineal structure was discovered within ocean basins using three population-level pairwise metrics (PhiST, FST, and Jost's D) (Ruck 2016; Young et al. 2016).

Ruck (unpublished data) then compared genetic samples of the two studies and detected significant maternal population structure within the western Atlantic which provides evidence of three matrilineal lineages in the western Atlantic (Young et al. 2016). Primarily, samples from the western Atlantic showed significant differentiation from Northwest Atlantic samples (ΦST Range: 0.058 – 0.078, FST Range: 0.063 – 0.078 (P ≤ 0.02) (Young et al. 2016). Whereas globally, clear phylogeographic partitioning of haplotypes was not detected and the most common mitochondrial haplotypes were shared by individuals in the Atlantic, Indian, and Pacific Oceans (Young et al. 2016). Weak but significant differentiation was detected between western Atlantic and Indo-Pacific Ocean populations using mitochondrial and nuclear analyses ($\Phi ST = 0.076$, $P = 0.0002$; $FST = 0.017$, $P < 0.05$ after correction for False Discovery Rate) (Ruck unpublished data; Young et al. 2016). Significant inter-basin population structure was apparent (Figure 32); however it was associated with evidence of contemporary migration between the western Atlantic and Indo-Pacific Oceans and deep phylogeographic mixing of mitochondrial haplotypes (Young et al. 2016).

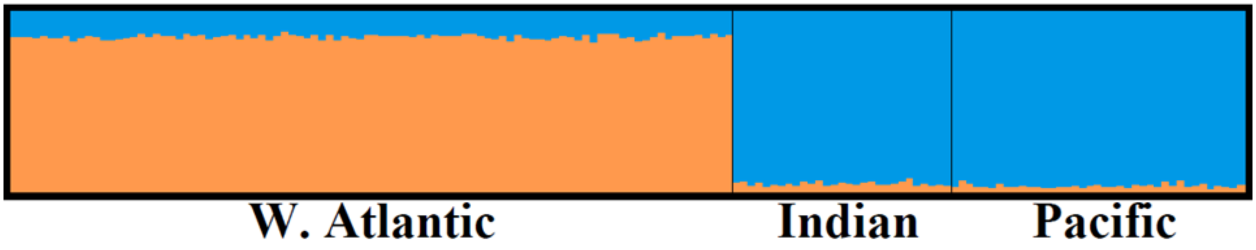

Figure 32. DISTRUCT plots summarizing STRUCTURE results of all genotyped samples: K = 2. Which clearly indicates strong sorting of two clusters between the Western Atlantic and the Indo-Pacific. Sources: Ruck 2016; Young et al. 2016.

Ruck (2016) suggests that oceanic whitetip sharks are obstructed by semi-permeable thermal barriers to inter-ocean movements, particularly the Benguela upwelling system around the tip of South Africa, which separates the western Atlantic and the Indo-Pacific into distinct genetic populations (Young et al. 2016).

Population structure within various ocean basins could also be influenced by Philopatry as current telemetry tracking studies indicate (Musyl et al. 2011; Howey-Jordan et al. 2013; Tolotti et al. 2015), although the sample sizes may not be reflective of the whole population (Ruck 2016; Young et al. 2016). Recent tagging studies also suggest that while oceanic whitetips are highly migratory they appear to display high levels of philopatry, most likely due to females remaining in certain locations to give birth and thus segregation from other regional populations occurs (Howey-Jordan et al. 2013; Tolotti et al. 2015; Camargo et al. 2016; Young et al. 2016). The Atlantic on the other hand does not provide a physical barrier between trans-Atlantic routes but appears genetic differentiation to be influenced by behavioral component, at least in female sharks (Camargo et al. 2016; Young et al. 2016). As mentioned previously, male-mediated gene flow is not reflected in the study but relied on mitochondrial DNA (mtDNA).

In both studies, genetic diversity of the oceanic whitetip shark appears to be small and ranks the fourth lowest in global mitochondrial control region sequence (mtCR) genetic diversity (0.33% ± 0.19%) when compared to various other species (Young et al. 2016). Carmago et al. (2016) noted that genetic variability was significantly lower in populations from the eastern Atlantic when compared to the populations in the western Atlantic (34.2% and 36.9%, respectively) and determined that this may pose a risk to the ability of the species to adapt to changes within the environment based on the haplotype and nucleotide diversity which was compared to various other circumtropical species (Testerman 2014; Bernard et al. 2015; Clarke et al. 2015a; Hoelzel et al. 2006; Castro et al. 2007; Carmago et al. 2016; Ruck 2016; Young et al. 2016).

Overall, the data does not show male-mediated gene flow within the Atlantic but relies solely on mtDNA to determine population structure (Young et al. 2016). Young et al. (2016) also believes that information regarding male-mediated gene flow would improve the fine-scale genetic structuring of oceanic whitetips in the Atlantic. While data from these studies are preliminary, three maternal populations are currently supported within the Atlantic and data analyses support at least two global genetic stocks using mitochondrial DNA and nuclear microsatellite (Young et al. 2016).

Diversity

Thus far, oceanic whitetip shark life history parameters have only been studied in the Southwest Atlantic Ocean (Lessa et al. 1999) and the North Pacific (Seki et al. 1998; Joung et al. 2016). The species has confirmed maximum ages of 12 and 13 years in the North Pacific and South Atlantic, respectively (Seki et al. 1998; Lessa et al. 1999) with an estimated maximum age of 17 years (Young et al. 2016). However, other information from the South Atlantic suggests the species likely lives up to 20 years old based on observed vertebral ring counts (Rodrigues et al. 2015). In the 1940s, the maximum length effectively measured for oceanic whitetip was 350 cm total length (TL) (Bigelow and Schroder 1948 cited in Lessa et al. 1999), with “gigantic individuals” perhaps reaching 395 cm total length (TL) (Compagno 1984), though Compagno’s length seems to have never been measured (Lessa et al. 1999). Lessa et al. (1999) noted that the length composition of the species may have been altered since the 1940s as a result of fishing pressure and evidenced by a scarcity of specimens larger than 270 cm TL. Lessa et al. (1999) recorded a maximum size of 250 cm TL in the Southwest Atlantic, and estimated a theoretical maximum size of 325 cm TL (Lessa et al. 1999); however, the most common sizes are below 300 cm TL (Compagno 1984). In terms of growth rate, early studies suggested that the oceanic whitetip shark is slow growing, but more recent studies have shown faster growth rates comparable to blue and silky sharks (Clarke et al. 2015b). In the Southwest Atlantic, male and female growth rates are similar; observed and back-calculated length at age von Bertalanffy parameters from Lessa et al. (1999) were as follows:

> Observed asymptotic length (L_∞) = 284.9 cm; growth coefficient (K) = 0.099 yr^{-1}, and T_0 = - 3.391 yr^{-1}
>
> Back-calculated asymptotic length (L_∞) = 325.4 cm; growth coefficient (K) = 0.075 yr^{-1}, and T_0 = -3.342 yr^{-1}

Growth rates are 25.2 cm yr^{-1} in the first free-living year; 13.6 cm yr^{-1} from ages 1 to 4; 9.7 cm yr^{-1} for adolescents of age 5; and 9.10 cm yr^{-1} for mature individuals (Lessa et al. 1999). In a more recent study from the western North Pacific (Joung et al. 2016), growth rates were also found to be similar between sexes. The von Bertalanffy growth parameters combining both sexes were as follows:

> Asymptotic length (L_∞) = 309.4 cm TL; growth coefficient (K) = 0.0852 yr^{-1}
>
> According to Branstetter (1990), growth coefficients (K) falling in the range of 0.05-0.10/yr is a slow-growing species; 0.1-0.2 is a moderate-growing species; and 0.2-0.5 is a fast-growing species. Under these parameters, the oceanic whitetip shark is considered a

slow-growing species. Figure 33 below shows the various growth curves for the oceanic whitetip shark.

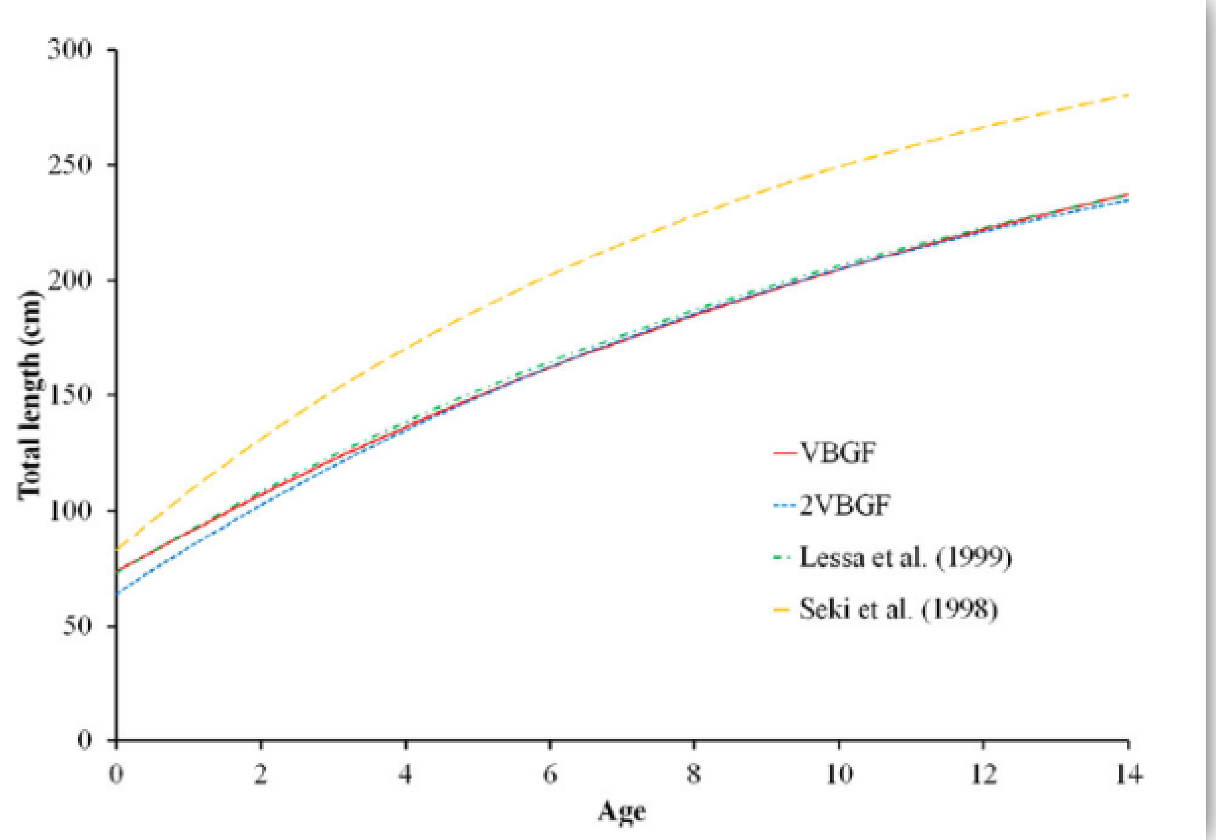

Figure 33. Comparison of the growth curves of the oceanic whitetip shark in different regions, from Seki et al. (1998), Lessa et al. (1999) and the current study. VBGF = von Bertalanffy growth function. 2VBGF was only used in the present study and VBGF was used in the other studies. Source: Joung et al. 2016; Young et al. 2016.

Since the status review was completed in 2016, an additional study compared von Bertalanffy growth curve values found in Figure 34 for oceanic whitetip sharks in Papua New Guinea and delineated curves for both genders which depicted slightly depressed curves (D'Alberto et al. 2017). The results were as follows:

Back calculated asymptotic length (L_∞) = 315.6 cm; growth coefficient (K) = 0.059 yr^{-1}.

A length-weight equation is provided by Romanov and Romanova (2009) (Figure 34) for total weight (TW): TW= (.386e-4)*FL^((2.75586)) (n = 587; both sexes).

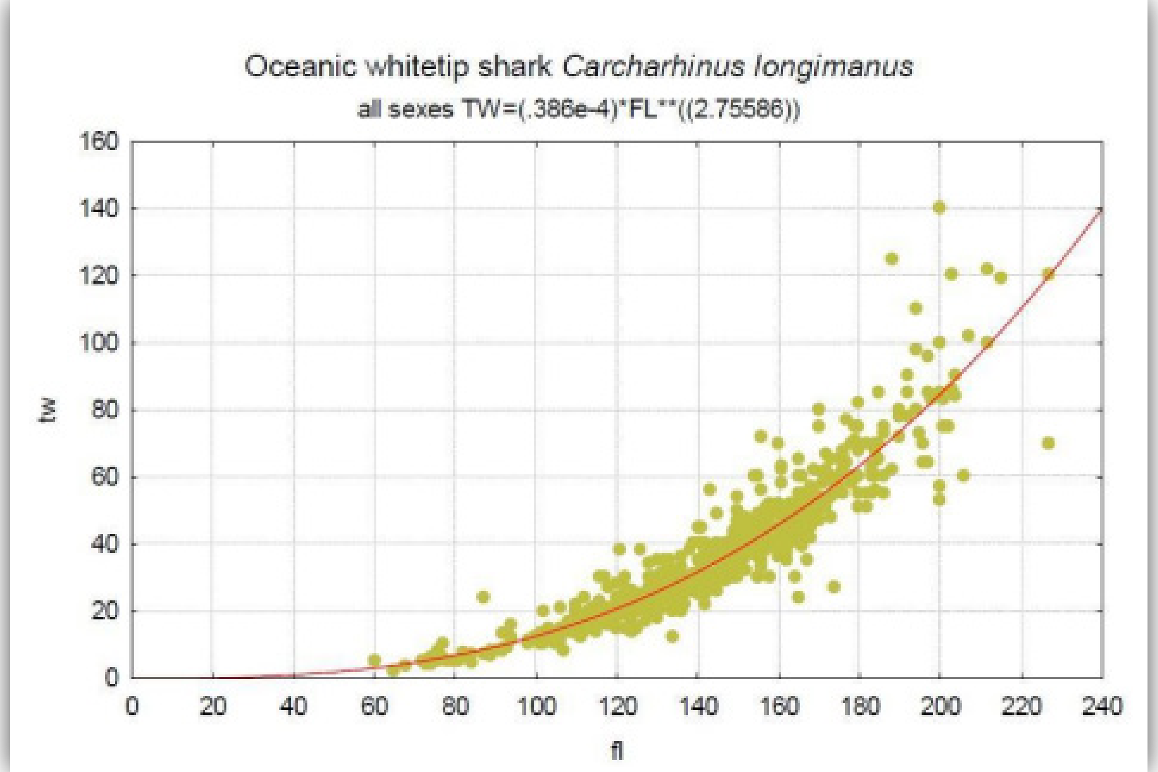

Figure 34. Length-weight scatterplot and relationship for oceanic whitetip shark (all sexes, n=587). Source: Romanov and Romanova 2009; Young et al. 2016.

Age of maturity is slightly different depending on location. For example, in the Southwest Atlantic, age and size of maturity in oceanic whitetips was estimated to be 6-7 years and 180-190 cm TL, respectively, for both sexes (Lessa et al. 1999). In the North Pacific, females become mature at about 168-196 cm TL, and males at 175-189 cm TL, which corresponds to an age of 4 and 5 years, respectively (Seki et al. 1998). However, more recently Joung et al. (2016) determined a later age of maturity in the North Pacific of approximately 8.5-8.8 years for females and 6.8-8.9 years for males. In the Indian Ocean, both males and females mature at around 190-200 cm TL (IOTC 2014). Similar to other Carcharhinid species, the oceanic whitetip shark is viviparous (i.e., the species gives birth to live young) with placental embryonic development. The reproductive cycle is thought to be biennial, giving birth on alternate years, after a lengthy 10-12 month gestation period. The number of pups in a litter ranges from 1 to 14, with an average of 6, and there is a potential positive correlation between female size and number of pups per litter, with larger sharks producing more offspring (Compagno 1984; Seki et al. 1998; Bonfil et al. 2008; IOTC 2015). Size at birth also varies slightly between geographic locations, ranging from 55 to 75 cm TL in the North Pacific, around 65-75 cm TL in the northwestern Atlantic, and 60-65 cm TL off South Africa, with reproductive seasons thought to occur from late spring to summer (Compagno 1984; Bonfil et al. 2008). Tropical Pacific records of pregnant females and newborns are concentrated between 20ºN and the equator, from 170ºE to 140ºW. In the Atlantic, young oceanic whitetip sharks have been found well offshore along the southeastern coast of the U.S., suggesting that there may be a nursery in oceanic waters over this continental shelf (Compagno 1984; Bonfil et al. 2008). In the southwestern Atlantic, the prevalence of immature sharks, both female and male, in fisheries catch data suggests that this area may serve as potential nursery habitat for the oceanic whitetip shark (Coelho et al. 2009; Tambourgi et al. 2013; Tolotti et al. 2013; Fredou et al. 2015). It appears that juveniles concentrate in equatorial latitudes, while specimens in other maturational stages are more widespread (Tambourgi et al. 2013). Pregnant females are often found close to shore, particularly around the Caribbean Islands. Evidence suggests female oceanic whitetip sharks may come close to shore to pup (Clarke et al. 2015b). However, the locations of the nursery grounds are not well known but thought to be in oceanic areas. Table 24 below provides a summary of life history characteristics reported in published literature to date.

Table 24: Life history parameters of C. longimanus from published literature (obs. = observed; m = male; f = female; PCL = Precaudal length; TL = Total Length). Source: Young et al. 2016.

Parameter	**Estimate**	**Reference**
Growth rate (von Bertalanffy k)	0.075-0.099 year-1 (SW Atlantic; both sexes)	Lessa et al. (1999)
	0.103 year-1 (N. Pacific; both sexes)	Seki et al. (1998)
	0.0852 year-1 (western N. Pacific; both sexes	Joung et al. (2016)
Max length	325 cm TL (SW Atlantic)	Lessa et al. (1999)
	245 cm PCL (342 cm TL; N. Pacific)	Seki et al. (1998)
	246 TL (f; obs; N. Pacific)	Joung et al. (2016)
	268 TL (m, obs; N. Pacific)	
	272 cm TL (Atlantic)	Cortés (2002, 2008a)
	252 cm TL (f; obs; SW Atlantic)	Coelho et al. (2009)
	253 cm TL (m; obs; SW Atlantic)	
	227 cm TL (f; obs; SW Atlantic)	Tambourgi et al. (2013)
	242 cm TL (m; obs; SW Atlantic)	
	252 cm TL (f; obs S. Atlantic)	Rodrigues et al. (2015)
	242 cm TL (m; obs; S. Atlantic)	
Age at maturity (years)	6-7 (SW Atlantic; both sexes)	Lessa et al. (1999)
	4–5 (N. Pacific; both sexes)	Seki et al. (1998)
	8.5-8.8 years (N. Pacific; females)	Joung et al. (2016)
	6.8 – 8.9 years (N. Pacific; males)	
Length at maturity (cm TL)	180-190 (SW Atlantic; both sexes)	Lessa et al. (1999)
	170 (SW Atlantic; f)	Tambourgi et al. (2013)

Parameter	Estimate	Reference
	170-190 (SW Atlantic; m)	
	168-196 (N. Pacific; f)	Seki et al. (1998)
	175-189 (N. Pacific; m)	
	190 cm TL (N. Pacific; f)	Joung et al. (2016)
	172 cm TL (N. Pacific; m)	
	190-240 (Indian Ocean; both sexes)	IOTC (2015a)
Longevity (years)	19 (obs; SW Atlantic)	Rodrigues et al. (2015)
	17 (theoretical; SW Atlantic)	Lessa et al. (1999)
	11-12 (obs; N. Pacific)	Seki et al. (1998); Joung et al. 2016
Gestation period	9 months (Pacific)	Bonfil et al. (2008)
	12 months (Pacific)	Chen 2006 in Liu and Tsai (2011)
	10-12 months (SW Atlantic)	Coelho et al. (2009)
Reproductive[5] periodicity	Every year (Pacific)	Chen 2006 in Liu and Tsai (2011)
	Every other year (SW Atlantic)	Tambourgi et al. (2013)
	Resting period of 12 months (Pacific)	Backus et al. (1956); Seki et al. (1998)
Size at birth	63-77 cm TL (N. Pacific)	Seki et al. (1998)
	64 cm TL (N. Pacific)	Joung et al. (2016)
	50-65 cm TL (Indian Ocean)	White (2007)
Litter size (# of pups)	5-6 (SW Atlantic)	Lessa et al. (1999)
	1-14 (average = 6; N. Pacific)	Seki et al. (1998)
	10-11 (N. Pacific)	Joung et al. (2016)
	12 (Indian Ocean)	IOTC (2015a)
Generation Time	7 years	Cortés (2002)
	11.1 years	Smith et al. (2008)

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[5] Most data suggest a resting period of one year (Clarke et al. 2015b)

Parameter	Estimate	Reference
Productivity (r, intrinsic rate of population increase, yr^{-1})	r= 0.067 (0.028-0.112)	Cortés (2008a)
	r= 0.094 (0.06-0.137)	Cortés et al. (2010)[6]
	r= 0.111 (0.038-0.197)	Cortés (2002)
	r= 0.121 (0.104-0.137)	Cortés et al. (2015)
	r= 0.15 (0.12-0.18)	Murua et al. (2012)

4.2.5.3 Status

Information on the global population size of the oceanic whitetip is lacking. However, several lines of evidence suggest that the once common and abundant shark has experienced declines of potentially significant magnitude due to significant fishing pressure. For example, the oceanic whitetip has declined by approximately 80 to 95% across the Pacific Ocean since the mid-1990s. Substantial abundance declines have also been estimated for the Atlantic Ocean, including an 88% decline in the Gulf of Mexico due to commercial fishing. Given their life history traits, particularly their late age of maturity and low reproductive output, oceanic whitetip sharks are inherently vulnerable to depletions, with low likelihood of recovery. Additional research is needed to better understand the population structure and global abundance of the oceanic whitetip shark.

A wide variety of existing laws and regulations have been implemented throughout the range of the oceanic whitetip shark that may positively affect the conservation status of the species including regulations to protect the species, retention prohibitions, improving data reporting, and increasing research (Young et al. 2016). These measures, with various caveats, could potentially reduce the overall bycatch mortality rates of oceanic whitetips as they have a relatively higher at-vessel survivorship compared to other shark species (Musyl et al. 2011), meaning a larger proportion may survive once released, as demonstrated in Hawaiian longline fishery (Young et al. 2016). While it's important to note that post-release mortality may still occur regardless of any implemented conservation measures and as the species become rarer, they may become more valuable as a target.

More information is required at this time to determine the magnitude of impact the shark trade is having specifically on oceanic whitetip sharks even with the downturn of the fin market. Data collection and research needs to be increased in most regions and reporting standards need to be standardized between nations (Young et al. 2016). Regulatory mechanisms, including a complete ban on shark fishing, appear to be working in places like the Bahamas while these mechanisms are making progress with the population potentially stabilized in the Northwest Atlantic and increased survivability in Hawaiian waters (Young et al. 2016). Young et al. (2016) recognizes that regulatory mechanisms alone may not equate effectiveness in achieving their intended

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[6] This value was deemed the most reasonable in a review conducted by the Pacific Shark Life History Expert Panel Workshop (Clarke et al. 2015b).

purpose. Complex issues limit the effectiveness of well-intended statutes and legislation within the range of this species (Young et al. 2016).

Only one stock assessment has been provided for the oceanic whitetip shark to date although the conclusions have been reinforced by additional studies (Clarke et al. 2011a; Brodziak et al. 2013; Rice et al. 2015). Rice and Harley (2012) concluded that the oceanic whitetip shark is not only experiencing overfishing in the Western and Central Pacific, but the stock is currently in an overfished state (Rice and Harley 2012; Young et al. 2016).

4.2.5.4 Threats to the Species

Threats to the oceanic whitetip shark are summarized in the status review (Young et al. 2016). However, the most significant threats identified for the species are fisheries bycatch and exploitation for the fin trade. As Young et al. (2016) describes, habitat degradation does not appear to be causing a range contraction in this species to date. However, climate change may factor into the species continued survival.

Natural Threats

Studies specific to oceanic whitetip sharks have not been conducted relating to the impacts of climate change. However, several factors, such as alterations in ocean temperatures, food chain dynamics, and modifications in ocean currents, could impact oceanic whitetips. Chin et al. (2010) looked at in situ changes and effects expected to occur in the Great Barrier Reef for a multitude of species which were lumped into categories based on habitat preferences. Exposure, sensitivity, and adaptive capacity were assessed for each species resulting in a ranking which were then related to the susceptibility of the various habitat groupings. Those factors include both water and air temperature, ocean acidification, freshwater input, ocean circulation, sea level rise, severe weather, light, and ultraviolet radiation (Young et al. 2016). Oceanic whitetips were placed into the pelagic group which displayed relatively low risk as a group with a handful of factors, such as oceanographic alterations, rising ocean temperatures, phenology and migration patterns, providing moderate to high exposure risks (Chin et al. 2010). Furthermore, all pelagic species displayed low sensitivity and rigidity which lowered the risk ranking to climate change factors (Chin et al. 2010; Young et al. 2016).

Hazen et al. (2012) on the other hand used data derived from an electronic tagging project and a climate change model to predict shifts in habitat and diversity for apex predators in the Pacific (Young et al. 2016). Results presented significant alterations in habitat transformation among species groups, especially for members of the shark guild which had the greatest risk of pelagic habitat loss (Hazen et al. 2012; Young et al. 2016).

When considering the studies by Hazen et al. (2012) and Chin et al. (2010), it’s important to note that these studies do not account for various factors like food web dynamics, species interactions, intra-specific competition, fine-scale habitat use, or migration patterns, only provide a conceptual risk assessment, and are not specific to the oceanic whitetip shark (Young et al. 2016). Additionally, ecosystems are extremely complex which complicates climate change modeling result interpretations and the expectations of how results will affect a species. However, it is certain that impacts from climate change specific to the oceanic whitetip are ambiguous and require additional species-specific studies. Field et al. (2009) discusses, although not specific to oceanic whitetips, that climate change might affect some species phenological and/or physiological processes with the most probable response by a species being alterations in their migrations, timing of those migrations, and overall shifts in a species distribution. NMFS expects while climate change may pose a minimal threat to the species in general, namely based on potential habitat modifications, shifts in ocean currents or temperatures, shifts in food web dynamics, or species-specific impacts; that the oceanic whitetip shark can adapt to these variations by transiting to areas favorable to their biological and ecological needs (Young et al. 2016).

Anthropogenic Threats

Global trends in the abundance of the oceanic whitetip shark is lacking based on poor quantitative data (Smale 2008; Young et al. 2016). Nevertheless, using the best available scientific and commercial information, significant reductions in the oceanic whitetip population have been associated with retention as bycatch and mortality in commercial fisheries from around the globe including the Indian Ocean, Western and Central Pacific, and the Northwest and Southwest Atlantic Ocean, (Romanov 2002; Huang and Liu 2010; Young et al. 2016). While all stocks of the oceanic whitetip are experiencing varying exploitation rates, it appears only the Northwest Atlantic population has stabilized. Information is not present for other stocks and those where data is present, note continued reductions in the population. Impacts from fishing mortality are poorly understood in all regions except for the Western and Central Pacific where stock assessments occur (Young et al. 2016). Several indices relating to abundance are available to infer population trends in certain regions.

Young et al. (2016) summarizes both qualitative and quantitative data from the Indian Ocean suggesting significant population declines are occurring, and will continue to occur, within this region. Fisheries catch data is present for a period of approximately 60 years, however the status of the oceanic whitetip is rather ambiguous due to lack of abundance data in the region or accurate catch accounting (Romanov et al. 2010; Herrera and Pierre 2011; O'Meara et al. 2011; Young et al. 2016). Decreases have been noted in the mean weights and nominal CPUE specific to the oceanic whitetip (Varghese et al 2015). Furthermore, data from purse seine fisheries in this region, while lacking robust data standards, establish probable declines in whitetip populations between 25 - 90% (Young et al. 2016). Oceanic whitetips are considered the fifth most vulnerable species to longline operations and first species for purse seine operations in the region due to the species susceptibility of being caught (Murua et al. 2012; IOTC 2015; Young et al 2016). Meanwhile, high numbers of oceanic whitetips are being reported to the IOTC, representing upwards of 11% of the total shark catch and corresponds with a high rate of at-vessel mortality within this region (Murua et al. 2013; IOTC 2015; Young et al. 2016). Due to the species low to moderate level of productivity in conjunction with the expected continued fishing pressure, it is expected that Indian Ocean populations of the oceanic whitetip will be impacted significantly.

Oceanic whitetips were once one of the most abundant pelagic shark species encountered in the Western and Central Pacific (Molony 2007). Substantial and sustained declines, upwards of 90%, have been documented for the oceanic whitetip shark population within the Western and Central Pacific region (Figure 35, Figure 36, and Figure 37) (Clarke et al. 2011a, 2011b; Clarke et al. 2012; Lawson 2011; Rice and Harley 2012; Rice et al. 2015; Young et al. 2016). The data shows larger numbers of sharks initially as fishing ramped up within the region. Considering the life history of the species, these data illustrate the inability of the species to recover from fishing pressure with decreased numbers of individuals caught each subsequent year.

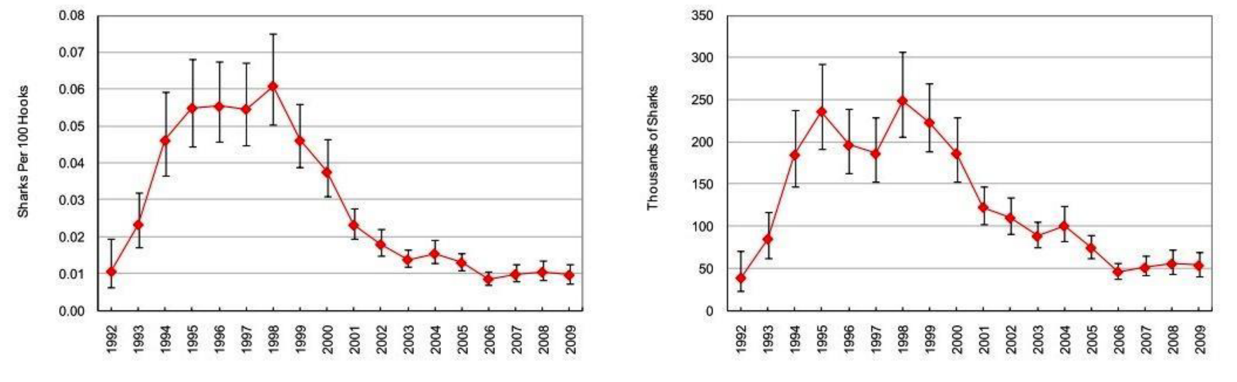

Figure 35. Estimates of longline catch rates (left) and catches (right) of oceanic whitetip sharks in the WCPFC Statistical Area east of 130°E. Source: Lawson 2011; Young et al. 2016.

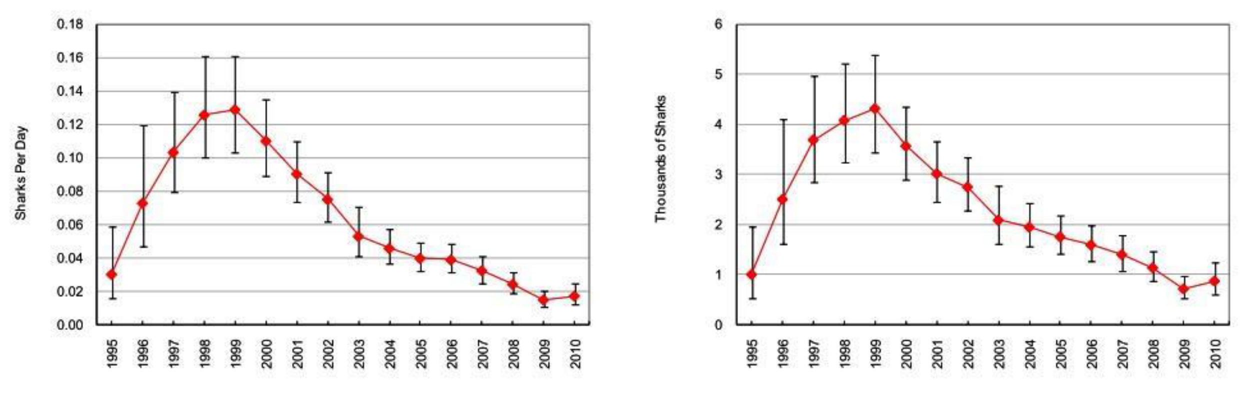

Figure 36. Estimates of purse seine catch rates (left) and catches (right) of oceanic whitetip sharks in the area from 20°S to 20°N and 130°E to 210°W. Source: Lawson 2011; Young et al. 2016.

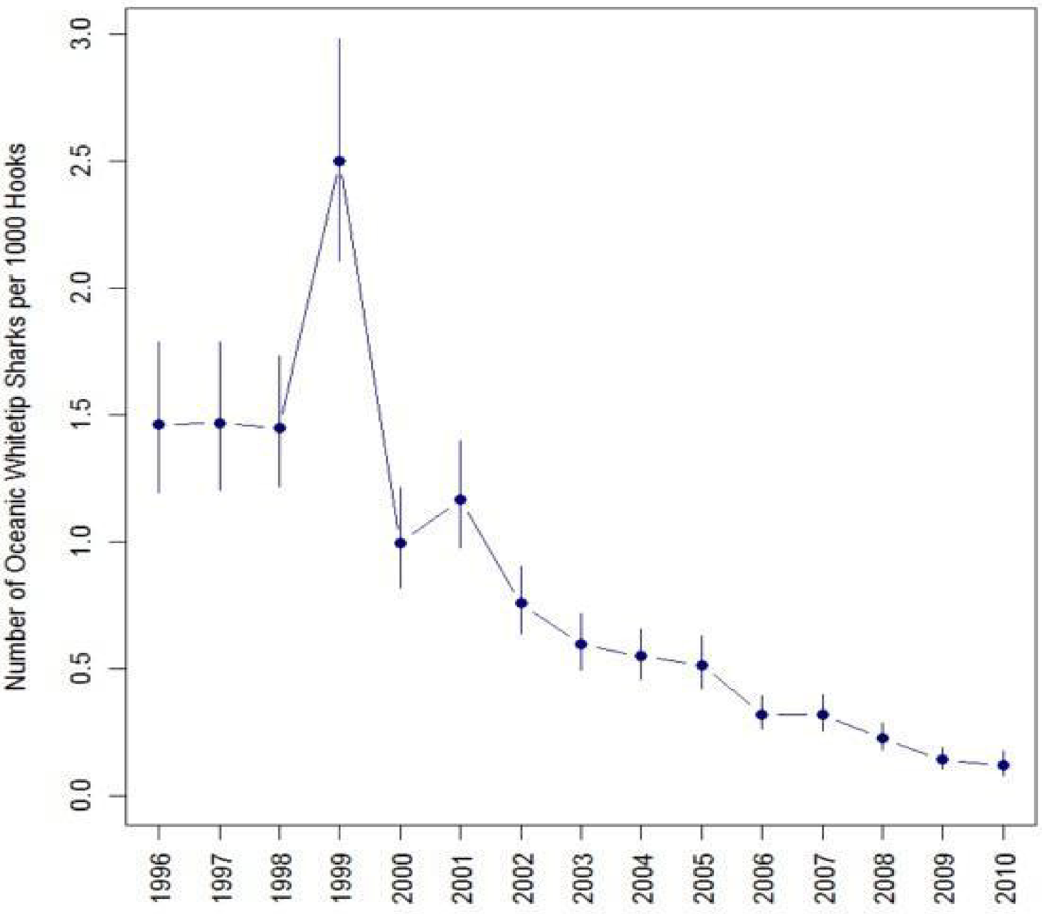

Figure 37. Standardized catch rates using a quasi-Poisson formulation of a generalized linear model for oceanic whitetip sharks in the Western and Central Pacific Source: Clarke et al. 2011a; Young et al. 2016.

Thus, the oceanic whitetip shark is not only experiencing overfishing in the Western and Central Pacific, but the stock is currently in an overfished state (Clarke et al. 2011a; Rice and Harley 2012; Brodziak et al. 2013; Rice et al. 2015; Young et al. 2016). Bycatch-related mortality in longline fisheries, with purse seine fisheries being secondary sources of mortality, are considered the primary drivers for these declines (Clarke et al. 2011; Rice and Harley 2012; Young et al. 2016). Additionally, these fisheries concentrate their fishing efforts in the species primary distribution range, thereby influencing fisheries-related mortality rates. Average landings for the Solomon Islands alone, between 2000 and 2014 were 12.9 tons with 121.2 tons unreported, annually (Hylton et al. 2017). Given the extensive fishing efforts already practiced by many nations within this region and considering reported increases in their fishing effort rates, overutilization of the oceanic whitetip shark within the Western and Central Pacific is expected to continue in this portion of the species global range which is further supported by decreased catch rates and reductions of biomass and size indices of caught fish (Clarke et al. 2011; Rice et al. 2015; Young et al. 2016). Some nations, like Fiji, have taken measures to reduce at-vessel mortality rates which show promise (Piovano and Gilman 2017). Median shark and ray catch estimates were modeled by Peatman et al. (2018) for multiple regions of the WCPFC using longline observer data. Table 25 shows statistics relevant to the oceanic whitetip shark between 10S and 10 N and South of 10S from 2003 to 2017 which includes a portion of the HI SSLL fishery action area and is an excerpt from Table 30 of Peatman et al. (2018). It should be noted that this data already includes U.S. 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 10N; and has a wide confidence interval for key shark species (Peatman et al. 2018).

Table 25: Median oceanic whitetip shark catch estimates (‘000 individuals) between 10S and 10N, and South of 10S. Source: Peatman et al. 2018.

Oceanic Whitetip Shark (reported in ‘000 individuals)		
Year	**10S to 10N**	**South of 10S**
2003	96	32.6
2004	87.9	25.8
2005	57.3	2.03
2006	45.3	18.3
2007	48.5	15.5
2008	39.4	14.4
2009	45.9	18.1
2010	41.5	25.2
2011	42	21.5
2012	44.6	17.2
2013	25.6	12.4
2014	21	10.1
2015	26.9	10.7
2016	21.5	11.1
2017	14.7	10.3
Totals	658.1	263.5

Significant population declines have been summarized by Young et al. (2016) in the Eastern Pacific, due to bycatch-related mortality in both the purse seine and longline fisheries. This was determined using tuna purse seine fishery catch data which showed a decline of 80-95% in the tropical Eastern Pacific (Hall and Roman 2013; Young et al. 2016). It should be noted that these values were derived from nominal catch rates, are not standardized to account for other catch rate variables and are somewhat ambiguous. However, the species has apparently disappeared from fishing grounds in the region and are rarely encountered while interactions with silky sharks, a close pelagic relative, appears to remain constant suggesting overutilization is the main and significant factor in the observed decline of this species (Hall and Roman 2013; Young et al. 2016). Arauz (2017) provided a response to the ESA listing for fisheries conducted in Costa Rican waters describing a decreasing trend for oceanic whitetips. In 2011, whitetips comprised 2.26% of the total shark carcasses landed out of a total of 91,822, 0.04% of 173,394 sharks in 2012, and 0.02% of 91,738 total sharks in 2013, with no oceanic whitetips landed since 2013 (Arauz 2017). Fishing effort in the region is only expected to increase, including the number of Floating Aggregating Devices (FADs) (Hall and Roman 2017) where upwards of 90% of oceanic whitetips are caught in the region; accompanied by the associated mortality from purse seine fisheries, will most likely result in continued overutilization of the oceanic whitetip shark (Young et al. 2016).

More recent data conflicts with the expectation of the Hall and Roman (2016) and the ERA team, which has shown approximately 1.6% of sharks caught in the purse seine fisheries from 2010 to 2016 are oceanic whitetips (Figure 38 and Table 26) which are most likely a result of decreased number of FADs and are indicative of population declines in the area (Peatman et al. 2017).

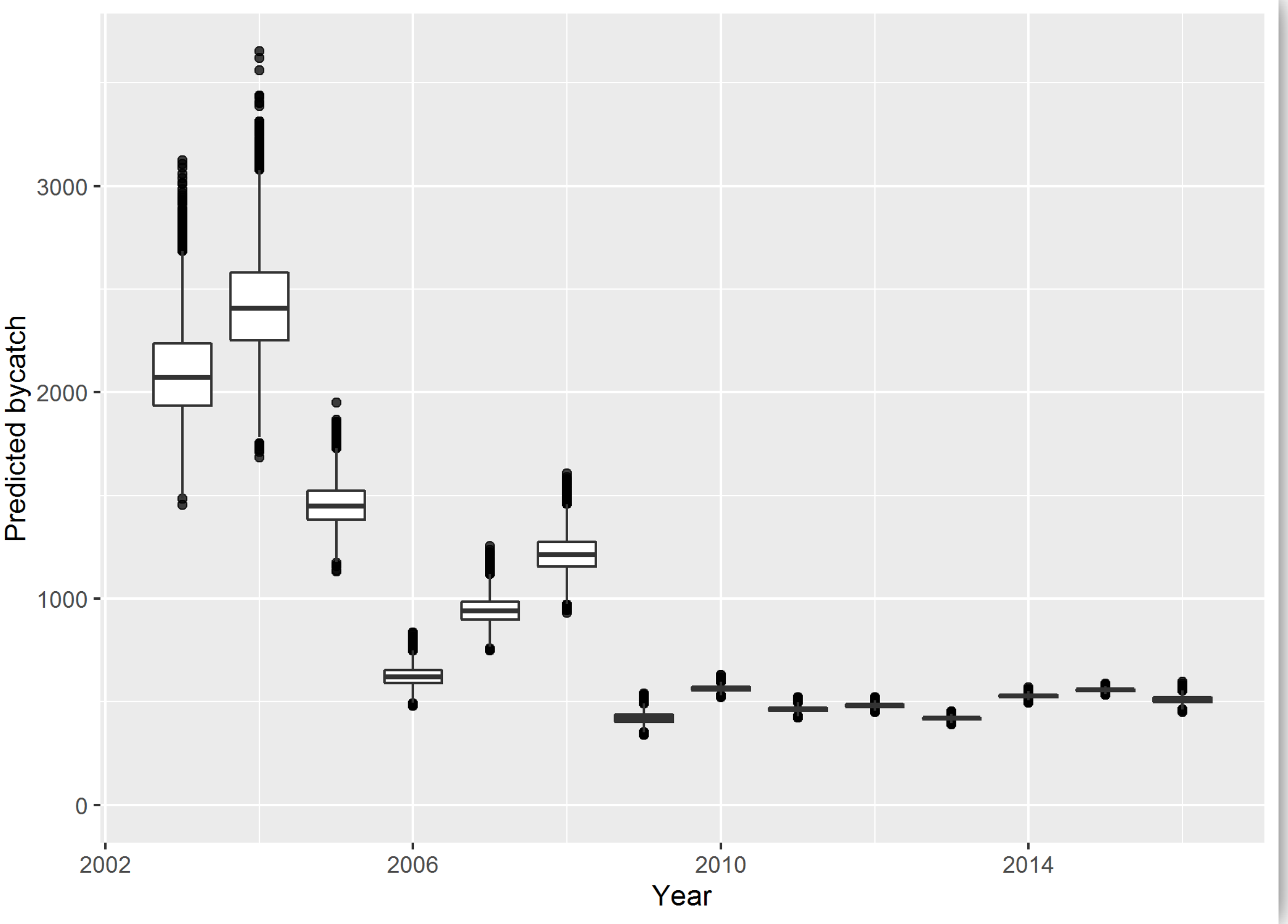

Figure 38. Predicted total annual oceanic whitetip bycatch (numbers) by year for large-scale purse seine fleets. Source Peatman et al. 2017.

Table 26. Total estimated annual oceanic whitetip bycatch in numbers (median, and lower and upper 95% confidence intervals) for large-scale purse seine fleets. Average annual bycatch rates by set and ‘000 metric tons of target catch are also included (i.e. skunk sets). Right: Proportion of annual estimated oceanic whitetip bycatch (numbers) by association type. Source Peatman et al. 2017.

	Estimated bycatch			Bycatch rate per							
Year	Low	Median	High	set	‘000 mt	aFAD	dFAD	log	FS	whale	Whale shk.
2003	1,709	2,073	2,586	0.068	2.08	7.0%	15.9%	63.7%	13.2%	0.0%	0.0%
2004	1,988	2,407	2,947	0.075	2.26	4.7%	14.6%	74.2%	6.6%	0.0%	0.0%
2005	1,267	1,449	1,675	0.040	1.21	6.1%	18.4%	61.5%	13.9%	0.0%	0.0%
2006	537	620	724	0.019	0.51	4.1%	15.7%	68.8%	11.4%	0.0%	0.0%
2007	822	939	1,083	0.026	0.69	4.0%	23.6%	57.3%	15.1%	0.0%	0.0%
2008	1,052	1,212	1,405	0.029	0.87	10.9%	40.8%	31.1%	17.1%	0.1%	0.0%
2009	373	421	476	0.010	0.28	4.7%	39.0%	38.5%	17.8%	0.0%	0.0%
2010	542	564	591	0.011	0.38	4.0%	36.0%	22.9%	36.7%	0.5%	0.0%
2011	439	463	490	0.009	0.33	6.3%	48.6%	21.6%	22.9%	0.6%	0.0%
2012	465	481	500	0.008	0.29	22.5%	28.4%	20.9%	28.2%	0.0%	0.0%
2013	404	419	436	0.007	0.26	3.1%	48.8%	26.1%	22.0%	0.0%	0.0%
2014	512	529	546	0.009	0.30	1.5%	53.9%	12.4%	32.2%	0.0%	0.0%
2015	543	556	571	0.012	0.35	3.0%	49.1%	10.7%	37.2%	0.0%	0.0%
2016	477	509	547	0.011	0.32	3.2%	53.2%	12.2%	30.8%	0.6%	0.0%

However, it should be noted that Peatman et al. (2017) provided modeled estimations due to the lack of record submissions to the *Western and Central Pacific Fisheries Commission* (*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=1822), for the large-scale purse seine fleet, are provided in Figure 39 (Peatman et al. 2017). An update was provided by Peatman et al. (2018) for the 2017 fishing season and reported 721 sharks, resulting in total median bycatch estimate of 13,882. As Peatman et al. (2018) 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.

The number of observed and estimated takes for the U.S. Purse seine fisheries in the Western Pacific is shown in Table 27. During this time, sets per year varied from 6,851-36,255, and the higher takes estimated for 2008 and 2009 result from total estimated sets of 36,225 and 22,636 respectively. NMFS anticipates total future fishing effort for the WCPO purse seine fishery to not exceed 12,000 sets annually.

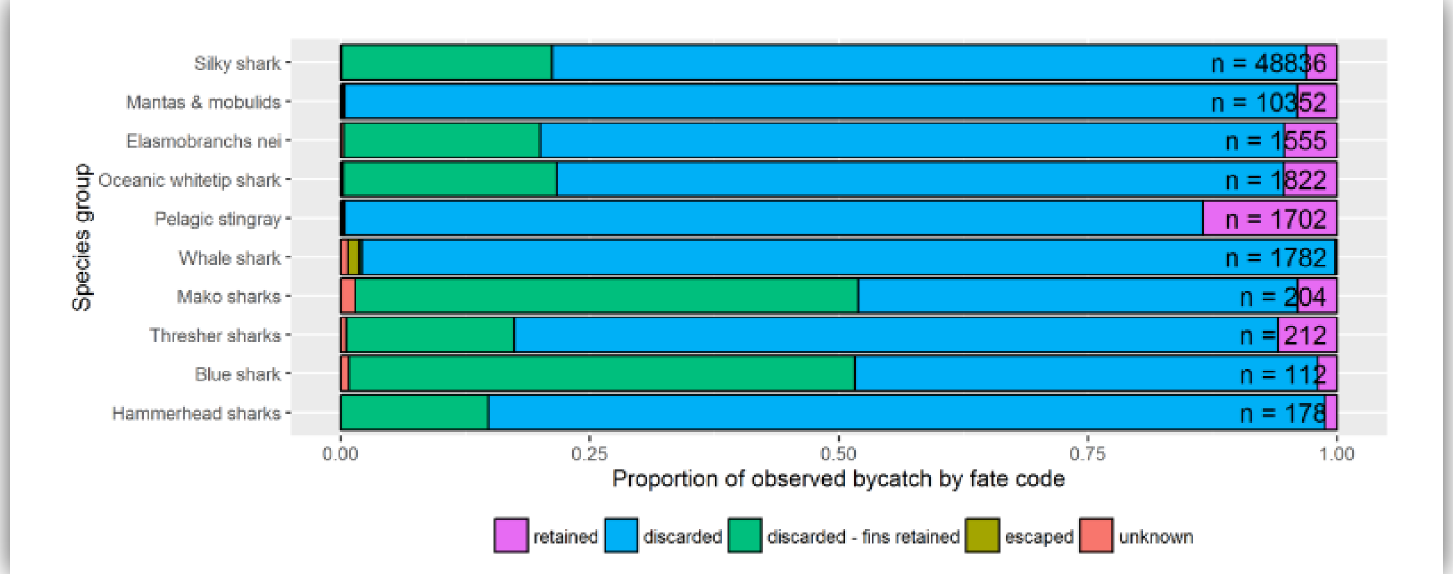

Figure 39. Recorded fate of observed sharks and rays bycatch (individuals) by species/species group, as a proportion of total observed bycatch for the species/species group in the purse seine fisheries. The number of records is provided (n = ...). Source: Peatman et al. 2017.

Table 27. Observed and estimated takes of oceanic whitetip sharks per year from 2008 to 2015. Observed interaction rates were calculated as observed takes divided by observed sets.

Year	**Number Observed**	**UCI Allocation – Unobserved Portion of Fishery**	**Estimated Total Takes**	**Estimated Interaction Rate**
2008	84	566	650	0.018
2009	20	274	294	0.013
2010	331	28	359	0.035
2011	279	11	290	0.042
2012	94	20	114	0.012
2013	95	11	106	0.013
2014	133	16	149	0.014
2015	107	22	129	0.015
Total:	1,143	947	2,091	0.018

Entanglement in FADs has been documented for silky sharks and for *Carcharinus* species which may include oceanic whitetip sharks (Chanrachkij and Loog-on 2003; Filmalter et al. 2013; Murua et al. 2017). There are no records of FAD entanglement for oceanic whitetip sharks in the WCPO purse seine fishery. Likewise, 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 have no way to quantify the exposure.

Between 2008 and 2015, 41% of oceanic whitetip sharks interacting with the WCPO purse seine fishery were caught on fishery FAD sets (Table 28). An additional 37% of sharks were caught on sets associated with natural objects including drifting logs, debris or dead animals, live whales, or undefined "other," for a total of 78% of oceanic whitetip shark takes occurring during associated sets. Pursuant to the WCPFC CMM 2017-01, NMFS issued regulations limiting the annual number of active drifting FADs to 350 per vessel. The regulation also extends the prohibition on FAD sets and sets on other natural and manmade objects, effective from July to September and from November to December for the WCPO purse seine fishery (83 FR 33851). Oceanic whitetip shark interaction rates spiked in 2010 and 2011 but have otherwise remained constant, ranging between 0.012-0.018 sharks per set. Hence, it is unclear if the FAD prohibitions have reduced oceanic whitetip shark interactions.

Table 28. School association for WCPO purse seine fishery sets where oceanic whitetip sharks were landed along with the targeted catch.

School Association	Number of Sets	Number of Oceanic Whitetip Sharks
Anchored raft FAD or payao	17	74
Drifting log debris or dead animal	38	365
Drifting raft FAD or payao	219	390
Feeding on baitfish	110	224
Live whale	6	6
Other	15	53
Unassociated	21	31
TOTAL:	426	1143

Finally, the observed and estimated total number of oceanic whitetip shark interactions in the U.S. American Samoa longline fishery from 2010-2018 are shown in Table 29 (NMFS unpub. data). Observer coverage was estimated at 20% for 2018 as data has not been finalized. The total number of observed interactions between 2010 and 2018 was 1,045 and the estimated number of interactions is 4,851 sharks.

Table 29. The number of oceanic whitetip sharks observed taken and the estimated total number of interactions based on observer coverage in the American Samoa longline fishery from 2010-2018.

Year	Observed Interactions	Observer Coverage (%)	Estimated Interactions
2010	130	25	520
2011	116	33	348
2012	71	19.80	363
2013	88	19.40	458
2014	104	19.40	541
2015	168	22	756
2016	197	19.40	1025
2017	63	20	315
2018	108	20[7]	525

Several studies have shown large historical declines in oceanic whitetip shark abundance in the Northwest Atlantic and Gulf of Mexico (Baum et al. 2003; Baum and Myers 2004; Cortes et al 2007; Baum and Blanchard 2010; FAO 2012), however, Burgess et al. (2005a, 2005b) challenged two of these studies within the literature suggesting the magnitude of the population decline was less than previously reported, and a stock assessment has not been conducted for this population (Young et al. 2016). Full details of the literature debate are summarized by Young et al. (2016). In response to the uncertainty, more recent analyses were conducted by the status review team (i.e. Young et al.) using NMFS observer logbook data from the U.S. Northwest Atlantic Pelagic Longline Fishery from 1992-2015. Results indicated this population may have stabilized, with an estimated decline of approximately 4% during this time interval (Young et al. 2016). Some caveats and limitations may apply as fishing pressure within this region started over two decades prior to the assessed interval. Current catch levels of oceanic whitetip sharks appear to be low and continue to decline in the region since regulations were implemented to protect the species in U.S. commercial International Commission for the Conservation of Atlantic Tunas-associated fisheries (NMFS 2012, 2014; Young et al. 2016). Consequently, while the port-release mortality rates of oceanic whitetips are unknown, overutilization of the species may not be as significant of a threat in the Northwest Atlantic and Gulf of Mexico, although should still be considered a viable threat (Young et al. 2016).

Oceanic whitetip sharks were common bycatch comprising nearly 30% of all shark catches in commercial longline fisheries in the Southwest Atlantic region, particularly by Brazil (Lessa et al. 1999; FAO 2012; Young et al. 2016). However extremely low CPUE in the region and high

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[7] Estimated %age; data not finalized.

capture rates of juveniles suggest they are increasingly less abundant (Lessa et al. 1999; Santana et al. 2004; Domingo et al. 2007; Coelho et al. 2009; Tambourgi et al. 2013; Tolotti et al. 2013; Fredou et al. 2015; Young et al. 2016). Tagging studies and fisheries information from area proposes that the species' preferred vertical and horizontal habitat is significantly exploited by the Brazilian longline fishery (Tolotti et al. 2013; Fredou et al. 2015; Tolotti et al. 2015; Young et al. 2016). Due to unsustainable fishing effort the species has undergone at least a 50% population decline in the region as suggested by a recent demographic study (Santana et al. 2004; Tolotti et al. 2015; Young et al. 2016). More recently, the French provided statistics on their purse seine fishery operations in the Eastern Atlantic providing valuable insights to the regime shifts being seen in the region. While the number of sharks caught were relatively low, Clavareau et al. (2018) reports a sex biased ratio of 61% of the 78 oceanic whitetips were female and of those which 59.32% were juveniles, which may further suggest philopatry is occurring in Gabon and Angola's Economic Exclusion Zones (EEZ).

The Shark Trade

Shark products including liver oil, hides, teeth and jaws, and meat, have been in demand since the early 1900s. More recent trends indicate shark fins dominate the market primarily in response to demand for shark fin soup making them the most valuable and expensive food item in the world (Fong and Anderson 2000; Biery and Pauly 2012; Young et al. 2016). In response, fisherman the world over attempted to maximize their profits and product capacity retaining only the shark's fins, cutting them off and returning the shark(s) to the ocean regardless if the shark is alive or dead (Dent and Clarke 2015; Young et al. 2016). This practice makes catch monitoring extremely difficult as it's difficult to establish the number of sharks harvested, doesn't provide reliable data, and is usually underestimated resulting in ineffective fisheries management throughout the species range (Jacquet et al. 2008; Young et al. 2016).

Young et al. (2016) summarizes various studies of the Hong Kong shark-fin market, which has been used as an indicator of the global shark-fin trade and has found oceanic whitetip sharks represent approximately 2% of the market (Vannuccini 1999; Fong and Anderson 2000; Clarke et al. 2006a, 2006b; Clarke 2008). This value equates to an annual estimate of up to 1.2 million oceanic whitetip sharks traded per year and provides a reliable metric based on the ease of species identification compared to other shark species (Young et al. 2016). Genetic studies have been conducted in markets throughout the world including Indonesia, Taiwan, and United Arab Emirates, indicating the pervasiveness of oceanic whitetip fins throughout the species range (Clarke et al. 2006a, 2006b; Liu et al. 2013; Jabado et al. 2015; Sembiring et al. 2015; Young et al. 2016). While the magnitude of the effect is uncertain, it is apparent that the shark fin trade appears to be the main driving factor which is producing significant pressure on the global oceanic whitetip shark population (Young et al. 2016).

Demand for oceanic whitetip shark fins remains high given their market preference and monetary value in Hong Kong, despite the recent declines in shark fin trade (Fong and Anderson 2000; Hazin et al. 2007; Lawson 2011; Clarke et al. 2012; Hasarangi et l. 2012; Brodziak et al. 2013; E-CoP16Prop.42 2013; Hall and Roman 2013; Young et al. 2016; Fields et al. 2018). In October of 2015, Indonesian authorities seized 3,000 illegal fins from oceanic whitetip sharks harvested in Indonesian waters, after national and international regulations were implemented to protect the species (Young et al. 2016). Young et al. (2016) notes that while demand for shark meat has increased in recent years, new markets for oceanic whitetip shark meat is not expected to occur since all regional fisheries management organizations have prohibited their retention.

4.2.5.5 Summary of the Status of the Oceanic Whitetip Shark

In this section of this biological opinion, we explained that the oceanic whitetip shark is threatened, and that the species' 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 40).

Based on the best scientific data available we've 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. 2016). Overutilization, including fisheries bycatch, and inadequate regulatory mechanisms are the most concerning threats that may contribute to the extinction risk of the species.

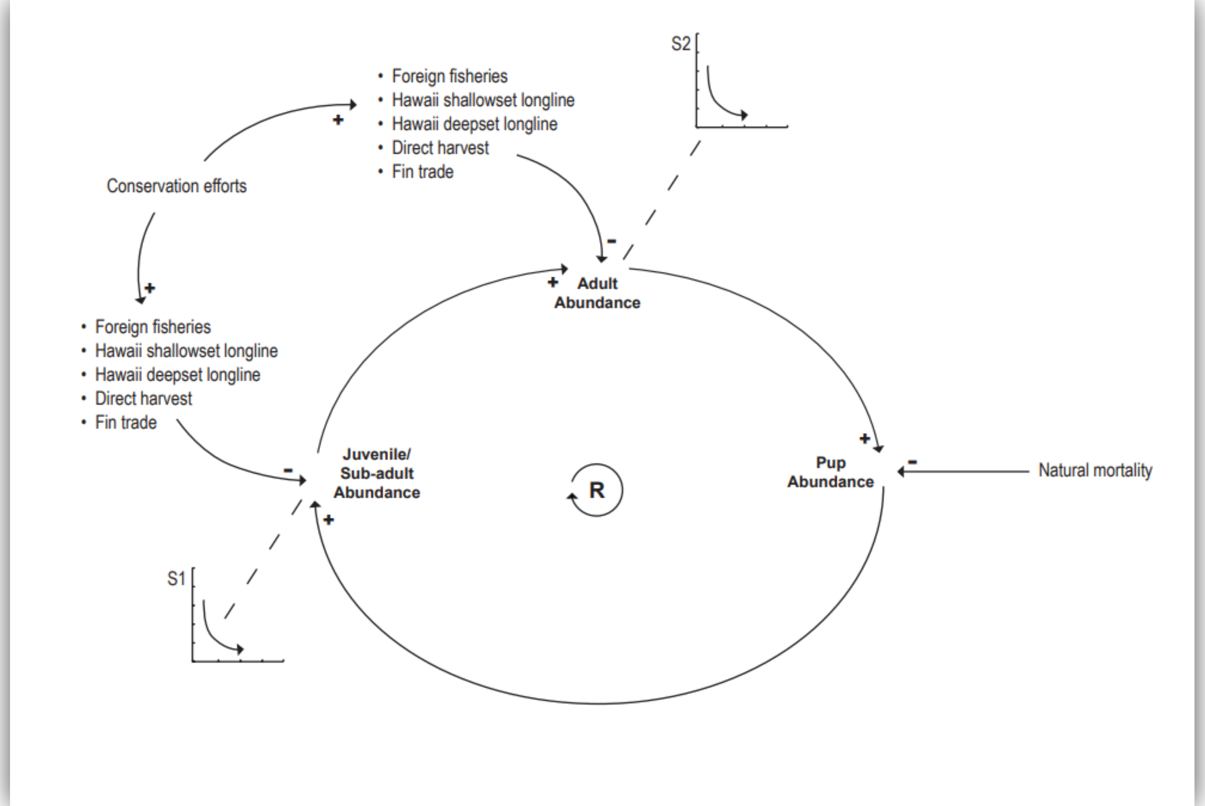


Figure 40. Oceanic whitetip shark causal loop diagram depicting various stressor-response relationships for each life cycle stage.

In addition to bycatch-related mortality, the oceanic whitetip shark is a preferred species for retention because its large fins obtain a high price per kg 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 driver of mortality of this species in commercial fisheries throughout its global range. In fact, growth in demand from the fin trade during the 1990s coincided with a pattern of soaring catches of oceanic whitetip sharks in numerous fisheries across the globe. Catches generally peaked from 1995 to 2000 followed by precipitous declines over the next 10 years due to severe overfishing (Hazin et al. 2007; Lawson 2011; Clarke et al. 2012; Hasarangi et al. 2012; Brodziak et al. 2013; Hall and Román 2013).

As a result of this fishing mortality, oceanic whitetip biomass declined by 86%. Currently, the population is overfished and overfishing is still occurring. 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 is 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 in this portion of its range is not just a localized trend, but rather a Pacific-wide phenomenon, although updated analyses of the Hawaii observer data indicate a stabilized trend at depleted levels in recent years. The significant declining trends observed in all available abundance indices (e.g., standardized CPUE, biomass, and median size) of oceanic whitetips occurred as a result of increased fishing effort in the longline fisheries, with lesser impacts from targeted longline fishing and purse-seining.

As a result of the significant fishing mortality in both longline and purse seine fisheries that has contributed to large abundance declines of the species, we've concluded that overutilization of the species is likely occurring throughout the Western and Central Pacific.

4.2.6 Giant Manta Ray

4.2.6.1 Distribution

The giant manta ray, the largest living ray, is distributed circumglobally in tropical and warm temperate bodies of water from 36°S to 40°N (Mourier 2012), however within this broad range, populations appear to be highly fragmented and sparsely distributed (Marshall et al. 2011a). As such, giant manta rays can be found in cooler water, as low as 19°C, although temperature preference appears to vary by region, with giant manta ray off the U.S. east coast commonly found in waters from 19 -22°C and those off the Yucatan peninsula and Indonesia between 25-30°C (Duffy and Abbott 2003; Marshall et al. 2009; Freedman and Roy 2012; Graham et al. 2012). Previously considered to be monospecific, Marshall et al. (2009) presented new data to support the splitting of the Manta genus into two species: giant manta ray (*M. birostris*) and reef manta ray (*M. alfredi*). Prior to 2009, all manta species were categorized as giant manta ray (*M. birostris*). Historical reports of distribution do not include the redescribed species, however, it has not been suggested to change the historical range of the giant manta ray (Miller and Klimovich 2016).

Commonly found offshore in oceanic waters near productive coastlines, continental shelves, offshore pinnacles, seamounts and oceanic islands, this species has also been observed at cleaning sites where they are cleaned of parasites by smaller organisms at offshore reefs, and feeding during the day in depths <10 m (O'Shea et al. 2010; Rohner et al. 2013). In addition, the giant manta ray has been observed in a predictable seasonal pattern in estuarine waters of Florida, Uruguay, and Brazil suggesting the estuaries may be used as a nursery ground during the summer months (Adams and Amesbury 1998; Milessi and Oddone 2003; Medeiros et al. 2015).

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), U.S. (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, with two individuals photographed off Montague Island in New South Wales and off the northeast coast of Tasmania, extending the known range of the species to 40°S. De Boer et al. (2015) recently confirmed the presence of giant manta ray in offshore shallow waters of Suriname, in depths between 28 and 42-m; however, local fishermen were familiar with the species, indicating that it likely occurs regularly in these waters.

Based on the systematic literature search conducted up to the year 2011 by Couturier et al. (2012) and the most recent search from 2012 to 2016 by Lawson et al. (2016), geographic distribution maps were created and recently refined based on current distribution knowledge of the giant manta ray (Lawson et al. 2016). These maps (Figure 41) depict the extent of occurrence and area of occupancy wherein the extent of occurrence is defined as: “the area contained within the shortest continuous imaginary boundary which can be drawn to encompass all the known, inferred or projected sites of present occurrence of a taxon”. The area of occupancy was defined as “the area within its 'extent of occurrence'” that is occupied by a taxon for each country. The area of occupancy measure reflects the fact that a taxon will not usually occur throughout the area of its extent of occurrence, which may, for example, contain unsuitable habitats or be beyond the maximum depth distribution.” The area of occupancy only included areas where the presence of the species was able to be confirmed.

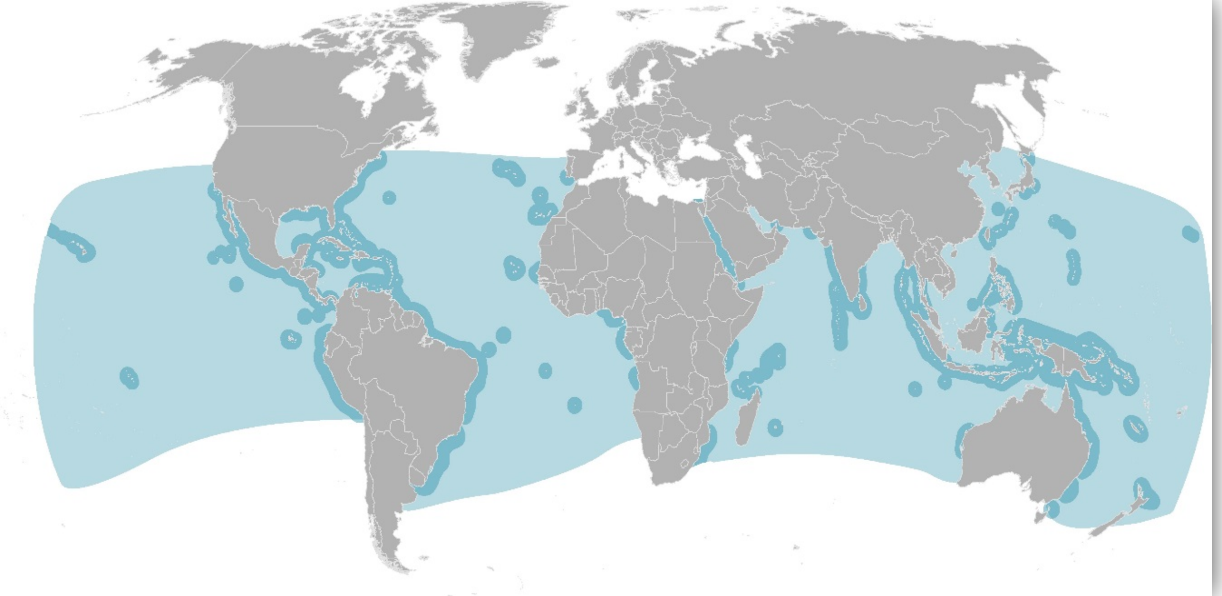


Figure 41. Distribution map for the giant manta ray. Extent of occurrence is depicted by light blue and the area of occupancy is noted in darker blue. Source: Lawson et al. 2016.

Despite this large range, sightings are often sporadic with the exceptions of (Table 30) 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), where more regular sightings are common during specific, predictable times of the year (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. 2011a; Stewart et al. 2016a).

Table 30. Locations with regular, predictable sightings of giant manta ray.

Country	**Specific location**
Thailand	Similan Islands
Indonesia	Raja Ampat
New Zealand	Northeast North Island
USA	Kona Hawaii
Brazil	Laje de Santos Marine Park
Ecuador	Isla de la Plata

Japan	Ogasawara Islands
Venezuela	Isla Margarita and Puerto la Cruz
Mexico	Isla Holbox, Revillagigedo Islands and Bagia de Banderas

The timing of these sightings varies by region (for example, the majority of sightings in Brazil occur during June and September, while in New Zealand sightings mostly occur between January and March) and seems to correspond with the movement of zooplankton, current circulation and tidal patterns, seawater temperature, and possibly mating behavior (Couturier et al. 2012; De Boer et al. 2015; Armstrong et al. 2016). As such, off the coast of South America, De Boer et al. (2015) suggest that the presence of the giant manta ray likely coincides with the rainy season and subsequent outflows of nutrient-rich water and low salinity front from associated river systems.

Feeding and Diving Behavior

The giant manta ray primarily feed on planktonic organisms such as euphausiids, copepods, mysids, decapod larvae and shrimp, but some studies have noted their consumption of small and moderate sized fishes (Bertolini 1933; Bigelow and Schroeder 1953; Carpenter and Niem 2001; The Hawaii Association for Marine Education and Research Inc. 2005). When feeding, groups of mantas hold their cephalic fins in an “o” shape and open their mouths wide. They tend to swim at a speed around 30 pectoral fin beats per minute when feeding, which is almost twice as fast as they swim when being cleaned (Kitchen-Wheeler 2013). After collecting water with zooplankton in their mouths, mantas use a transverse curtain on the roof of the mouth as a valve to hold the water in as the pharynx contracts during swallowing (Bigelow and Schroeder 1953). This movement of the pharynx pulls plankton towards the stomach when the gills are closed (Kitchen-Wheeler 2013). Intestinal eversion was also observed, probably to clear the intestines of indigestible material and parasites (Clark and Papastamatiou 2008).

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. 2011a; Rohner et al. 2013). In fact, 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; 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)). The species has a rete mirabile cranica as a counter-current heat exchanger around the brain that possibly facilitates its use of these cooler habitats (Alexander 1996). Stewart et al. (2016b) found diving behavior may be influenced by season, and more specifically, shifts in prey location associated with the thermocline, with tagged giant manta rays (n=4) observed spending a greater proportion of time at the surface from April to June and in deeper waters from August to September.

4.2.6.2 Population Structure

Since the splitting of the Manta genus, most of the recent research has examined the genetic discreteness, phylogeny and the evolutionary speciation in manta rays (Cerutti-Pereyra et al. 2012; Kashiwagi et al. 2012; Poortvliet et al. 2015). Unlike the reef manta ray, the giant manta ray does not appear in large schools (<30 individuals) and despite having a larger distribution when compared to the reef manta, they are encountered with far less frequency. Therefore, the population structure for the wider-ranging giant manta ray is less clear than it is for the reef manta.

While the Clark (2010) study found low site-fidelity for giant manta ray and much higher rate of immigration, indicative of a population that is pelagic rather than coastal or island-associated, Stewart et al. (2016a) provided recent evidence to show that the giant manta rays off Pacific Mexico may exist as isolated subpopulations, with distinct home ranges. Additionally, researchers are presently investigating whether there is a potential third manta ray species resident to the Yucatán coastal waters of the Gulf of Mexico (previously identified as giant manta ray) (Hinojosa-Alvarez et al. 2016). Using the mitochondrial ND5 region (maternally-inherited DNA), Hinojosa-Alvarez et al. (2016) found shared haplotypes between Yucatán manta ray samples and known giant manta ray samples from Mozambique, Indonesia, Japan, and Mexico, but discovered four new manta ray haplotypes, exclusive to the Yucatán samples. While analysis using the nuclear RAG1 gene (bi-parentally-inherited DNA) showed the Yucatan samples to be consistent with identified giant manta ray sampled, the authors suggest that the ND5 genetic evidence indicates the potential for a third, distinctive manta genetic group or possibly a subspecies. At this time, additional studies, including in-depth taxonomic studies and additional genetic sampling, are needed to better understand the population structure of both species throughout their respective ranges.

Considered a migratory species, the giant manta ray can travel estimated distances up to 1,500 km (Dewar et al. 2008; Marshall et al. 2009; Miller and Klimovich 2016). Satellite tracking studies using pop-up satellite archival tags register movements of the giant manta ray from Mozambique to South Africa (a distance of 1,100 km), from Ecuador to Peru (190 km), and from the Yucatan, Mexico into the Gulf of Mexico (448 km) (Marshall et al. 2011a). In a tracking study of six giant manta ray individuals from off Mexico's Yucatan peninsula, Graham et al. (2012) calculated a maximum distance travelled of 1,151 km (based on cumulative straight-line distance between locations; tag period ranged from 2 to 64 days). Similarly, Hearn et al. (2014) report on a tagged giant manta ray that was tracked from Isla de la Plata (Ecuador) to west of Darwin Island (tag was released after 104 days), a straight-line distance of 1,500 km, further confirming that the species is capable of fairly long-distance migrations but also demonstrating connectivity between mainland and offshore islands. The limited data on its migratory ecology suggest these movements are timed with enhanced productivity associated with seasonal oceanographic events driven more by broader-scale processes than local conditions (Rohner et al. 2013; Corturier et al. 2015).

Conversely, a recent study by Stewart et al. (2016a) suggests that the species may not be as highly migratory as previously thought. Using pop-up satellite archival tags in combination with analyses of stable isotope and genetic data, the authors found evidence that giant manta ray may actually exist as well-structured subpopulations that exhibit a high degree of residency. For example, unlike the giant manta ray in the Hearn et al. (2014) study, tagged giant manta ray individuals from locations nearshore to Mexico (Bahia de Banderas ; n=5) and offshore Mexico

(Revillagigedo Islands; n=4) showed no movements between locations (tag deployment length ranged from 7 days to 193 days, Stewart et al. 2016a). The stable isotope analysis showed higher $\delta^{13}C$ values for the nearshore mantas compared to those offshore, indicating these mantas were foraging in their respective locations rather than moving between nearshore and offshore environments (Stewart et al. 2016a). Additionally, using double digest restriction-site associated DNA sequencing from DNA extracted from white muscle tissue of mantas, the authors found evidence of population structure between the coastal Mexico and offshore Mexico populations (Stewart et al. 2016a). While the authors note that the species may be capable of occasional long-distance movements, the results from the study indicate that these movements may be rare and may not contribute to substantial gene flow or interpopulation mixing of individuals (Stewart et al. 2016a).

4.2.6.3 Life Stages

Not much is known about the giant manta ray's life history stages: neonate, juvenile, and adult. Growth and developmental habitats are not well described in the literature. Yet, these stages form the foundation of our causal loop diagram, which we describe in more detail later in this chapter.

The giant manta ray breeding sites have been identified off the coast of Ecuador and in the Galapagos Islands, with pregnant females observed off Isla de la Plata in the Machalilla National Park and Galapagos Marine Reserve (Hearn et al. 2014). The mating displays can last hours or days, with the female swimming rapidly ahead of the males and occasionally somersaulting or turning abruptly (Deakos et al. 2011). Males were never observed to compete with each other directly for the attention of the female, so these mating trains may function as a kind of endurance rivalry (Andersson 1994; Deakos 2012). No copulations have been observed in the wild, so it is difficult to determine which males have a mating advantage, but this kind of endurance trial usually selects for the success of larger males (Andersson and Iwasa 1996; Deakos 2012). Yano et al. (1999b) reported that giant manta ray near the Ogasawara Islands (Japan) were also seen engaging in "mating trains," where multiple males pursue a single female. They recorded the primary breeding season of giant manta ray as July-August.

Differences in mating season may exist as seasonal breeding is usually due to some kind of advantage such as food availability or predation pressure for the pup when it is born; this may vary by region. Because gestation appears to last 12-13 months, it therefore appears all mating attempts do not result in conception and females may have multiple ovulations per year if they do not become pregnant on their first mating attempt.

Giant manta rays are viviparous and reproduce most commonly by histotrophy. This is a type of matrotrophy (involving maternal input of nutrients to fetus) where the female produces lipid-rich histotroph (sometimes called uterine milk) for the embryos to consume via long villi called trophonemata (Alcock 1892; Amoroso 1960). Due to maternal nutrition having an effect on embryo development through the histotroph (Tomita et al. 2012), suggests the mother's health will impact fetal development. Reports of manta ray births and dissections have all revealed only a single embryo (Homma et al. 1999; Uchida et al. 2008).

A captive giant manta ray gave birth to a 1.8-m disk width 66-kg neonate (Okinawa Churaumi Aquarium cited in Deakos 2012). Free swimming wild mantas have been observed as small as 1.02-m DW and 1.22-m DW, so it is likely these young mantas are representative of a normal wild birth size with gestation period around a year. The young mantas were only able to swim properly after a few minutes when their wings fully unfurled, meaning that neonates would be at risk for predation during this time. Thus, the mother's choice of birth site may make a difference in survival rate (Berriman 2007; Kitchen-Wheeler 2013) as mantas do not provide any parental care to their offspring after birth. Some data suggests that estuaries are used as a nursery ground during the summer months (Adams and Amesbury 1998; Milessi and Oddone 2003; Medeiros et al. 2015).

Homma et al. (1999) hypothesized that age at sexual maturity was 8-13 years in mantas and the data of Uchida et al. (2008), Marshall et al. (2011a) and Marshall and Bennett (2010b) confirmed this estimate. Males reach maturity at a disc width of approximately 4 meters while females reach maturity at approximately 5 meters, although the size at maturity may vary slightly throughout its range (CITIES 2013).

Given their large sizes, manta rays are assumed to have fairly high survival rates after maturity (e.g., low natural predation rates) and have been reported to live to at least 40 years old (Marshall and Bennett 2010b; Marshall et al. 2011b; Kitchen-Wheeler 2013) however, the time needed to grow to maturity and the low reproductive rates mean that a female will be able to produce only 5-15 pups in her lifetime (CITES 2013). Generation time for the species (based on reef manta life history parameters) is estimated to be 25 years (Marshall et al. 2011a; Marshall et al. 2011b). Using estimates of known life history parameters for both giant and reef manta rays, and plausible range estimates for the unknown life history parameters, Dulvy et al. (2014) calculated a maximum population growth rate of *Manta* spp. and found it to be one of the lowest values when compared to 106 other shark and ray species. Specifically, the median maximum population growth rate (r_{max}) was estimated to be 0.116, a rate that is more similar to those calculated for marine mammal species than chondrichthyan species (Dulvy et al. 2014; Croll et al. 2015). Maximum population growth rates among scientists who use different model assumptions and criteria for assessing productivity have varied, however, all estimates still place both manta ray species into the "very low" productivity category ($r < 0.05$), based on the productivity parameters and criteria in Musick (1999).

In order to determine how changes in survival may affect populations, Smallegange et al. (2016) modeled the demographics of reef 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 (based on *M. alfredi* demographics) showed that increases in yearling or adult annual survival rates resulted in much greater responses in population growth rates, mean lifetime reproductive success, and cohort generation time compared to similar increases in juvenile annual survival rates (Smallegange et al. 2016). Based on the elasticity analysis, population growth rate was most sensitive to changes in the survival rate of adults (Smallegange et al. 2016). In other words, in order to prevent populations from declining further, Smallegange et al. (2016) found that adult survival rates should be increased, such as through protection of adult aggregation sites or a reduction in fishing of adult manta rays (Smallegange et al. 2016). For those populations that are currently stable, like the Yaeyama Islands (Japan) population (where adult annual survival rate is estimated at 0.95; noted above), Smallegange et al. (2016) note that any changes in adult survival may significantly affect the population.

4.2.6.4 Status

There are no current or historical estimates of the global abundance of giant manta ray and due to their global nature, accurate population estimates will always be difficult to assess (Marshall et al. 2011a). Most estimates of subpopulations are based on anecdotal diver or fisherman observations, which are subject to bias. These populations seem to potentially range from around 100-1,500 individuals (Table 31). The Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) (2013) reports that because 10 populations of giant manta ray have been actively studied, 25 other aggregations have been anecdotally identified, and all other sightings are rare, the total global population may be small.

Ecuador is thought to be home to the largest identified population of giant manta ray in the world, with large aggregation sites within the waters of the Machalilla National Park and the Galapagos Marine Reserve (Hearn et al. 2014). Within the Indian Ocean, numbers of giant manta rays identified through citizen science in Thailand's waters (primarily on the west coast, off Khao Lak and Koh Lanta) have been increasing over the past few years, from 108 in 2015 to 288 in 2016. These numbers reportedly surpass the estimate of identified giant mantas in Mozambique (n=254), possibly indicating that Thailand may be home to the largest aggregation of giant manta rays within the Indian Ocean (MantaMatcher 2016). In the Atlantic, very little information on giant manta ray populations is available, but there is a known, protected population within the Flower Garden Banks National Marine Sanctuary in the Gulf of Mexico. However, researchers are still trying to determine whether the manta rays in this area are only giant manta ray individuals or potentially also comprise individuals of a new, undescribed species (Marshall et al. 2009; Hinojosa-Alvarez et al. 2016).

Table 31. Numbers of recorded individuals and subpopulation estimates of giant manta ray at identified locations adapted from CITES (2013).

Location	Recorded Individuals	Subpopulation Estimate	Reference
Mozambique	180 - 254	600	Marshall et al. (2009) & pers. comm. cited in CITES (2013); MantaMatcher (2016)
Egypt	60		Marine Megafauna
Republic of Maldives	63		G. Stevens, pers. comm. cited in CITES (2013)
Kona, Hawaii	29		Clark (2010)
Thailand	>288	-	MantaMatcher (2016)
Raja Ampat, Indonesia	72	-	MMP & The Manta Trust, unpubl. cited in CITES (2013)
Isla de la Plata, Ecuador	~650	1,500	M. Harding, pers. comm. cited in CITES (2013); Sanchez (2016)

Brazil	60	-	Laje Viva Institute unpubl. cited in CITES (2013), Luiz et al. (2009)
Mexico (Revillagigedos Is.)	412	-	R. Rubin & K. Kumli, pers. comm. cited in CITES (2013)
Mexico (Isla Holbox)	> 200	-	R. Graham, pers. comm. cited in CITES (2013)
Flower Garden Banks	>70	-	Graham and Witt (2008) cited in CITES (2013)
Japan (Ogasawara Islands)	42	-	Kashiwagi et al. (2010)

Given the migratory nature of the species, population declines in waters where mantas are protected have also been observed but attributed to overfishing of the species in adjacent areas within its large home range. For example, White et al. (2015) provide evidence of a substantial decline in the giant manta ray population in Cocos Island National Park, Costa Rica, where protections for the species have existed for over 20 years. Using a standardized time series of observations collected by dive masters on 27,527 dives conducted from 1993 to 2013, giant manta ray relative abundance declined by approximately 89% (95% CI 85-92%) (Figure 42). Years of higher abundance of the species were correlated with lower El Niño activity. However, based on the frequency of the species' presence on dives (4%), with a maximum of 15 individuals observed on a single dive, the authors suggest that Cocos Island may not be a large aggregating spot for the species, and suggest that the decline observed in the population is likely due to overfishing of the species outside of the National Park (White et al. 2015).

In regions where giant manta rays are (or were) actively targeted or caught as bycatch, such as the Philippines, Mexico, Sri Lanka, and Indonesia, populations appear to be decreasing. In Indonesia, manta ray landings are estimated to have declined by 71% to 95%, with potential extirpations noted in certain areas (Lewis et al. 2015).

Targeted Fisheries

In the absence of a global abundance estimate for the giant manta ray, data from targeted fisheries and bycatch data collected over time can be useful to identify trends in abundance, specifically in the regions where this data is available. Giant manta rays are reportedly targeted in fisheries in Indonesia, Philippines, India, Thailand, Mozambique, Tonga, Micronesia, Peru,

Figure 42. Observed and modeled estimates of mean annual numbers of giant manta rays in Cocos Island National Park, based on diver observations from 1993 to 2015 (Source: White et al. 2015).

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). al. 2011). Manta and devil ray fisheries span the majority of the Indonesian archipelago, with most landing sites along the Indian Ocean coast of East and West Nusa Tenggara and Java (Lewis et al. 2015). This commercial trade of manta ray products, particularly gill plates, coupled with emerging technological advances (e.g., motorized vessels) and an increase in the number of boats in the fishery, greatly increased fishing pressure and harvest of manta rays in the 1990s and 2000s (Dewar 2002). In Lamakera, Indonesia, one of the main landing sites for mobulids, and particularly manta rays, Dewar (2002) estimates that the total average harvest of "mantas" during the 2002 fishing season was 1,500 (range 1,050- 2,400), a significant increase from the estimated historical levels of around 200-300 mantas per season; however, Lewis et al. (2015) note that this estimate likely represents all mobulid rays, not just mantas. Fishermen from Lamalera, whose fishing grounds overlap with the Lamakera fishing fleet, reported landings of around 200-300 per season but noted that very few mantas were caught from 1998-2001, and attributed the low catch to the presence and competition of Taiwanese fishing ships, which also began fishing off Lamalera in large numbers in the 1990s (Barnes 2005).

Although fishing for manta rays was banned within the Indonesian 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).

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

In India, manta rays are mainly landed as bycatch in tuna gillnetting and trawl fisheries; however, a harpoon fishery at Kalpeni, off Lakshadweep Islands, is noted for "abundantly" landing mantas (likely *M. alfredi*; A.M. Kitchen-Wheeler pers. comm. 2016) during June – November, with peak season from June-August (Raje et al. 2007). Specific landings figures were unavailable.

According to Heinrichs et al. (2011), dive operators in the Similan Islands have observed an increase in fishing for manta rays, including in protected Thai national marine parks, and between 2006 and 2012, sightings of *Manta spp.* (likely the giant manta ray*)* had decreased by 76% (CITES 2013b).

In southern Mozambique, the giant manta ray represents only 21% of the identified manta rays in this area, and is rarely observed in the local fishery (one observed caught over 8-year period), indicating that fishing pressure is likely low on this species (Rohner et al. 2013; Marine Megafauna Foundation 2016).

Opportunistic hunting of manta rays has been reported in Tonga and Micronesia (B. Newton and J. Hartup pers. comms. 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 fishery, it is unlikely that fishing pressure is significant on the species.

Little information is available on the level of take of manta rays by Peruvian fisheries. Heinrichs et al. (2011), citing to a rapid assessment of the mobulid fisheries in the Tumbes and Piura regions of Peru, reported estimated annual landings of giant manta rays on the order of 100-220 rays for one family of fishermen. As such, total landings for Peru are likely to be much larger.

There is no available data on the amount of manta rays landed in Ghanaian fisheries; however, Debrah et al. (2010) observed that giant manta rays were targeted using wide-mesh drift gillnets in artisanal fisheries between 1995 and 2010. D. Berces (pers. comm. 2016) confirmed that mantas are taken during artisanal fishing for pelagic sharks, and not "infrequently," with manta rays consumed locally.

In the Maldives, Anderson and Hafiz (2002) note that manta rays may be caught in the traditional fisheries, with meat used for bait for shark fishing and skin used for musical drums; however, the authors state that catches are generally very small, potentially as little as 10 tons per year.

Manta and mobula rays were historically targeted for their meat in the Gulf of California. In 1981, Notarbartolo di Sciara (1988) observed a seasonally-active mobulid fishery located near La Paz, Baja California Sur. Mobulids were fished in the Gulf of California using both gillnets and harpoons, with their meat either fileted for human consumption or used as shark bait. The giant manta ray was characterized as "occasionally captured" by the fishery. While it is unclear how abundant the giant manta was in this area, by the early 1990s, Homma et al. (1999) reported that the mobulid fishery had collapsed and CITES (2013), referencing anecdotal dive reports by a filmmaker, noted a decrease in manta ray sightings from 3-4 individuals per dive in 1981 to zero in 1991-1992.

Bycatch

Given the global distribution of the giant manta ray, they are frequently caught as bycatch in a number of commercial and artisanal fisheries worldwide. In a study of elasmobranch bycatch patterns in commercial longline, trawl, purse-seine and gillnet fisheries, Oliver et al. (2015) presented information on species-specific composition of ray bycatch in 55 fisheries worldwide. Based on the available data, Oliver et al. (2015) found that giant manta rays comprised ~40% of ray bycatch in the purse-seine fisheries operating in the Indian Ocean and especially the Eastern Pacific Ocean, but were not large components of the ray bycatch in the longline, trawl, or gillnet fisheries in any of the ocean. In the Atlantic Ocean, bycatch of giant manta rays has been observed in purse-seine, trawl, and longline fisheries; however, as was noted in the Oliver et al (2015) study, based on the available data, the giant manta ray does not appear to be a significant component of the bycatch.

In the European purse-seine fishery, which primarily operates in the Eastern Atlantic off western Africa, the giant manta ray is also caught, primarily in Fish Aggregating Device (FAD) purse-seine sets, however, it does not appear to be caught in large numbers (Amandè et al. 2010; Hall and Roman 2013). Based on data from French and Spanish observer programs, collected over the period of 2003-2007 (27 trips, 598 sets; observer coverage averaged 2.93%), only 11 giant manta ray were observed caught by the European purse-seine fleet (Amandè et al. 2010). Additionally, fishing effort by the Atlantic tuna purse seine fishery is significantly less compared to the effort of the purse-seine fisheries in the other ocean basins, accounting for only 7% of the total number of tuna purse seine sets a year (Croll et al. 2015).

In the U.S. bottom longline and gillnet fisheries operating in the western Atlantic, the giant manta ray is also a rare occurrence in the elasmobranch catch. Based on data from the NMFS shark bottom longline observer program, between 2005 and 2014, only 2 giant manta rays were observed caught by bottom longline vessels fishing in the Gulf of Mexico and South Atlantic, with one discarded alive and one kept (NMFS 2014).

In the U.S. gillnet fishery, which has been greatly reduced since the implementation of Amendment 2 to the Consolidated Atlantic Highly Migratory Species Fishery Management Plan (NMFS 2007), bycatch of manta rays is low. Based on 1998 – 2015 data from the NMFS Southeast Gillnet Observer Program, which covers all anchored (sink and stab), strike, or drift gillnet fishing by vessels operating in waters from Florida to North Carolina and the Gulf of Mexico, the number of observed mantas in a given fishing year has ranged from zero to only 16, with the vast majority (around 89%) discarded alive (NMFSa 2015).

Off Mauritania, Zeeberg et al. (2006) documented giant manta ray in the bycatch of the European pelagic freezer-trawler fishery. In fact, between October 2001 and May 2005, 9% of the retained pelagic megafauna bycatch (including sharks, bill fish, rays, and cetaceans) from over 1,400 freezer-trawl sets consisted of giant manta rays (Zeeberg et al.2006). The authors note that the probability of catching manta rays in this region is minimal during the winter and spring (December to June) due to colder water temperatures (around 18°C), but increases as the trade winds decrease towards the end of spring. Based on observer data from July to November (2001-2005), Zeeberg et al. (2006) estimated that between 120 and 620 mature manta rays were removed annually in the Mauritanian EEZ by trawler fisheries, a level the authors concluded was likely unsustainable for the giant manta ray population. However, as a result of an agreement between the European Union (EU) and Mauritania, which set technical conditions that, according to the Pelagic Freezer-trawler Association (PFA), made commercial fishing economically unviable, the European freezer-trawler fleets have rarely operated in Mauritanian waters since 2012 (PFA 2016).

In the Indian Ocean, giant manta rays are mainly caught as bycatch in purse-seine and gillnet fisheries. In the pelagic tuna purse-seine fishery, Romanov (2002) estimated mobula and manta ray bycatch from the western Indian Ocean using observer data collected on Soviet purse seine vessels from 1985-1994. Romanov (2002) further extrapolated the observer data across the principal fishing nations operating within the western Indian Ocean (France, Spain, USSR, Japan, and Mauritius). In terms of numbers of individuals, Romanov (2002) estimated that between 253 and 539 mobulas and mantas (lumped together) were taken per year, with bycatch generally increasing over the time period. When compared to the total non-tuna bycatch, mantas and mobulas comprised less than 1% of the total per year. More recent data suggest that these rays have remained an insignificant portion of the bycatch, indicating that these animals may not be highly susceptible to purse-seine fishery operations in the western Indian Ocean. Both giant manta ray and mobula species are caught in similar amounts, primarily in FAD sets; however, based on 1,958 observed sets from 2003- 2007 (coverage rate averaged 4%), rays, as a species group, comprised only around 0.7% of the total bycatch. Less than 35 giant manta rays were observed in the bycatch, with around 60% discarded dead and 40% discarded alive (Amandè et al. 2008). Within the Indian Ocean, the available data indicate that manta rays appear to be at higher risk of capture from the fisheries operating in the eastern Indian Ocean, with two of the top three largest giant manta ray fishing and exporting range states (Sri Lanka and India) located in this region (Heinrichs et al. 2011).

In Sri Lanka, manta rays are primarily caught as bycatch in the artisanal gillnet fisheries. While fishermen note that they generally tend to avoid deploying nets near large aggregations of mantas, or regularly release particularly large mantas (due to the difficulties associated with entanglement and killing the species and loss of boat time), as recently as 2011, manta rays were observed being sold at the Negombo and Mirissa fish markets (Fernando and Stevens 2011). While Sri Lankan fishermen state that they try to release pregnant and young manta rays alive, based on 40 observed giant manta ray being sold at markets (from May through August 2011), 95% were juveniles or immature adults (Fernando and Stevens 2011).

Extrapolating the observed numbers to a yearly value, Fernando and Stevens (2011) estimated annual landings of the giant manta ray at Negombo to be 194 individuals and at Mirissa to be 126 individuals. Using these values, and after making general assumptions about the landings at all of the other fish markets within the country, the authors estimated total annual landings for the giant manta ray in Sri Lanka to be around 1,055 individuals (Fernando and Stevens 2011). While it is difficult to determine whether these levels equate to overutilization of the species within the Indian Ocean, given the lack of baseline population estimates throughout most of the species' range, the authors concluded that the Sri Lankan fisheries are likely having a "significant and detrimental impact" on the giant manta ray population that may result in a population crash (Fernando and Stevens 2011). Recent data from the Indian Ocean Tuna Commission database covering the time period of 2012 - 2014 indicate that over 2,400 mt of giant manta ray were recorded caught by the Sri Lankan gillnet and longline fleets primarily engaged in artisanal fishing. This amount is almost double the 1,413 mt total catch that was reported in Clarke and Indian Ocean Tuna Commission Secretariat (2014) by both Sri Lanka and Sudan fleets from a time period that was more than twice as long (2008-2013). Using the maximum observed weight of giant manta ray in the Indian Ocean (2,000 kg; which was described as “unusually large” (Kunjipalu and Boopendranath 1982)), this translates to a minimum of around 400 giant manta rays caught annually in recent years by Sri Lankan fishing fleets.

In India, mobulids are landed as bycatch during tuna gillnetting and trawling operations and are auctioned off for their gill plates, while the meat enters the local markets. Historical reports (from 1961 – 1995) indicate that the species was only sporadically caught by fishermen along the east and west coasts of India, likely due to the fact that the species was rarely found near the shore (Pillai 1998). However, based on available information, it appears that landings have increased in recent years, particularly on the southwest coast. In a snapshot of the Indian tuna gillnet fishery, Nair et al. (2013) provides evidence of the significant number of mobulids being taken off the coast of Vizhinjam, Kovalam and Colachel, documenting over 1,300 mobulids (50 t) that were landed by fishermen over the course of only 7 days. Of these mobulids, 5 individuals were identified as giant manta rays. For the years 2003 and 2004, Raje et al. (2007) reported 647 tons of giant manta rays from the southwest coast of India by the trawl fisheries. The significant increase in landings since the mid-1990s is likely due to the demand for the species’ gill rakers, with giant manta ray gill plates characterized as “First Grade” and fetching the highest price at auction at the Cochin Fisheries Harbour (Nair et al. 2013).

While not as abundant on the east coast of India, the species is still occasionally landed as bycatch, primarily in gillnet gear. Raje et al. (2007) documented 43 tons of giant manta rays in 2003 and 2004 and Rajapackiam et al. (2007) reported the landing of 3 mature individuals in 2006 at the Chennai fishing harbor.

In Australian waters, giant manta rays were identified as potential bycatch in the Commonwealth Skipjack Tuna Fishery and Western Tuna and Billfish Fishery. However, in a sustainability assessment of these fisheries, Zhou et al. (2009) determined that the current fishing effort poses a low risk to many non-target species caught within this fishery, including manta rays. This is likely due to the minimal spatial overlap between the fishing effort and the species’ distribution, with the fraction of distribution area within the fishery area of operation equating to <0.005 for both fisheries (Zhou et al. 2009). Overall, Simpfendorfer (2014) states that there are no data to suggest that giant manta rays are caught with any frequency or retained in Australian fisheries.

Overall, given that the majority of observed declines in landings and sightings of manta rays originate from this portion of their range, additional pressure on these species through bycatch mortality may have significant negative effects on local populations within this region. This is particularly a risk for the giant manta ray, which appears to be the species more frequently observed in the fisheries catch and bycatch, with this pressure already contributing to declines in the species (of up to 95%) throughout many areas (i.e., Indonesia, Philippines, Sri Lanka, Thailand, Madagascar) within this portion of its range. Given the high market prices for manta ray gill plates, the practice of landing the species as valuable bycatch will likely continue through the foreseeable future.

In the Western Pacific fisheries, *Manta* spp. are rarely reported in the bycatch. In the tropical tuna purse seine fisheries, Hall and Roman (2013) note that *M. japonica* represents the most abundant mobulid in the fishery bycatch. Analysis of the catch of Western and Central Pacific Fisheries Commission (WCPFC) purse seine and longline fisheries from 1995-2015 (based on observer data) showed that giant manta rays are rarely caught (Tremblay- Boyer and Brouwer

2016). In purse seine sets, the species is observed at a rate of 0.0017 individuals per associated set (sets made around a FAD) and 0.0076 individuals per unassociated set (sets on free swimming schools of tuna) (Tremblay- Boyer and Brouwer 2016). The available standardized purse seine CPUE data from the Western and Central Pacific Ocean show strong reporting bias trends (as observer reporting in the purse seine fisheries to species-level became more prevalent after 2008), and, therefore, are not particularly useful for accurately assessing abundance trends (Tremblay-Boyer and Brouwer 2016).

Peatman et al. (2018), summarize observer data from 2003 to 2017 to estimate the catch and catch composition of the longline fisheries of the western and central Pacific Ocean. Observer coverage levels in the region are generally less than 5% and observer coverage can be expressed in a variety of units (e.g. trips with observers on board, hooks with observer onboard, hooks observed, Peatman et al. 2018). Observer coverage over the whole Convention Area (Figure 43) tends to be consistent from 2003 – 2010 (1 to 1.5%) before reaching a maximum of ~4.5% in 2013 and then varying between 2 and 4% up to 2017 (Peatman et al. 2018). Since the U.S. is a cooperating commission member, the data from the Hawaii deep and shallow set longline fisheries is included in this summarized observer data report.

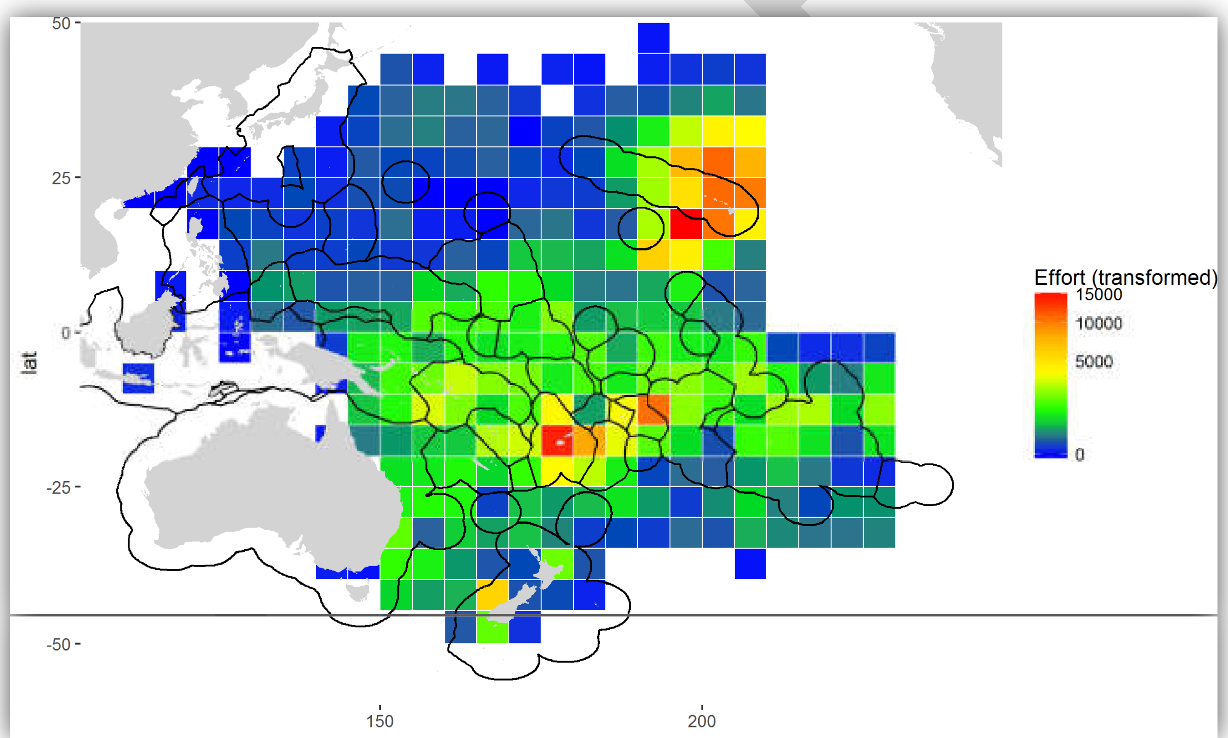


Figure 43. Observed effort in number of hooks (square root transformed) for longliners during the 2003-2016 time period in the WCPFC convention area.

Longline observers record catch data specific to each individual caught. As such, the natural catch unit for the estimation of catches is numbers of individuals. Currently, the observers are not instructed to distinguish between mantas and mobulids. In Figure 44, the total number of manta and mobula ray interactions documented by observers from 2003-2017 totals 1,800.

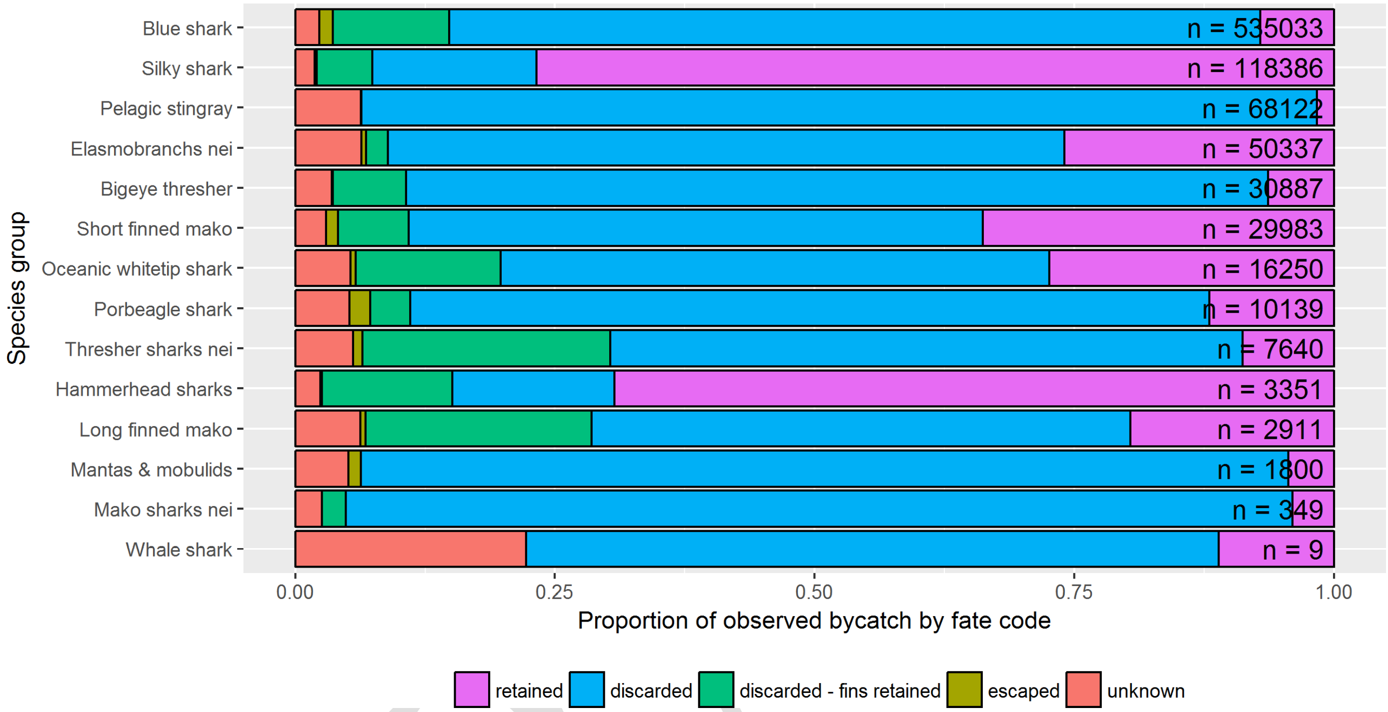


Figure 44. Recorded fate of observed sharks and rays catch by species/species group, as a proportion of total observed catch (number of fish) for the species/species group in the longline fisheries. The number of records is provided (Source: Peatman et al. 2018).

As evident in the above figure, very few mantas or mobulas were retained, the majority captured were discarded. In Figure 45 below, of the mantas and mobulas discarded, the majority were discarded either alive-healthy-injured or alive-unknown. Many were discarded in an unknown condition while a smaller proportion were discarded alive-dying or dead.

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

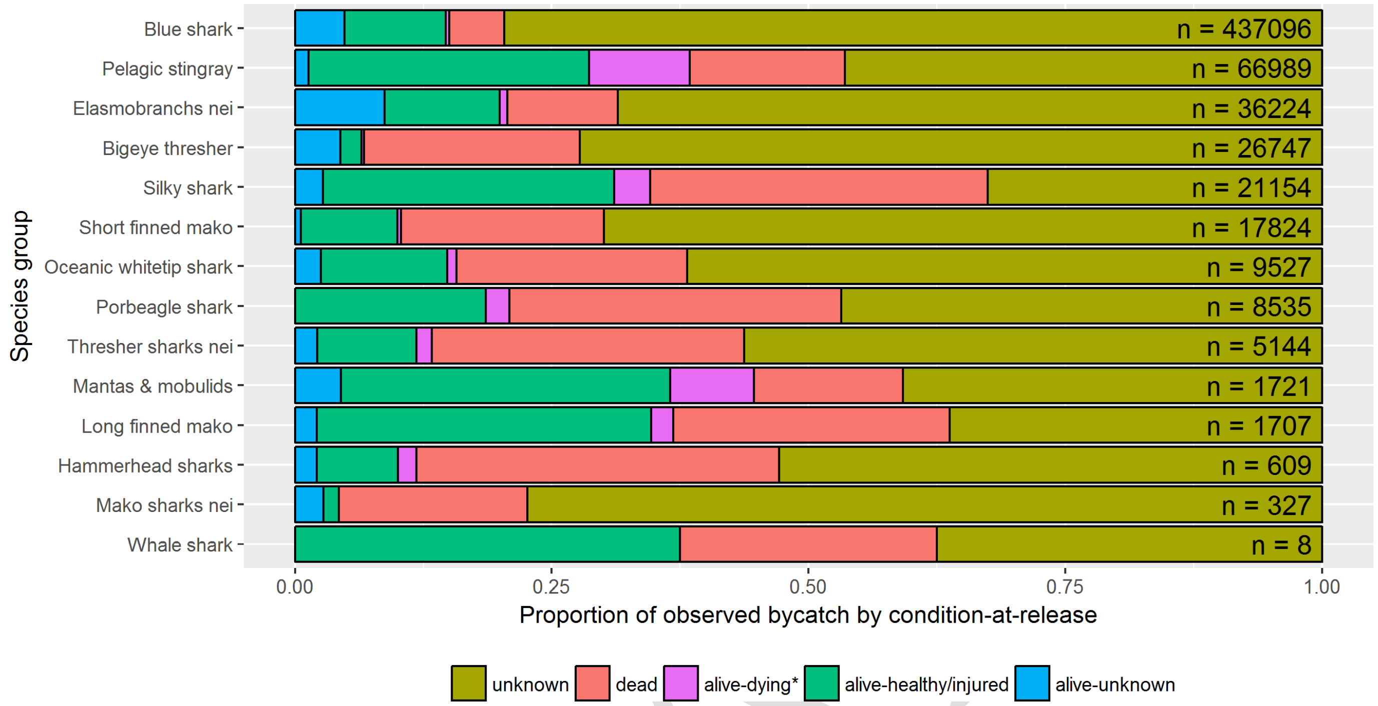


Figure 45. Recorded condition at release of observed sharks and rays catch by species/species group, as a proportion of total observed catch (number of fish) for the species/species group in the longline fisheries. The number of records is provided for each species/group. Note – alive-dying* is individuals that alive but considered unlikely to survive. Source: Peatman et al. 2018).

Manta ray bycatch has also been identified in a few fisheries operating in the South and Central Pacific. Manta rays have been identified in U.S. bycatch data from fisheries operating primarily in the Central and Western Pacific Ocean, including the U.S. tuna purse seine fisheries, the Hawaii-based HI DSLL fisheries for tuna, Hawaii-based shallow-set longline fisheries for swordfish and the American Samoa pelagic longline fisheries. Giant manta ray bycatch in the U.S. tuna purse seine fishery totaled 1.69 mt in 2015 (Secretariat of the Pacific Community, unpublished data, 2016). Observer data (20% coverage) for the Hawaii-based deep set longline fishery recorded 56 interactions with giant manta rays between 2004-2018 (NMFS Observer Program, unpublished data). Observer data (100% coverage) for the Hawaii-based shallow set longline fishery recorded 21 interactions with giant manta rays between 2004-2018 (NMFS Observer Program, unpublished data). Lastly, since 2010, there have been 17 giant manta rays recorded as bycatch in the American Samoa longline fishery by the observer program. (NMFS Observer Program, unpublished data).

Overall, there is little evidence of significant fishing pressure on giant manta ray throughout the Central or South Pacific. In the Eastern Pacific, giant manta rays are frequently reported as bycatch in the purse seine fisheries; however, identification to species level is difficult, and, as such, most manta and devil ray captures are pooled together (Hall and Roman 2013). Based on reported giant manta ray catch to the Inter-American Tropical Tuna Commission (IATTC), including available national observer program data, an average of 135 giant manta rays were estimated caught per year from 1993-2015 in the Eastern Pacific purse seine fishery by IATTC vessels (Hall unpublished data). Bycatch per set ranged from 0.001 individuals (in log associated sets) to 0.027 individuals (in school associated sets) (Hall unpublished data).

Bycatch of manta rays in the Eastern Pacific has also been reported from the U.S. and Peru. In U.S. west coast fisheries, the giant manta ray is occasionally observed as bycatch in the California drift gillnet fishery targeting swordfish and threshers, but in low numbers and only during El Niño events. In fact, from 1990 – 2006, only 14 giant manta rays were observed caught, with 36% released alive. Estimated (extrapolated) catch for the entire period was 90 individuals (95% CI: 26 – 182; CV = 0.48) (Larese and Coan 2008). Since 2010, no manta rays have been observed caught in the California drift gillnet fishery (NMFS 2018).

In 2005, interviews with northern Peruvian fishermen indicate that manta rays are rather frequently caught in gillnet gear, with 55% of respondents noting the giant manta ray as bycatch (Ayala et al. 2008). In 2005, gillnet boats comprised 33% of the total artisanal fishing fleet of Peru (Ayala et al. 2008). However, fishermen off Salaverry and Chimbote did not view manta rays as a commercially viable species. Additionally, Ayala et al. (2008) noted that catching manta rays is actually dangerous for the fishermen operating the smaller artisanal vessels, as the animals tend to cause nets to be lost and can also potentially sink the small boats (Ayala et al. 2008). Giant manta ray have also been reported as bycatch in the Peruvian merluza fishery, which uses mid-water trawls in 50-150 m depths (Stewart et al. 2016b), and in the small-scale Peruvian drift net fishery targeting primarily blue and short fin mako sharks (Alfaro-Shigueto et al. 2010).

Given the sustained fishing pressure on giant manta ray in the Eastern Pacific, particularly its susceptibility to the industrial tuna purse-seine fisheries operating near high productivity areas where the species is likely to aggregate, and evidence of subsequent population declines of up to 88%, it is likely that current fisheries-related mortality rates are a threat significantly contributing to the overutilization of the species in this portion of its range.

Overall, in many regions, the status of the population of the giant manta ray appears be declining, up to as much as 80% over the last 75 years, and >30% globally (Marshall et al. 2011a). Based on the best available scientific and commercial information, and after considering efforts being made to protect the giant manta ray, NMFS found that the giant manta ray is not currently endangered throughout its range (83 FR 2916). However, the giant manta ray is likely to become an endangered species within the foreseeable future throughout a significant portion of its range (the Indo-Pacific and Eastern Pacific portion). Without the members in that portion, the species would be likely to become in danger of extinction in the foreseeable future throughout all of its range.

4.2.6.5 Threats to the Species

Natural Threats

In terms of predation, manta rays are frequently sighted with non-fatal injuries consistent with shark attacks, although the prevalence of these sightings varies by location (Homma et al. 1999; Ebert 2003; Mourier 2012). In terms of fatal encounters, there are a couple of records of killer whales feeding on manta rays. In Papua New Guinea, Visser and Bonoccorso (2003) observed on two separate occasions killer whales fatally attacking and feeding on manta rays. Killer whales were also recorded preying on manta rays in the Galapagos Islands (Fertl et al. 1996). As discussed previously, given their large sizes, manta rays are assumed to have low natural predation rates after they reach maturity.

Anthropogenic Threats

As described above, giant manta rays are both targeted and caught as bycatch in fisheries worldwide (Miller and Klimovich 2016). Targeted fisheries for mantas have existed for decades and historically, the giant manta ray was exploited for meat, cartilage, and skin (Lawson 2016). However, driven by the international trade in gill plates, fisheries targeting mantas have expanded and pose a serious threat to the giant manta ray (CITIES 2013). Although the market for mobulid gill plates does not have a long history of traditional use (Croll et al 2015), the tonic of which gill plates are the essential ingredient, is advertised to prevent sickness by boosting the immune system and enhancing blood circulation (Lawson et al 2016), although there is no scientific evidence to support these claims. The growing market for gill plates has grown since the report of the first gill plate trade from the Philippines to China in the 1960's (Lawson et al 2016) and has become a lucrative trade market. For example, a mature giant manta ray can yield up to 7-kilograms of dried gills that retail in China for as much as US$500 per kilogram (Heinrichs et al 2011). Scientist speculate that as shark population's decline, the established shark fin trade networks will refocus their efforts to exploit the gill plate market (Heinrichs et al 2011).

Migrations into offshore environments with high fishing pressure make the giant manta ray vulnerable to risks from an array of fishing gear types including driftnets, gillnets, traps, trawls, long lines, and purse seines, (CITIES 2013, Stewart et al. 2016a) however, they are most susceptible to industrial purse-seine and artisanal gillnet fisheries (Croll et al 2015). Mobulids have been reported as bycatch in 21 small-scale fisheries and 9 large-scale fisheries (Croll et al 2015). Croll et al (2015) believe bycatch poses the most significant threat to mobulids, specifically the purse seine fishery given the reported bycatch, spatial distribution, and intensity of effort.

Purse seine fisheries in tropical waters target skipjack, yellowfin and bigeye tunas while in other regions, smaller tuna or tuna-like fish are targeted such as black skipjack, frigate and bullet tunas and bonito (Hall and Roman 2013). Purse seines are designed to catch schooling fish by surrounding them from both sides and underneath. Dependent on the tuna's detected behavior, the main ways a purse seine sets to encircle its catch are as follows: school sets are made when activity is at or near the surface of the water; dolphin sets are made when a group of dolphins are detected since yellowfin tuna are known to associate with the striped and common dolphin; and seamount and floating object sets are made when tuna are detected since they are known to associate with these ocean features (Hall and Roman 2013). Not surprisingly, the schooling sets have the greatest mobulid bycatch due to the shared preference of tunas and mobulids for oceanographic features that generate high biological productivity (Hall and Roman 2013, Croll et al 2015). The global tuna purse seine fishery is estimated to capture approximately 13,000 mobulids annually (Hall and Roman 2013).

Other threats to the manta species include vessel strikes and nearshore infrastructure including mooring, beach protection nets and aquaculture facilities (Croll et al 2015). Because manta ray aggregation sites are sometimes in areas of high maritime traffic, manta rays are at potential risk of being struck and killed by boats (Marshall et al. 2011a; Graham et al. 2012). Mooring and boat anchor line entanglement may also wound manta rays or cause them to suffocate (Deakos et al. 2011; Heinrichs et al. 2011). Manta rays only swim forward and must do so continuously to move water over their gills. When a line catches their body, gills, or fins, they roll backwards in an attempt to become free, however this maneuver entangles them further (MantaTrust 2018).

Photographs of mantas with injuries indicate that these injuries are consistent with boat strikes and line entanglements, and manta researchers report that they may affect manta ray fitness in a significant way (The Hawaii Association for Marine Education and Research Inc. 2005; Deakos et al. 2011; Heinrichs et al. 2011; Couturier et al. 2012; CMS 2014; Germanov and Marshall 2014; Braun et al. 2015), potentially similar to the impacts of shark or orca attacks. However, there is very little quantitative information on the frequency of these occurrences and no information on the impact of these injuries on the overall health of the population.

Manta rays may also suffer mortality in nets deployed to control sharks off the coasts off Australia and South Africa. In Australia, shark control nets are deployed off the east coast of Queensland and New South Wales (NSW). Since 2001, 194 manta rays (species not identified) have been observed caught in the Queensland nets, with around 52% released alive (https://data.qld.gov.au/dataset/shark-control-program-non-target-statistics-by-year). In New South Wales, only two Giant manta ray individuals were caught in 2014 by the shark control nets, both released alive. Prior years (2010-2013) reported no captures of manta rays (annual reports of the NSW Shark Meshing Program).

Due to their association with nearshore habitats (congregating at inshore cleaning sites, mating and occasionally feeding in shallow waters), manta rays are at elevated risk for exposure to a variety of contaminants and pollutants, including brevotoxins, heavy metals, and polychlorinated biphenyls. Many pollutants in the environment have the ability to bioaccumulate in fish species; however, only a few studies have specifically examined the accumulation of heavy metals in the tissues of manta rays (Essumang 2010; Ooi et al. 2015), with findings that are put in relation to human health risks and consumption of manta rays. For example, Essumang (2010) found platinum levels within giant manta ray samples taken off the coast of Ghana that exceeded UK dietary intake recommendation levels. While consuming manta rays may potentially pose a health risk to humans, there is no information on the lethal concentration limits of these metals or other toxins in manta rays. Additionally, at this time, there is no evidence to suggest that current concentrations of these environmental pollutants are causing detrimental physiological effects to the point where the species may be at an increased risk of extinction.

Plastics within the marine environment are also a threat to the manta ray species. Filter feeders such as the giant manta ray are particularly susceptible to ingesting high levels of microplastics (Germanov 2018) and being exposed to toxins (Worm et al. 2017), similar to those described above, 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. For example, in Thailand, where recent sightings data have identified over 288 giant manta rays (MantaMatcher 2016), mismanaged plastic waste is estimated to be on the order of 1.03 million tons annually, with up to 40% of this entering the marine environment (Jambeck et al. 2015). Approximately 1.6 million tons of mismanaged plastic waste is being disposed of in Sri Lanka, again with up to 40% entering the marine environment (Jambeck et al. 2015), potentially polluting the habitat used by the nearby Maldives aggregation of 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 tract (Germanov et al. 2018), they can harbor high levels of toxins and persistent organic pollutants and transfer these toxins to the animal once ingested (Worm et al. 2017).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 2015b; Germanov 2015a).

Changes in climate and oceanographic conditions, such as acidification, are also known to affect zooplankton structure (size, composition, and diversity), phenology, and distribution (Guinder and Molinero 2013). As such, the migration paths and locations of both resident and seasonal aggregations of manta rays, which depend on these animals for food, may similarly be altered (Australian Government 2012; Couturier et al. 2012). Because manta rays are migratory and considered ecologically flexible (e.g., low habitat specificity), they may be less vulnerable to the impacts of climate change compared to other sharks and rays (Chin et al. 2010). However, as 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 ray. 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 impact 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.

Manta ray-focused tourism is far more sustainable when compared to consumptive uses such as fishing, however, if wildlife-centered marine ecotourism is not well-managed it has the potential to have unintended consequences that could harm the species (O'Malley et al. 2013). Manta ray-focused tourism generates significant economic benefits worldwide with a global estimate of direct economic impact (estimated tourist expenditures on dives and associated spending on lodging, food, local transportation) totaling $140 million per year (O'Malley et al. 2013). The value of the marine ecotourism industry is substantially greater than that of fisheries that incidentally capture and target manta rays (O'Malley et al. 2013). Manta ray-focused tourism industries are established in several countries, although, formal ecotourism management programs to protect the species are rare (Venables et al. 2016). According to a case study from the Ningaloo Marine Park, Western Australia, behavioral responses exhibited by manta rays to in-water tourism interactions including avoidance, alterations in swimming speed, direction and abrupt movements, and behavioral state (termination of feeding and departure from cleaning stations) (Venables et al. 2016). Although it is unknown if these responses will have a long-term biological effect on the species, this study indicates that interactions with tourists can disrupt behavior that serve critical biological functions such as feeding and cleaning (Venables et al. 2016). Additional studies are needed to evaluate the short-term, long-term and cumulative impacts of manta ray-focused tourism to evaluate the potential threat to the specie's survival and to effectively manage in-water tourism.

4.2.6.6 Protections and Conservation Efforts

In many portions of the species' range, current U.S. fishery regulations as well as U.S. state and territory regulations prohibit the retention of manta rays by persons under U.S. jurisdiction. For example, in the Eastern Pacific Ocean, U.S. commercial fishing vessels are prohibited from retaining on board, transshipping, landing, storing, selling, or offering for sale any part or whole carcass of a mobulid ray caught by vessel owners or operators in the Inter-American Tropical Tuna Convention Area (81 FR 50401, August 1, 2016). In Guam, it is unlawful for any person to possess, sell, offer for sale, take, purchase, barter, transport, export, import, trade or distribute ray parts, unless for subsistence, traditional, or cultural sharing purposes (Article 1, Chapter 63 of Title 5, Guam Code Annotated, Sec. 63114.2), and in the Commonwealth of the Northern Mariana Islands, it is illegal to feed, take, possess, sell, purchase, barter, offer to sell, purchase or barter, transport, export or import, any ray, alive or dead, or any part thereof (Pub. L. 15-124). 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, Western Australia, and New Zealand (Miller and Klimovich 2017; Table 32. Legal Protection Measures for Mana spp. Source: CITES, CoP16 Prop. 46 (Rev.2) Annex IX). 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. As mentioned previously in the status section, 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). In addition, it is unlikely that fishing for mantas will decrease in the Philippines regardless of any restrictions 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).

Table 32. Legal Protection Measures for Mana spp. Source: CITES, CoP16 Prop. 46 (Rev.2) Annex IX

Location	Species	Legal Protection / Conservation Measure
International		
CMS Signatories	*M. birostris*	Convention on the Conservation of Migratory Species of Wild Animals (CMS), Appendix I and II, 2011
Regional		
Micronesia: Federated States of Micronesia, Guam, Mariana Islands, Marshall Islands, Palau	All ray species	Micronesia Regional Shark Sanctuary Declaration to prohibit possession, sale, distribution and trade of rays and ray parts from end 2012
National		
Ecuador	*M. birostris*	Ecuador Official Policy 093, 2010
European Union	*M. birostris*	Article 1 of COUNCIL REGULATION (EU) No 692/2012
Maldives	*Manta spp.*	Exports of all ray products banned 1995
Mexico	All ray species	NOM-029-PESC-2006 Prohibits harvest and sale
New Zealand	*M. birostris*	Wildlife Act 1953 Schedule 7A (absolute protection)
Philippines	*M. birostris*	FAO 193 1998 Whale Shark and Manta Ray Ban
Yap (FSM)	*Manta spp.*	Manta Ray Sanctuary and Protection Act 2008
State		
Florida, USA	Genus *Manta*	FL Admin Code 68B-44.008 – no harvest
Guam, USA Territory	All ray species	Article 1, Chapter 63 of Title 5, Guam Code Annotated, Sec. 63114.2
Commonwealth of the Northern Mariana Islands	All ray species	Public Law No. 15-124
Hawaii, USA	*Manta spp.*	HI Rev Stat Sec. 188-39.5
Raja Ampat Regency,	*Manta spp.*	Shark and Ray Sanctuary Bupati Decree 2010

Indonesia		

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 (due to its uncertain effects at reducing the threats of foreign domestic overutilization and inadequate regulations, and unknown post-release mortality rates from bycatch in industrial fisheries), 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 has 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. The agreement for sharks, which includes the giant manta ray, is a Memoranda of Understanding with 41 Signatories: 39 national governments and the European Union. The objectives of this Memoranda of Understanding include: ensuring that directed and non-directed fisheries are sustainable, protect critical habitats, migration corridors and critical life stages, increase public awareness of threats and participation of conservation activities (CMS 2014).

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 (http://www.mantatrust.org/), the Marine Megafauna Foundation (http://www.marinemegafauna.org/), the Manta Pacific Research Foundation (http://www.mantapacific.org/) and MantaWatch (http://mantawatch.com). Others are driven by the countries whose economies largely depend on manta ray tourism (Erdmann 2014).

The efforts by these organizations to educate the public, such as through awareness campaigns, could eventually lead to decreases in the demand for manta ray products. For example, Lawson et al. (2016), citing unpublished data, noted an 18-month awareness- raising campaign conducted in 2015 in Guangzhou, China that seemed to indicate a level of success in decreasing consumer demand of gill rakers, which, in turn, decreased the interest of traders to carry gill plates in the future. While more monitoring of trade and consumer behavior is required to evaluate the success of these efforts, it may indicate that awareness-raising campaigns could be successful tools for influencing customer behavior and, as demand reduction is viewed as a potential avenue to indirectly reduce fishing pressure on manta rays, these campaigns may ultimately help decrease the main threat to the species (Lawson et al. 2016).

4.2.6.7 Summary of the Status of the Giant Manta Ray

In this section of this biological opinion, we explained that the population of the giant manta ray is highly fragmented and sparsely distributed which contributes to the lack of information on this species. It is one of the least understood of the marine mega vertebrates. Anecdotal and citizen science data indicates the giant manta ray's population appears to be declining, up to as much as 80% in regions where giant manta rays are actively targeted or caught as bycatch and at least 30% globally (Marshal et al. 2011a).

Generally speaking, giant manta ray's face various threats throughout each stage of their respective life cycles, even as early as an embryo. Given the migratory nature of the species, population declines in waters where mantas are protected have also been observed and attributed to overfishing of the species in adjacent areas within its large home range.

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 46, 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 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 population of the giant manta ray.

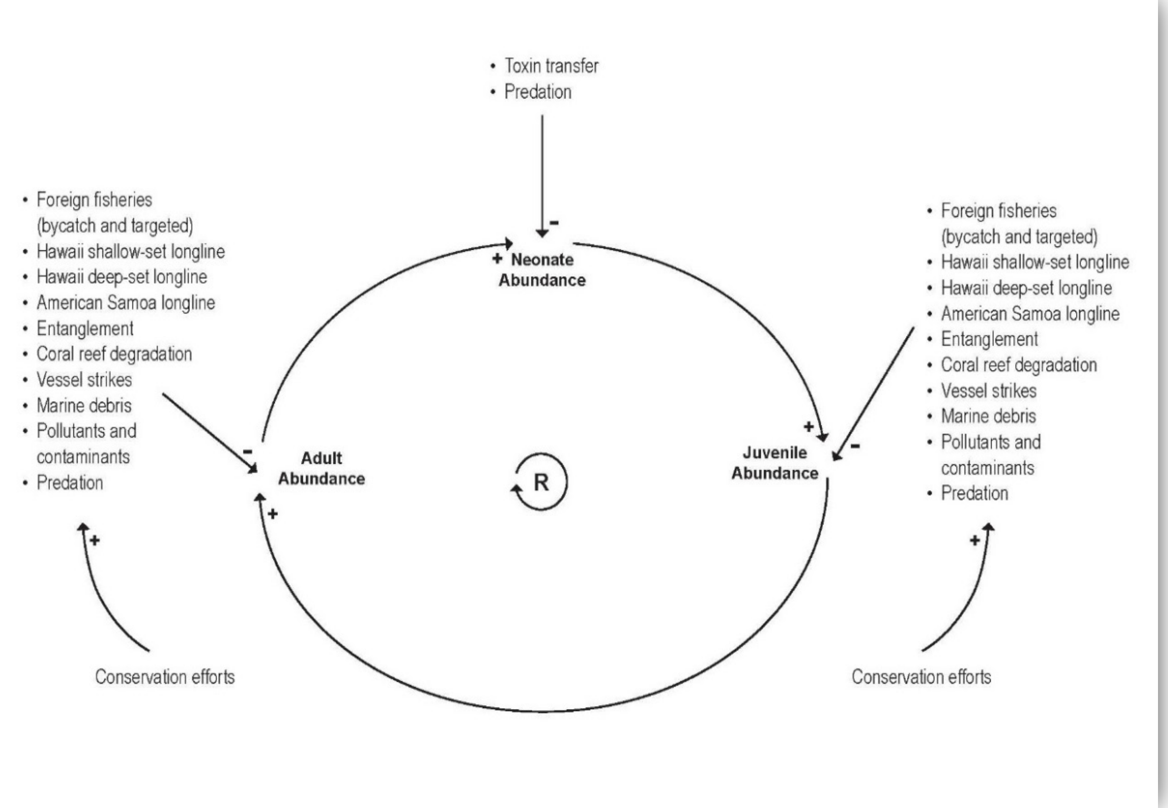


Figure 46. Giant manta ray casual loop diagram depicting various stressor-response relationships for each life cycle stage.

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 2015).

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; Dulvy et al. 2014, Couturier et al. 2012; Dulvy et al. 2014). Despite the best efforts of protections and conservation measures, the overall trend of the giant manta ray continues to decline.

4.2.7 Guadalupe Fur Seal

4.2.7.1 Distribution

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 (Figure 47). However, the species may be exhibiting re-expansion into their previous historical range due to various hypothesized variables (Hanni et al. 1997; Ethnier 2002; Lambourn et al. 2009; Lambourn 2015; Elorriaga-Verplanken et al. 2016a; Carretta et al. 2017; Warlick et al. 2018). It is the only species of the *Arctocephalus* genus that occurs north of the equator.

Along the U.S. west coast, strandings occur almost annually in California waters and animals are increasingly observed in Oregon and Washington waters. In 2015-2016, Guadalupe fur seal strandings totaled approximately 175 animals along the coast of California. Individuals have stranded or been sighted inside the Gulf of California and as far south as Zihuatanejo, Mexico (Hanni et al. 1997 and Aurioles-Gamboa and Hernadez-Camacho 1999). 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; Figure 48). Lambourn et al. (2009) suggests the northernmost location this species has been documented was Katcemak Bay in Alaska.

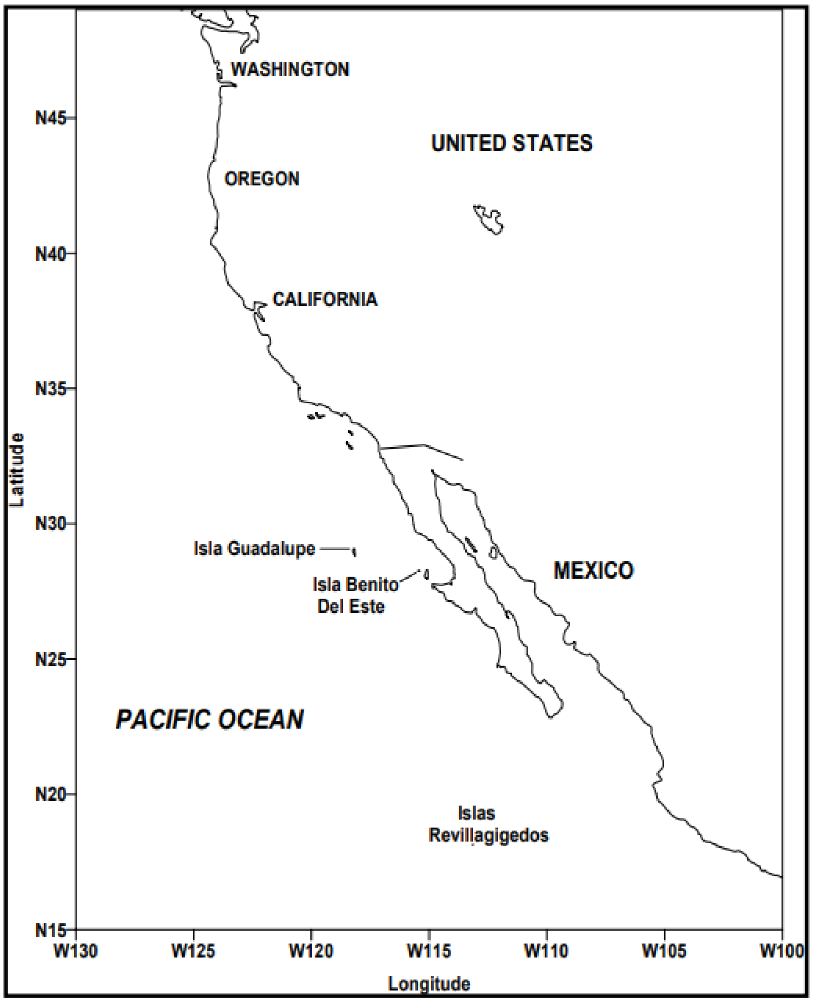


Figure 47. Geographic range of the Guadalupe fur seal, showing location of the two rookeries at Isla Guadalupe and Isla Benito Del Este. Source: Carretta et al. 2017

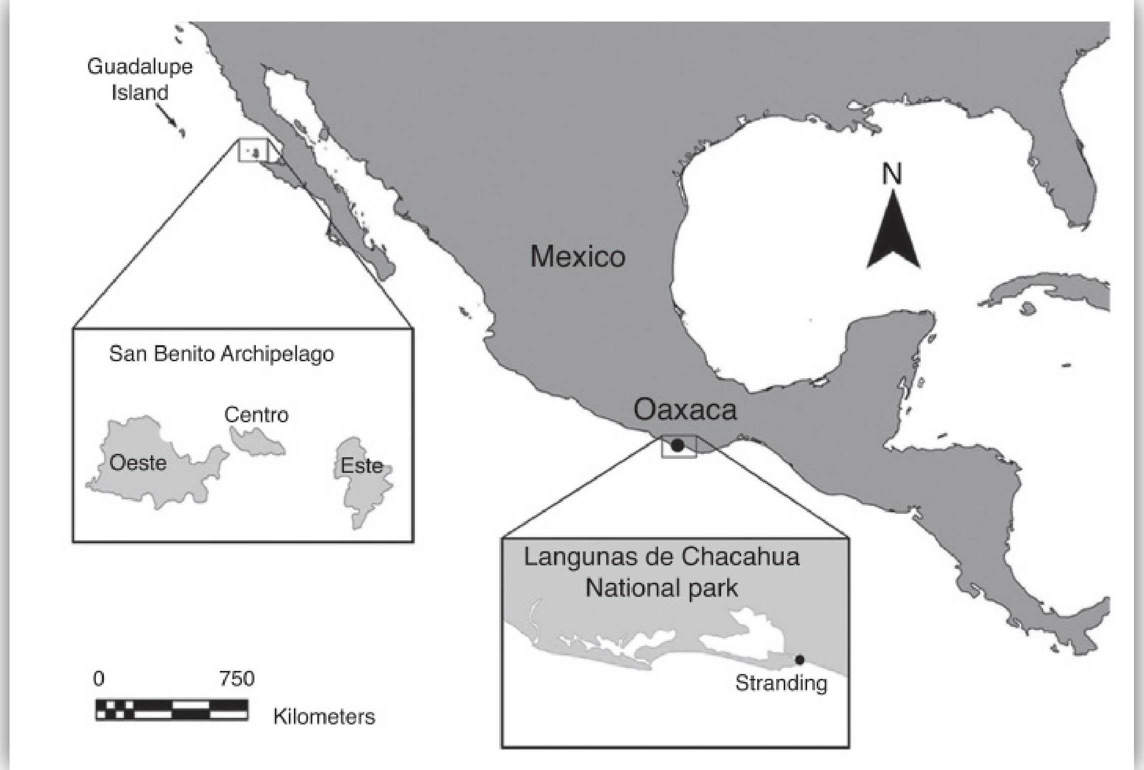


Figure 48. Distribution of Guadalupe fur seals, Arctocephalus townsendi, indicating the breeding (Guadalupe Island) and recolonization (San Benito Archipelago) sites, as well as the atypical stranding in Parque Nacional Lagunas de Chacahua in Oaxaca, Mexico. Source: Villegas-Zurita et al. 2015.

Guadalupe fur seals that stranded in central California and treated at rehabilitation centers were fitted with satellite tags and documented to travel as far north as Graham Island and Vancouver Island, British Columbia, Canada (Norris et al. 2015). Some satellite-tagged animals traveled far offshore outside the U.S. EEZ to areas 700 nmi west of the California/Oregon border.

Feeding and Diving Behavior

Many studies have been conducted to determine diet preferences of the Guadalupe fur seal throughout the species range. Review of the literature reveals cephalopods appear to be the main source of the Guadalupe fur seals diet across all studies (Gallo-Reynoso 1994; Hanni et al. 1997; Aurioles-Gamboa and Camacho-Rios 2007; Hernandez-Montoya 2009; Pablo-Rodriguez 2009; Esperon Rodriguez and Gallo-Reynoso 2012a, 2012b, 2013; Amador-Capitanachi et al. 2017; Juarez-Ruiz et al. 2018). The remainder of the diet consists of approximately 11 to 14 species of fish, various crustaceans, and even seaweed with variations between studies, locations, colonies, and expected habitat usage.

Most studies are based on scat analysis while Amador-Capitanachi et al. (2017) and Juarez-Ruiz et al. (2018) also included stable isotope analysis in addition to scat analysis. These two studies assess differences related to intraspecific competition either between two island populations or between the individuals of one island, respectively. As with most dietary studies, caveats exists regarding sample size, seasonality, age-class, habitat use, etc. Furthermore, variations in diet composition occur due to habitat structure. For instance, the colony at Isla Guadalupe must dive deeper to reach preferred pelagic squid species then those animals located at San Benito, due to the continental shelf (Gallo-Reynoso and Esperon-Rodriguez 2013). A shift in diet to rudderfish (*Girella nigricans*) was reported by Segeul et al. (2018), a species not previously reported in any diet analysis. However this was likely due to a shift in habitat use and vertical migration to procure food due to adverse ocean conditions (Segeul et al. 2018).

Guadalupe fur seals typically forage within the top 50m of the water column based on the pelagic and coastal squid species which comprise their diet (Gallo-Reynoso 1994; Aurioles-Gamboa and Camacho-Rios 2007; Gallo-Reynoso and Esperon-Rodriguez 2013). Foraging occurs beginning at dusk and through the night as squid migrate vertically in the water column (Gallo-Reynoso 1994; Gallo-Reynoso and Esperon-Rodriguez 2013).

4.2.7.2 Population Dynamics

Structure

The size of the population prior to the commercial harvests of the 19th century is not known, but estimates range from 20,000 to 100,000 animals (Fleischer 1987). Surveys conducted between 2008 and 2010 resulted in a total estimated population size of approximately 20,000 animals, with ~17,500 at Isla Guadalupe and ~2,500 at Isla San Benito (Garcia-Capitanachi 2011, Aurioles-Gamboa 2015). These estimates are corrected for animals not seen during the surveys. The population is considered to be a single population because all are recent descendants from one breeding colony at Isla Guadalupe, Mexico.

All the individuals of the population cannot be counted because all age and sex classes are never ashore at the same time and some individuals that are on land are not visible during the census. Direct counts of animals at Isla Guadalupe and Isla San Benito during 2010 resulted in a minimum of 13,327 animals and 2,503 animals respectively, for a minimum population size of 15,830 animals (García-Capitanachi 2011).

Counts of Guadalupe fur seals have been made sporadically since 1954. Records of Guadalupe fur seal counts through 1984 were compiled by Seagars (1984), Fleischer (1987), and Gallo-Reynoso (1994). The count for 1988 was taken from Torres et al. (1990). More recent counts from 1977-2010 are summarized in García-Capitanachi (2011). Also, the counts that are documented in the literature generally provide only the total of all Guadalupe fur seals counted (i.e., the counts are not separated by age/sex class). The counts that were made during the breeding season, when the maximum number of animals are present at the rookery, were used to examine population growth (Gallo-Reynoso 1994, García-Capitanachi 2011). These data indicate that Guadalupe fur seals are increasing at an average annual growth rate of 10.3% (Figure 49).

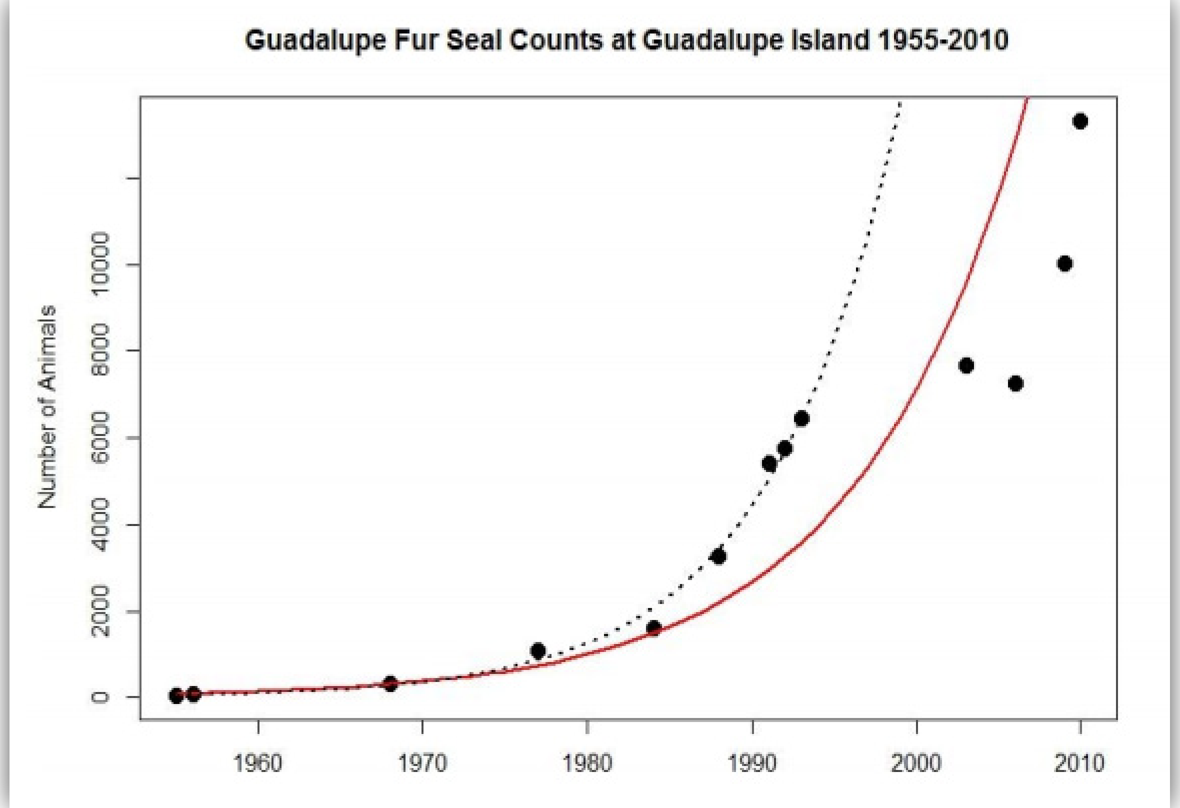


Figure 49. Counts of Guadalupe fur seals at Guadalupe Island Mexico, and the estimated population growth curves derived from counts made during the breeding season. Direct counts of animals are shown as black dots. An estimated annual growth rate of 13% is based on counts made between 1955 and 1993 (black dashed line). The estimated growth rate over the period 1955-2010 is approximately 10% annually (solid red line). Source: Carretta et al. 2017

Diversity

Guadalupe fur seals are considered medium sized otariids and display clear sexual dimorphism (Peterson et al. 1968; Gallo-Reynoso and Figuerosa-Carranza 1996; Sanvito et al. 2014). Like most otariids, Guadalupe fur seals sexually mature at approximately four to five years of age. Gallo-Reynoso and Figuerosa-Carranza (1996) reviewed the available literature and compared size and weight data to animals they were able to capture. Animals in their study were slightly larger than previously reported metrics (Fleischer 1987; King 1983; Reeves et al. 1992) with territorial males having a mean length of 219.2 ± 18.8 cm (range 190-245 cm, n=12), subadult males were 151.7 ± 5.6 cm (range 147.7 – 155.6 cm, n=2), and adult females were 148.2 ± 8 cm (range 137-165 cm, n=14) (Gallo-Reynoso and Figuerosa-Carranza 1996). Mean weight for territorial males range upwards of approximately 125 kg with females weighting in at 49.1 ± 5.7 kg (range 40-55 kg, n=14) as reported by Gallo-Reynoso and Figuerosa-Carranza (1996). Jefferson et al. (1993) reports males may reach up to between 160 and 170 kg. Later, Gallo-Reynoso et al. (2008) reported three multiparous females with a mean weight of 52.5 ± 1.3 kg, and three younger females with a mean weight of 41.2 ± 1.3 kg. We therefore expect males may range upwards of 400 pounds with females roughly 110 pounds.

Guadalupe fur seals pup and breed mainly at Isla Guadalupe, Mexico. In 1997, a second rookery was discovered at Isla Benito del Este, Baja California (Maravilla-Chavez and Lowry 1999) and a pup was born at San Miguel Island, California (Melin and DeLong 1999). Since 2008, individual adult females, sub-adult males, and between one and three pups have been observed annually on San Miguel Island (NMFS, unpublished data). The population at Isla Benito del Este is now well-established, though very few pups are observed there. Population increases at Isla San Benito are attributed to immigration of animals from Isla Guadalupe (Aurioles-Gamboa et al. 2010, Garcia-Capitanachi 2011).

In general, the breeding season extends from June through August (Gallo Reynoso 1994; Juarez-Ruiz et al. 2018), although historical literature included accounts in May (Peterson et al. 1968). Adult males typically return to the colonies in early June, while sub-adult males arrive earlier and leave later (Gallo-Reynoso 1994). Like most pinnipeds, males create and defend territories for breeding purposes. Sub-adult males roam throughout the colony during the breeding season providing competition to adults whom are defending territories.

Guadalupe fur seal males are polygamous and may mate with up to 12 females during a single breeding season. Adult females arrive to the colonies in early June, giving birth a few days later (Gallo-Reynoso 1994). Pups are born from early June through early July, with a peak in late June. Adult females will usually mate approximately a week after giving birth to her pup. Weaning occurs at approximately 9 months.

Status

The Guadalupe fur seal is listed as threatened, which automatically qualifies this stock as "depleted" and "strategic" stock under the MMPA. There is insufficient information to determine whether fishery mortality in Mexico exceeds the potential biological removal for this stock but given the observed growth of the population over time, this is unlikely. The total U.S. fishery mortality and serious injury for this stock (≥3.2 animals per year) is less than 10% of the calculated PBR for the entire stock, but it is not currently possible to calculate a prorated PBR for U.S. waters with which to compare serious injury and mortality from U.S. fisheries. Therefore, it is unknown whether total U.S. fishery mortality is insignificant and approaching zero mortality and serious injury rate. The population is growing at approximately 10% per year.

4.2.7.3 Threats to the Species

Natural Threats

Studies relating to alterations in climate dynamics have begun to show several factors which are impacting the Guadalupe fur seal. Variations in ocean currents, ocean temperatures with subsequent modifications in food chain dynamics, foraging strategies, interspecific competition, and consequential malnutrition have been documented thus far (Trillmich et al. 1991; Elorriage-Verplancken et al. 2016a, 2016b; Pablo-Rodriguez et al. 2016; NMFS 2017; Juarez-Ruiz et al. 2018). Pup mortality appears to be high from negative effects caused by El Nino events and tropical cyclones (Fleischer 1987; Trillmich et al. 1991; Gallo-Reynoso 1994; Aurioles-Gamboa 2015).

Two unusual mortality events have been declared for this species. The first event in the Pacific Northwest was summarized by Lambourn et al. (2009) occurred from June of 2007 lasting through December 2009. Through this unusual mortality event, a total of twenty-nine Guadalupe fur seals stranded throughout Washington and Oregon states (Lambourn et al. 2009). The second event was declared in California by NOAA beginning in January of 2015 and is still occurring at the time of this writing in mid-2018. Stranding's from both events are mostly weaned pups and juveniles which are displaying signs of malnutrition with secondary bacterial and parasitic infections. One study, conducted by Segeul et al. (2018) describes an unusual case of intravascular infection by a sea lion lungworm, *Parafilaroides decorus,* from a stranded male Guadalupe fur seal yearling. Shifts in diet composition to an intermediate host species of fish may have increased the exposure rates of this nematode to Guadalupe fur seals and was attributed to warming sea surface temperatures in the region as described by Elorriaga-Verplanken et al. (2016b) (Segeul et al. 2018). Otherwise, stranding's appear to be seasonal peaking between April and June of each year and the second event appears to coincide with the 2015-2017 California sea lion unusual mortality event. Figure 50 illustrates the number of stranding's from 2010 through 2018 in California as reported by NOAA to-date. Additionally, Warlick et al. (2018) reports 167 total stranding's from the period of 2002 to 2016 with 60% occurring on the Oregon coast, 34% for the Washington coast, and 5% for inland waters of Washington State (i.e. mouth of the Columbia River).

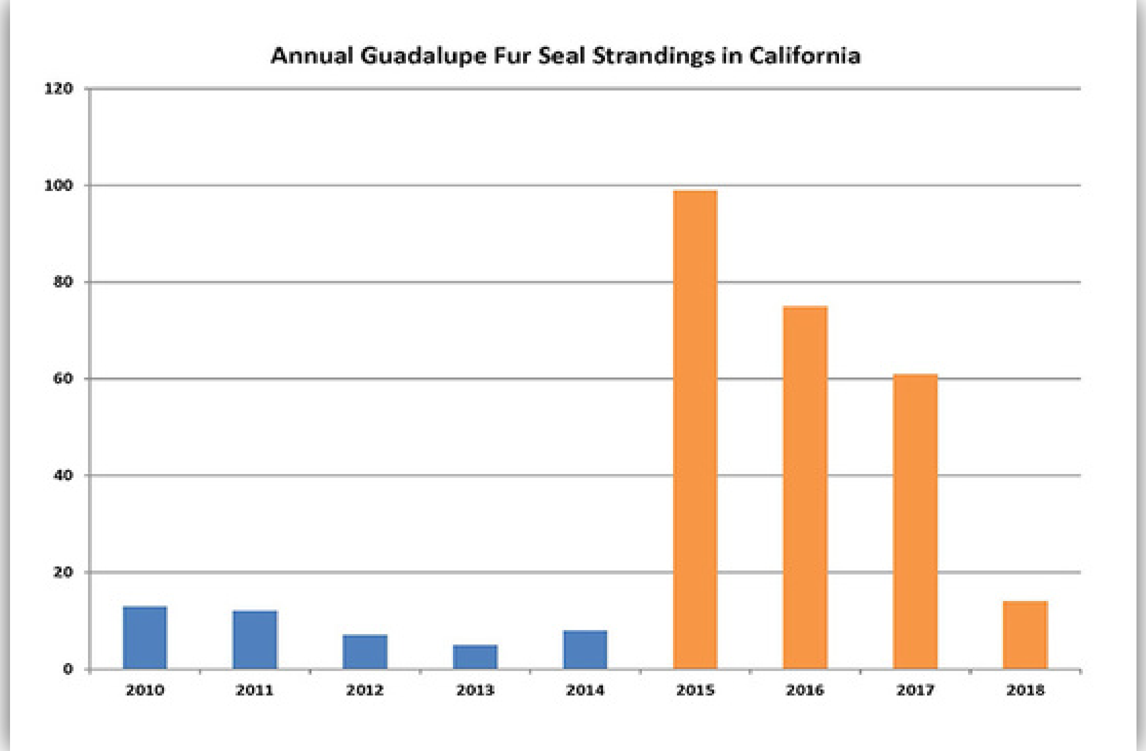


Figure 50. Annual Guadalupe fur seal strandings in California from 2010-2018.

Natural predation of Guadalupe fur seals may occur from the great white shark (*Carcharodon carcharias*) and killer whale (*Orcinus orca*) although quantification of these events is difficult (Domeier and Nasby-Lucas 2007; Domeier 2012; Aurioles-Gamboa 2015; Aurioles-Gamboa and Trillmich 2017). However, it has been noted that great white's summer migration, for at least the males, back to Guadalupe Island corresponds with the Guadalupe fur seal pupping season (Pierson 1978; Domeier and Nasby-Lucas 2008). Hoyos-Padilla (2009) documented what is believed to be the first, and only, observed predation event of a Guadalupe fur seal by a great white shark to date. Jamie-Rivera et al. (2014) suggest that pinnipeds at Isla Guadalupe are important prey for the great white shark from stable isotopes studies using dermis samples from various species. However species specific data suggesting Guadalupe fur seal ingestion or a proportion of this species compared to other pinniped species in the region is not apparent from the data.

Anthropogenic Threats

There were 16 records of human-related deaths and/or serious injuries to Guadalupe fur seals from 2010-2014 which are accredited to commercial fisheries and other unidentified fisheries (Carretta et al. 2016a). Three fisheries related stranding's occurred during this interval and were attributed to human induced gunfire, entanglements in gillnet of unknown origin and within marine debris. Thus the average annual observed human caused mortality and serious injury of Guadalupe fur seals for 2010-2014 is 3.2 animals annually (16 animals /5 years) and is considered the most current assessment.

The California gillnet fisheries have not observed Guadalupe fur seals within the fishery between 1990 and 2014 (Julian and Beeson 1998, Carretta et al. 2004, Carretta et al. 2017). According to Carretta et al. (2017) the observed human-caused mortality and serious injury for this stock likely represents only a proportion of the impacts to the species since documentation of these events does not always occur. Furthermore, correction factors to account for undetected mortality and injury are not currently available for pinnipeds along the U.S. west coast. Additionally, since this assessment was completed, Warlick et al. (2018) describe 167 stranding events throughout Oregon and Washington State with 67%, or approximately 112 individuals, attributed to fisheries interactions. It should also be noted that Warlick et al.'s (2018) study did not identify the exact fisheries where these interactions occurred, but summarized data from 15 regional response networks from within these states. Since the Stock Assessment Report was published, additional information has been presented by Hernandez-Camacho et al. (2017) specifically Magdalena Island off of Baja, California in Mexican waters where a productive squid fishery exists. A total of 59 individuals stranded between 2003 and 2015, and were confirmed either by onsite observers or by examination of skull features in a laboratory (Hernandez-Camacho et al. 2017). Most strandings occurred after the breeding season and were primarily sub-adults or adults (Hernandez-Camacho et al. 2017).

In the Hawaii shallow set longline fishery, a total of four confirmed interactions with Guadalupe fur seals have occurred. The first confirmed record of the species interacting with this fishery occurred in 2015 with three additional interactions in 2017. All animals were released alive with hook related injuries. Confirmation of species identification was made by a panel of U. S. West coast pinniped experts after reviewing observer footage of the animals during the interactions. Three additional videos were also reviewed in the same fishery in 2015, however the results were inconclusive and reported at a higher taxonomic listing, as unidentified pinniped. All three interactions occurred outside of the U.S. EEZ, west of the California Current.

4.2.7.4 Summary of the Status of the Guadalupe Fur Seal

In this section of this biological opinion, we explained that the Guadalupe fur seal is threatened, 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 Guadalupe fur seal population dynamics (Figure 51).

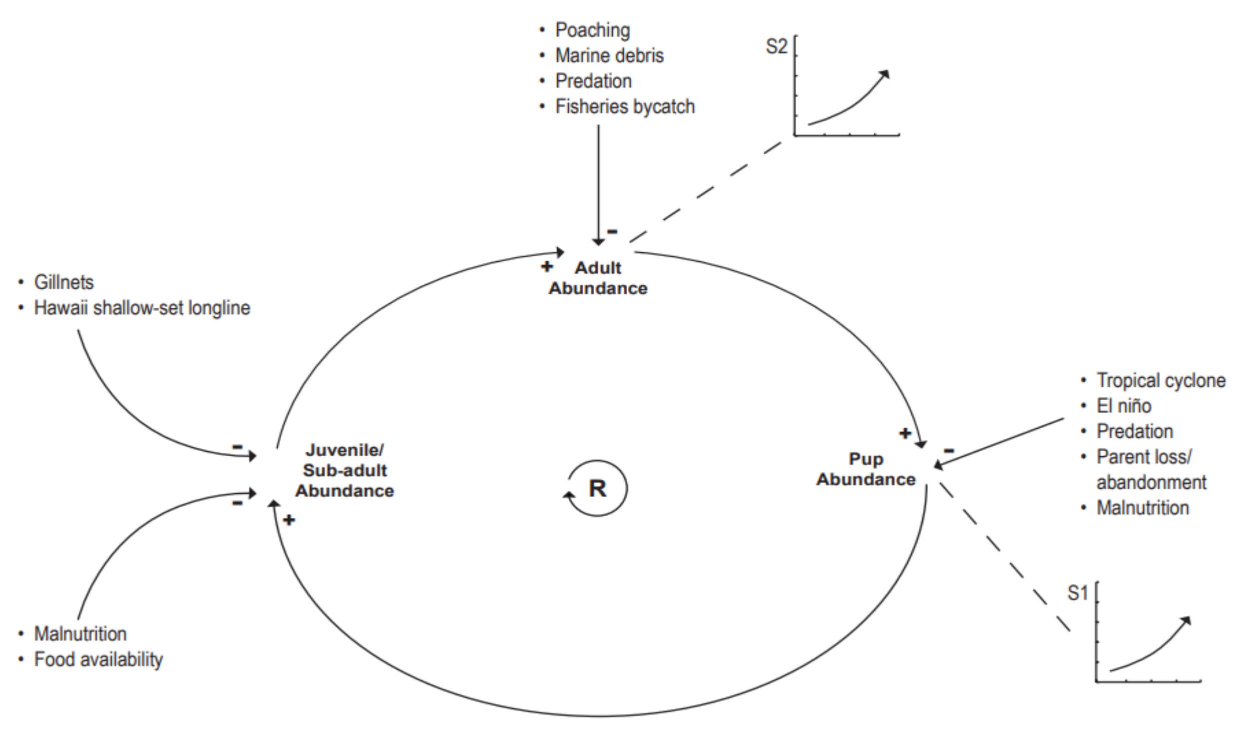


Figure 51. Guadalupe fur seal causal loop diagram depicting various stressor-response relationship for each life cycle stage.

Our graphic illustrates the population behavior of Guadalupe fur seals in S1 and S2, which is based on available rookery count data (Figure 51). All the individuals of the population cannot be counted because all age and sex classes are never ashore at the same time and some individuals that are on land are not visible during the census. However, both pup and adult abundances drive the overall population trend. While this species faces both natural and anthropogenic threats at all life cycle stages, the number of recruits into the population appears to far out way 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 Guadalupe fur seal.

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 exponentially positive.

5 ENVIRONMENTAL BASELINE

By regulation, the environmental baseline for an opinion includes the past and present impacts of all state, federal or private actions and other human activities in the *Action Area*, anticipated impacts of all proposed federal projects in the *Action Area* that have already undergone formal or early section 7 consultation, and the impact of state or private actions which are contemporaneous with the consultation in process (50 CFR 402.02). The 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*” (USFWS 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 12 ESA-listed marine species addressed by this 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 opinion are described below.

Information in this section is summarized from the several past biological opinions on the HI SSLL fishery and other fisheries that occur within the same *Action Area* as the HI SSLL fishery (NMFS 2004, 2005 DSLL opinion, 2012 SSLL opinion, and the 2014 DSLL). We also used the 2017 pelagics report (WPRFMC 2018), the green sea turtle 5-year status review (NMFS and FWS 2007b), the 2009 loggerhead sea turtle status review, the 2011 loggerhead sea turtle listing, the olive ridley sea turtle 5-year status review (NMFS and FWS 2014), 2016 status review report for oceanic white tip sharks (Young et al. 2016), and the other sources as cited in subsequent subsections.

5.1 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) (Wuebbles et al. 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 (Wuebbles et al. 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 (Wuebbles et al. 2017). With significant reductions in emissions, the increase in annual average global temperature could be limited to 3.6 ºF (2 ºC) or less (Wuebbles et al. 2017). 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 (Wuebbles et al. 2017).

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

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

Climate change is projected to have substantial direct and indirect effects on individuals, populations, species, and the community structure and function of marine, coastal, and terrestrial ecosystems in the near future (IPCC 2014; McCarty 2001). 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 (Karl 2009; Mantua et al. 1997).

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 (e.g., for pupping) make many marine mammal populations particularly vulnerable to climate change (Silber et al. 2017). Marine mammals with restricted distributions linked to water temperature may be exposed to range restriction (Issac 2009; Learmonth et al. 2006). 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. He predicted up to a 35% change in core habitat area for some key marine predators in the Pacific Ocean, with some species predicted to experience gains in available core habitat and some predicted to experience losses. Notably, leatherback sea turtles were predicted to gain core habitat area, whereas loggerhead sea turtles and blue whales were predicted to 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 (Birney et al. 2015; 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; Chan et al. 2017; Haigh et al. 2015). 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 (Bermúdez et al. 2016). Blue whales, as predators that specialize in eating krill, are likely to change their distribution in response to changes in the distribution of krill (Clapham et al. 1999; Payne et al. 1986; Payne et al. 1990). Krill have been shown to suffer decreased larval development and survival under lower pH conditions (McLaskey et al. 2016). Krill also have lower metabolic rates after both short-term and long-term exposure to low pH (Cooper et al. 2016). Increased ocean acidification may also have serious impacts on fish 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. 2009a).

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 (Jensen et al. 2018; Newson et al. 2009; Patrício et al. 2017). Within the *Action Area* for this opinion, female biased green sea turtle sex ratios have been reported at foraging locations in San Diego Bay, California (Allen et al. 2017). For the Hawaii green sea turtle population, Chaloupka et al. (2008) reported no gender bias in strandings data from 1982-2003. The 2014 published sea turtle strandings report for Hawaii also indicates little to no apparent bias in green sea turtle sex ratio (50 females, 43 males, 155 unknown/indeterminable) (NMFS 2015b). However, preliminary (unpublished) data from Allen et al. (2017) suggests there may be a female biased sex ratio in this population. 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 (Fuentes et al. 2009a; Mazaris et al. 2008; Reina et al. 2008; Robinson et al. 2008). 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. 2009a). 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 (Azanza-Ricardo et al. 2017; Fuentes et al. 2010; Fuentes et al. 2011; Fuentes et al. 2009b).

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. 2009; 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 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. (2009b) noted that extant marine turtle species have survived past climatic shifts, including glacial periods and warm events, and therefore may have the ability to adapt to ongoing climate change (e.g., by finding new nesting beaches). However, the authors also suggested since the current rate of warming is very rapid, expected changes may outpace sea turtles' ability to adapt.

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

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

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

Environmental changes associated with climate change are occurring within the *Action Area* and are expected to continue into the future. Marine populations that are already at risk due to other threats are particularly vulnerable to the direct and indirect effects of climate change. Several ESA-listed species and habitats considered in this opinion have likely already been impacted by this threat through the pathways described above.

5.2 Fisheries

Past and present fisheries interactions have been, and continue to be, a threat to sea turtles, sharks, and marine mammals within the *Action Area*. Bycatch of ESA-listed species occurs in many fisheries throughout the broad geographic oceanic ranges of these species. Currently, the primary fishing activity in the *Action Area* is longline fishing, except for nearshore fisheries that operate within longline prohibited areas around the Hawaiian Islands. In the past, drift gillnetting also occurred on a large scale within the *Action Area*, but because of high bycatch rates of protected species, a United Nations resolution banned this fishing method, instituting a global prohibition in 1992. Other types of fishing may occur in the *Action Area* outside of longline prohibited areas (e.g., main Hawaiian Islands offshore handline mixed gear).

Longline fishing is conducted by many countries in this region and some of it occurs in the baseline for the *Action Area* but there is also a great deal of fishing that occurs adjacent or further away from the *Action Area* (Figure 52). The *Action Area* is in the management areas of two tuna RFMOs that have different handling and reporting requirements for bycatch. In the Western Pacific, the WCPFC is comprised of 26 nations, with 7 participating territories, and seven cooperating non-member nations. In the Eastern Pacific, the Inter-American Tropical Tuna Commission (IATTC) is comprised of 21 nations and 5 cooperating non-member nations. We include available bycatch information from both RFMOs but we cannot estimate the number that occur in the *Action Area* with any precision and therefore summarize the number of interactions that occur in the North Pacific Ocean.

There are two types of vessels: (1) large distant-water freezer vessels that undertake long voyages (months) and operate over large areas of the region; and (2) smaller offshore vessels with ice or chill capacity that typically undertake trips of about one month (like the Hawaii longline fleet). The total annual number of longline vessels in the western central Pacific region has fluctuated between 3,000 and 6,000 for the last 30 years, this includes the 100-145 vessels (WPRFMC 2018) in the Hawaii longline fisheries (the majority of which are involved in the deep-set fishery). The four main target species are yellowfin tuna, bigeye and albacore tuna, and swordfish. The distribution of longline effort from 2000-2016 is shown in Figure 52. The *Action Area* is shown by the red rectangle, and consists mostly of high seas areas, although some effort continues to occur within the U.S. EEZ.

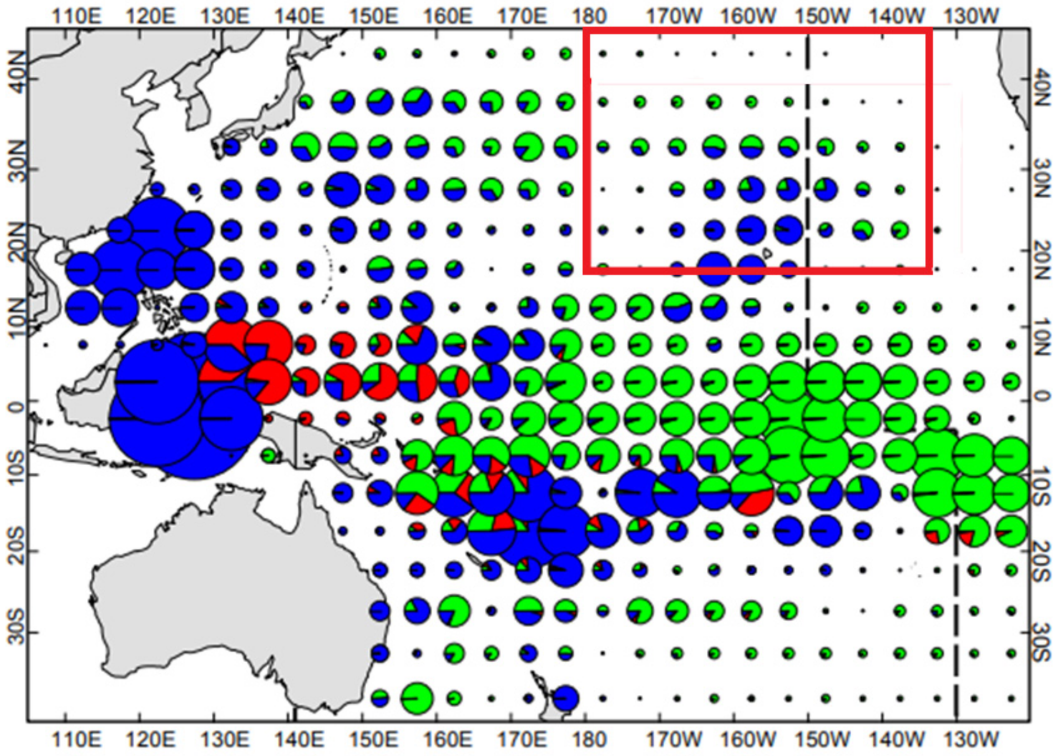


Figure 52. Distribution of longline effort for distant water-fleets (green), foreign-offshore fleets (red) and domestic fleets (blue) for the period of 2000-2016. Source: Williams et al. 2017. The *Action Area* for the HI SSLL fishery where fishing occurs is highlighted by the red rectangle and consists mostly of international waters. The dashed black line going down from 150W represents the boundary between the Western Central Pacific Fisheries Commission (WCPFC) (west of the line) and the Inter-American tropical tuna Commission (IATTC) (east of the line).

While mitigation and minimization measures have reduced fisheries bycatch in the U.S. in recent years, large numbers of ESA-listed species are still routinely captured in federal and state commercial fisheries that target other species. Some ESA-listed species also interact with recreational hook-and-line fisheries.

Fisheries management plans developed for federally regulated fisheries with ESA-listed species bycatch are required to undergo ESA section 7 consultations, including a NMFS issued biological opinion and incidental take statement (ITS). The ITS includes the anticipated amount of take (lethal and nonlethal) and reasonable and prudent measures with specific terms and conditions for mitigating and minimizing the adverse effects of the proposed action on ESA-listed species and designated critical habitat. Some state-managed fisheries with ESA-listed species bycatch have also been the subject of section 7 consultations with NMFS for issuance of ESA section 10(a)(1)(B) incidental take permits (ITPs). ITPs are issued based on NMFS approval of a state's Conservation Plan, which includes ESA-listed species mitigation and minimization measures. Although Hawaii's state fisheries adversely affect listed marine species, Hawaii's state fisheries are not exempt from the prohibitions of take under the ESA; meaning Hawaii state fisheries are not covered by an approved conservation plan, ITP, or subsequent ESA section 7 consultation.

5.2.1 Sea Turtles

5.2.1.1 International Bycatch

Sea turtle bycatch occurs in both large-scale commercial fishing operations as well as small-scale, artisanal fisheries throughout the world. Because of low observer coverage and inconsistent reporting from international fleets, the total number of sea turtle interactions in all Pacific longline fisheries (domestic and international) must be estimated.

Longline fisheries operating in the *Action Area*, such as the Taiwan and China tuna fisheries, have bycatch rates several times higher than the U.S. fisheries (Kaneko and Bartram 2008, Chan and Pan 2012). 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 U.S. reported 27% of those interactions which we will describe in detail in the U.S. fisheries section below.

From 1989-2015, 331 leatherback interactions were reported by 16 countries that operate in the WCPO, therefore we estimate the total leatherback interactions to be approximately 6620 for those 16 countries that participated in the areas beyond national jurisdiction (ABNJ) exercise (Table 33). There were 549 loggerhead sea turtles reported with a total estimate of 10,980 loggerheads caught in the region from 1989-2016 from 16 countries (Table 33). There were 325 green sea turtles reported with an estimate of 6,500 green sea turtles caught in the region from 1989-2016 (Table 33). There were 762 olive ridley sea turtles reported and 556 without the U.S. interactions. When extrapolated from 5% observer coverage the estimate is 15,240 olive ridley sea turtles caught in the region from 1989-2016 (Table 33).

Table 33. Turtle interactions reported for the ABNJ workshop from 16 countries from 1989-2015 and expanded based on 5% coverage in the WCPFC.

	Leatherbacks	**Loggerheads**	**Greens**	**Olive ridley**
Total reported	331	549	325	762
Total estimated	6620	10980	6500	15,240

Table 34 and Table 35 show estimates for sea turtle interactions north of 10N from 2003 to 2017 which includes a portion of the HI SSLL fishery *Action Area* and are adapted from Tables 19 and 32 of Peatman et al. (2018). The range was provided for loggerhead sea turtles and only the median was described for the remaining sea turtle species. It should be noted that this data already includes U.S. 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 10N (Peatman et al. 2018). Lastly, the data have wide confidence intervals for sea turtles (Peatman et al. 2018). Peatman also states that these estimates are likely overestimated after comparing results from the ABNJ report for olive ridleys where their model accounted for estimates of sea turtle distributions and also had additional observer data provided for the workshop. Estimated interactions with loggerhead and leatherback sea turtles in the North Pacific Ocean are listed in Table 36 and Table 37.

Table 34. Low, median, and high (95%) confidence intervals for loggerhead sea turtles in the region North of 10N. Adapted from Tables 19 and 32 in Peatman et al. 2018. (‘000 individuals)

Year	Low	Med	High (95%)
2003	2	34	501
2004	30	141	635
2005	286	753	1976
2006	906	2158	4967
2007	937	2117	5139
2008	231	753	2236
2009	204	553	1592
2010	222	616	1679
2011	288	787	2018
2012	247	666	1807
2013	322	804	1987
2014	765	1785	4168
2015	1220	2636	5899
2016	819	1905	4398
2017	140	555	2178

Year	Low	Med	High (95%)
Totals	6,619	16,263	41,180

Table 35. Median estimates for olive ridley, green, and leatherback sea turtles reported for the region North of 10N. Adapted from Table 32 in Peatman et al. 2018.

Year	Olive Ridley	Green	Leatherback	Turtles nei
2003	4183	848	558	437
2004	1374	1196	697	896
2005	1330	852	884	681
2006	1293	470	617	481
2007	4568	1672	669	366
2008	6840	3750	493	242
2009	6820	3240	617	69
2010	5606	1327	814	28
2011	3570	960	809	45
2012	1922	682	629	76
2013	1535	720	578	134
2014	2976	1092	757	279
2015	4824	1378	891	1076
2016	4767	1481	611	406
2017	3561	2421	299	4
Totals	55,169	22,089	9,923	5,220

Table 36. Summary of estimated interactions of loggerhead sea turtles in the North Pacific Ocean.

Source	Estimate given	Time frame	Annual average range
Beverly and Chapman 2007	520-1,200 juvenile and adults	annually	520-1,200
Lewison et al 2004	2,600-6,000	Year 2000	2,600-6,000
ABNJ 2017	10,980	1989-2016	407
Peatman et al. 2018	6,619-41,180	2003-2017	473-2941

Table 37. Summary of estimated interactions of leatherback sea turtles in the North Pacific Ocean.

Source	Estimate given	Time frame	Annual average range
Beverly and Chapman 2007	200-640 juvenile and adults	annually	200-640
Lewison et al 2004	1000-3,200	Year 2000	1000-3,200
ABNJ 2017	6620	1989-2016	245
Peatman et al. 2018	9,923 median	2003-2017	709

5.2.1.2 U.S. Fisheries Sea Turtle Bycatch

The combined Hawaii longline fisheries (deep-set plus shallow-set) captured about 110 leatherback sea turtles annually before 2001 (McCracken 2000). Since 2004, leatherback turtle interactions in the HI SSLL fishery have been reduced by 83% (Gilman et al. 2007; Swimmer et al. 2017). Between 2004 and 2018, the HI SSLL fishery captured 101 leatherback sea turtles, and an estimated 22 leatherback sea turtles died from their capture based on the post-hooking criteria (Ryder et al. 2006, NMFS 2018X). Since 2012 the anticipated level of take in the HI SSLL fishery has been 26 annually and 54 have been captured since 2012 (Table 38). The Hawaii-deep set fishery is anticipated to interact with up to 72 leatherback turtles (with 27 anticipated mortalities) over a three-year period (NMFS 2014). Between 2005 and 2018, an estimated 146 interactions have occurred in the deep-set fishery, with 85 occurring since 2012 (Table 39) (WPRFMC 2018). Since 2005, the Hawaii longline fisheries combined have reduced their estimated mortality to an average of seven annually (Table 38, NMFS 2018X).

Table 38. Leatherback sea turtle interactions (captures (c) and mortalities (m)) by U.S. fisheries in the North Pacific between 2012 and 2018, and the number of turtles exempt from the prohibitions of take under the most current incidental take statement and ESA section 7 consultation. Parenthetical numbers represent estimated turtle mortalities after the application of the Ryder 2006 post-hooking mortality criteria.

Fishery	**ITS Leatherbacks C(m)**		**Leatherback sea turtle interactions (captures and mortalities (m))**						
			2012 C (m)	**2013 C (m)**	**2014 C (m)**	**2015 C (m)**	**2016 C (m)**	**2017 C (m)**	**2018 C (m)**
SSLL	2 yr	52 (12)	7 (2)	11 (1)	16 (3)	5 (1)	5 (1)	4 (1)	6 (2)
DSLL	3 yr	72 (27)	6 (3)	6 (3)	34 (14)	19 (8)	10 (4)	0 (0)	10 (4)
Ca drift gillnet	5 yr	10 (7)	0 (0)	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)

The HI SSLL fishery captured an estimated 417 loggerhead sea turtles annually (McCracken 2000), with about 40% mortality (Gilman et al. 2007) before it was closed by court order in 2001. Since the fishery re-opened in 2004 loggerhead sea turtle interactions have been reduced by 95% (Swimmer et al. 2017). Between 2004 and 2018, the HI SSLL fishery captured 176 loggerhead sea turtles. Using the 2006 post-hooking criteria by Ryder et al. (2006), an estimated 27 loggerhead sea turtles died out of the 176 loggerhead sea turtles that were captured. Two loggerhead sea turtles came up dead on the line.

Table 39. Loggerhead sea turtle interactions (captures (c) and mortalities (m)) by U.S. fisheries between 2012 and 2018, and the number of turtles exempt from the prohibitions of take under the most current incidental take statement and ESA section 7 consultation. Parenthetical numbers represent estimated turtle mortalities after the application of the Ryder 2006 post-hooking mortality criteria.

Fishery	**ITS Loggerhead C (m)**	**Loggerhead sea turtle interactions (captures and mortalities (m))**						
		2012 C (m)	**2013 C (m)**	**2014 C (m)**	**2015 C (m)**	**2016 C (m)**	**2017 C (m)**	**2018 C (m)**

Fishery	**ITS Loggerhead C (m)**		**Loggerhead sea turtle interactions (captures and mortalities (m))**						
			2012 C (m)	**2013 C (m)**	**2014 C (m)**	**2015 C (m)**	**2016 C (m)**	**2017 C (m)**	**2018 C (m)**
SSLL	2 yr	68 (14)	6 (1)	7 (1)	15 (2)	13 (1)	15 (2)	22 (4)	33 (4)
DSLL	3 yr	18 (13)	0 (0)	10 (7)	0 (0)	10 (7)	10 (7)	15 (9)	0 (0)
Ca drift gillnet	5 yr	10 (7)	0 (0)	0(0)	0(0)	0(0)	0(0)	0(0)	-
CA DSLL	10 yr	1 (1)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	-

Since 2004 the HI SSLL fishery has captured 10 each of olive ridley and green sea turtles, with up to one mortality of each (NMFS 2018X). Since 2005, the deep-set fishery has caught approximately 625 olive ridley sea turtles, and there have been an estimated 593 mortalities (NMFS 2018X). Since 2005, the deep-set fishery has caught approximately 67 green sea turtles, and there have been an estimated 63 mortalities (NMFS 2018X).

The West coast drift gillnet fishery targets swordfish and thresher sharks in the U.S. EEZ and adjacent high seas off the coasts of California, Oregon, and Washington (NMFS 2013). In 2001, NMFS established Pacific Sea Turtle Conservation Areas that prohibit drift gillnet fishing in large portions of the historical fishing grounds, either seasonally or conditionally, to protect endangered leatherback and loggerhead sea turtle populations (66 FR 44549; August 24, 2001). Oregon and Washington state laws currently prohibit landings caught with drift gillnet gear, although ships still fish drift gillnets in federal waters off these states and land their catch in California. The drift gillnet fishery can also be closed during El Niño events in order to reduce bycatch of loggerhead turtles that move further north on the warm El Niño currents from Mexico into U.S. waters (72 FR 31756, June 8, 2007). In 2013, NMFS issued a biological opinion on the continued authorization of the West Coast drift gillnet fishery (NMFS 2013). There have been two loggerhead interactions and two leatherback interactions observed since 2001, with zero interactions since 2012 (NMFS unpublished data).

Table 40. Olive ridley sea turtle interactions (captures (c) and mortalities (m)) by U.S. fisheries between 2012 and 2018, and the number of turtles exempt from the prohibitions of take under the most current incidental take statement and ESA section 7 consultation. Parenthetical numbers represent estimated turtle mortalities after the application of the Ryder 2006 post-hooking mortality criteria.

Fishery	**ITS Olive Ridley**	**Olive ridley sea turtle interactions (captures and mortalities (m))**

	C (m)		2012 C (m)	2013 C (m)	2014 C (m)	2015 C (m)	2016 C (m)	2017 C (m)	2018 C (m)
SSLL	2 yr	4 (2)	0(0)	0 (0)	1 (0)	1 (0)	0 (0)	4 (0)	1 (0)
DSLL	3[a] yr	E. 144(134)	26 (25)	34 (32)	29 (28)	49 (46)	97 (92)	98 (93)	59 (56)
		W. 42 (40)	8 (7)	10 (10)	8 (8)	15 (14)	29 (28)	29 (28)	18 (17)
Ca drift gillnet	5 yr	10 (7)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	
CA DSLL	10 yr	6 (6)	0 (0)	0 (0)	0 (0)	1 (1)	0 (0)	0 (0)	

[a] Corresponds with the 2017 DSLL opinion.

The West Coast longline fishery operates in the North Pacific Ocean, mainly from the U.S. EEZ west to 140 degrees West longitude and from the equator to 35 degrees North (NMFS 2016b). This fishery primarily targets bigeye tuna, although other tuna and non-tuna species are also caught and retained. As of 2016, there was only one boat participating in this fishery. There has been one interaction with an olive ridley sea turtle since 2005 (NMFS unpublished data).

Table 41. Green sea turtle interactions (captures (c) and mortalities (m)) by U.S. fisheries between 2012 and 2018, and the number of turtles exempt from the prohibitions of take under the most current incidental take statement and ESA section 7 consultation. Parenthetical numbers represent estimated turtle mortalities after the application of the Ryder 2006 post-hooking mortality criteria.

Fishery	ITS Green C (m)		Green sea turtle interactions (captures and mortalities (m))						
			2012 C (m)	2013 C (m)	2014 C (m)	2015 C (m)	2016 C (m)	2017 C (m)	2018 C (m)
SSLL	2 yr	6 (2)	0 (0)	0 (0)	1 (0)	0 (0)	0 (0)	2 (0)	1 (0)
Ca drift gillnet	5 yr	10 (7)	0 (0)	0(0)	0(0)	0(0)	0(0)	0(0)	-
CA DSLL	10yr	1 (1)	0 (0)	0(0)	0(0)	0(0)	0(0)	0(0)	-

Table 42. DSLL 3 year ITS from 2017 Supplemental opinion for Green sea turtle DPSs percentage assigned to each DPS in parentheses.

Green DPS	Captures	Mortalities
East Pacific DPS (70)	12	12
Central North Pacific DPS (12)	6	6
East Indian-West Pacific DPS (8)	6	6
Southwest Pacific DPS (7)	6	6
Central West Pacific DPS (1)	3	3
Central South Pacific DPS (1)	3	3

Table 43. Green sea turtle interactions by DPS (captures (c) and mortalities (m)) by U.S. fisheries between 2012 and 2018, and the number of turtles exempt from the prohibitions of take under the most current incidental take statement and ESA section 7 consultation. Parenthetical numbers represent estimated turtle mortalities after the application of the Ryder 2006 post-hooking mortality criteria.

DPS	2012	2013	2014	2015	2016	2017	2018
East Pacific DPS	0 (0)	4 (3)	11 (11)	3 (3)	4 (3)	11 (10)	10 (9)
Central North Pacific DPS	0 (0)	1 (0)	2 (2)	1 (1)	1 (1)	2 (2)	2 (2)
East Indian-West Pacific DPS	0 (0)	0 (0)	1 (1)	0 (0)	0 (0)	1 (1)	1 (1)
Southwest Pacific DPS	0 (0)	0 (0)	1 (1)	0 (0)	0 (0)	1 (1)	1 (1)
Central West Pacific DPS	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Central South Pacific DPS	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)

5.2.2 Oceanic Whitetip Shark

The oceanic whitetip shark is predominantly caught as bycatch and the reporting requirements for bycatch species have changed over time and differ by organization and have therefore affected the reported catch. Oceanic whitetip sharks within the *Action Area* are within multiple RFMO boundaries separated by 150 degrees W longitude north of the equator and 130 degrees W longitude south of the equator.

5.2.2.1 International Fisheries Oceanic Whitetip Shark Bycatch

In the Western Pacific, annual reports provided to the Commission from the member countries, lack species-specific data and do not provide sufficient data to allow assessments of shark stocks (Clarke and Harley 2014; Harley and Piling 2015). Furthermore, some of the world's leading shark fishing nations fail to provide aggregated annual catch data in their annual reports (Clarke and Harley 2014). Young et al. (2016) summarized the status snapshot provided by Clarke (2011), showing reduced trends in catch per unit effort CPUE across the entire Western Pacific. Portions of the *Action Area* are considered within the WCPFC boundaries. To date, only one stock assessment has been completed for the oceanic whitetip shark and only pertains to the Western Pacific. The estimated biomass was 7,295 tones or approximately 200,000 individuals for this entire Region (FAO 2012; Rice and Harley 2012; Young et al. 2016). Additionally, Rice and Harley (2012) reported the oceanic whitetip shark was currently overfished and the stock was in an overfished state with fishing mortality estimated 6.5 times greater than the maximum sustainable yield within this Region. Median shark and ray catch estimates were modeled by Peatman et al. (2018) for multiple regions of the WCPFC using longline observer data. Table 44 shows statistics relevant to the oceanic whitetip shark north of 10N from 2003 to 2017 which includes a portion of the HI SSLL fishery *Action Area* and is an excerpt from Table 30 of Peatman et al. (2018). It should be noted that this data already includes U.S. 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 10N (Peatman et al. 2018). Lastly, the data has a wide confidence interval for key shark species (Peatman et al. 2018).

Table 44. Median oceanic whitetip shark catch estimates (‘000 individuals) north of 10N. Source: Peatman et al. 2018.

Year	Low	Median	High
2003	37.2	51.2	73.1
2004	36.9	51	71.7
2005	37.7	50.7	69.1
2006	28.9	41.5	59.0
2007	33.0	46.8	67.0
2008	32.5	46.2	64.8
2009	32.7	47.6	66.9

Year	Low	Median	High
2010	25.3	38.1	55.4
2011	31.8	46.9	70.7
2012	21.0	29.7	43.8
2013	11.5	16.1	22.9
2014	12.8	18.2	26.4
2015	16.9	23.6	33.3
2016	17.6	25	35.3
2017	11.4	15.9	23.1
Total	387.2	548.5	782.5

The IATTC reports 65 t of sharks were caught within the EPO for 2016, however the data is considered incomplete, and should be considered with caution (IATTC 2018). Additionally, we do not know what portion of the sharks would come from the *Action Area*. Furthermore, the vast majority of fishing effort conducted within the RFMOs range is conducted south of the *Action Area* and not expected to overlap the HI SSLL fishery and was previously discussed in the *Status of Listed Resources*. No stock assessments currently exist for the Eastern Pacific.

5.2.2.2 Other U.S. Fisheries Oceanic Whitetip Shark Bycatch

Table 45 lists the number of interactions of oceanic whitetip sharks that have occurred in the Hawaii deep-set fishery between 2004 and 2017. Interactions with the HI SSLL fishery are addressed in the *Effects of the Action*. The total number of observed sharks was 4,468 individuals with an expanded estimate of 22,002 sharks over this time period. The observer program recorded 38 observed interactions with oceanic whitetip sharks in the first quarter (Q1) of 2018 with 17.9% coverage, and 43 interactions in the second quarter (Q2) with 22.0% coverage (NMFS 2018b, NMFS 2018c).

Table 45. Oceanic whitetip sharks (observed and estimated numbers) captured in the Hawaii deep-set longline fishery between 2004 and 2017. Estimates are calculated using an expansion factor to adjust for less than 100% observer coverage.

Year	Observed	% Observer Coverage	Expansion Factor [1]	Estimated Interactions [2]
2004	730	24.6%	4.07	2968
2005	341	26.1%	3.83	1307
2006	331	21.2%	4.27	1562

Year	Observed	% Observer Coverage	Expansion Factor [1]	Estimated Interactions [2]
2007	262	20.1%	4.98	1304
2008	144	21.7%	4.61	664
2009	244	20.6%	4.85	1185
2010	253	21.1%	4.74	1200
2011	225	20.3%	4.93	1109
2012	172	20.4%	4.90	961
2013	196	20.4%	4.90	1799
2014	374	20.8%	4.81	2578
2015	531	20.6%	4.85	2105
2016	423	20.1%	4.98	2073
2017	242	20.4%	4.90	1187
2018	237	20.4%	4.90	1161

[1] 100/observer coverage. For example, for 2017, 100/20.4 = 4.9.

[2] (Observed interactions) x (Expansion factor). For example, for 2017, 242(4.9) = 1186.3, rounds to 1187.

Source: NMFS Observer Program, unpublished data.

Overall, oceanic whitetip sharks were generally not landed, or are rarely landed in the Pacific Islands Region. Brodziak et al. (2013) concluded that the relative abundance of oceanic whitetip declined within a few years of the expansion of the longline fishery, which suggests these fisheries are contributing to the overutilization of oceanic whitetip within this portion of its range (Young et al. 2016). The majority of oceanic whitetip sharks are now released alive in this fishery, and the number of individual sharks retained by the fishery has declined.

Young et al. (2016) indicated that the oceanic whitetip shark population in the operational range of the fishery might have stabilized in recent years based on a preliminary analysis of annual standardized CPUE from 1995-2014. Since then, observer data from 2015 and 2016 shows nominal CPUE was approximately same or slightly higher than 2014 (NMFS Observer Program, unpubl. data), however these are unstandardized data and should be interpreted with caution.

No interactions have occurred with oceanic whitetip sharks in any federal West Coast fishery (Fahy and Villafana, pers. comm. 2019). There were approximately 167 oceanic whitetip shark commercially landed in the State of Hawaii from 1999 to 2015 according to commercial fishing reports provided by Hawaii DAR (2019). The oceanic whitetip was not differentiated to species prior to 1999. Additionally, three years had insufficient data to report landings for the species; 2009, 2014, and 2016 (Hawaii DAR 2019). These are likely the minimum number of oceanic whitetip sharks taken due to the inconsistency and underreporting in State fisheries.

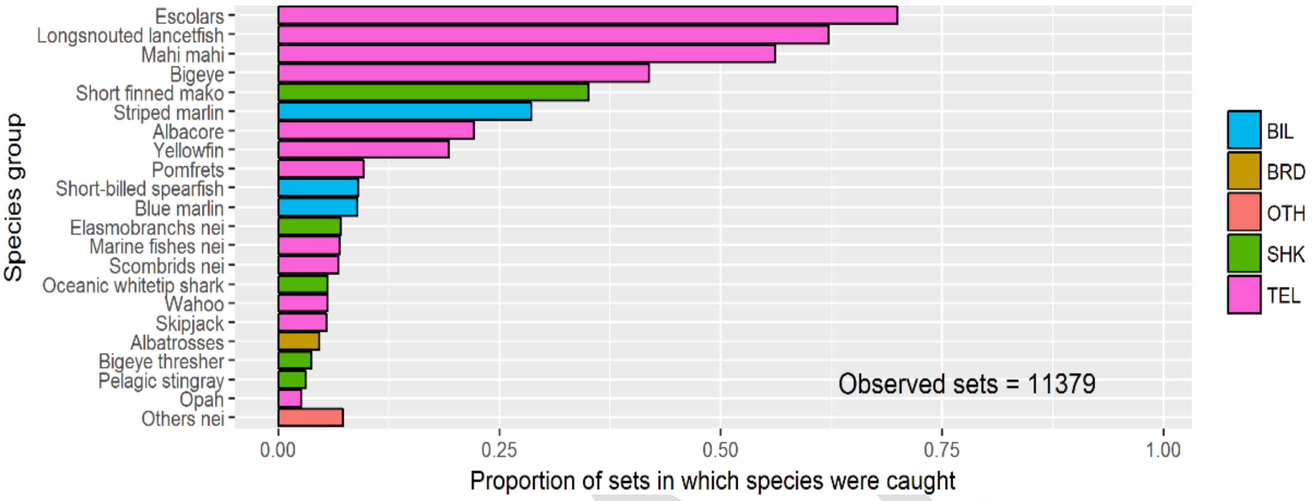
5.2.3 Giant Manta Ray

5.2.3.1 International Fisheries Giant Manta Ray Bycatch

As mentioned previously, the *Action Area* overlaps with two RFMO's, the WCPFC and the IATTC. Cooperating commission members of the WCPFC have been providing estimates of the catch of non-target species to the commission since 2005. However, inconsistencies are common between the reports form cooperating countries and within reports from the same country. Therefore, the Secretariat of the Pacific Community's longline observer data is the best data available. A report from Peatman et al. (2018), summarizes observer data from 2003 to 2017 to estimate the catch and catch composition of the longline fisheries of the western and central Pacific Ocean at the regional level. These estimates and associated tables represent all fisheries within the WCPFC (except for a few excluded fisheries, i.e. former shark fisheries in the EEZ's of Papua New Guinea and Solomon Islands) therefore, they are included in the status section. Peatman et al. (2018) also separate bycatch catch data and estimates at the regional level.

The catch and estimate data for North temperate fisheries (areas North of 10N) are applicable to the baseline discussion, however, it does include areas outside of the *Action Area* and it also includes the Hawaii based longline fishery data discussed in detail below. At this time, it is not possible to isolate the *Action Area* nor exclude the Hawaii fishery data from the Secretariat of the Pacific Community's longline observer data and related estimates. An additional constraint to note, is that Peatman et al. (2018) combined mantas and mobulas into a larger category of elasmobranchs not included elsewhere. As evident in Figure 53 below, the proportion of elasmobranchs nei is slightly less than 10% of the catch for shallow sets and approximately 20% of the catch for deep sets.

North temperate fisheries – shallow sets



North temperate fisheries – deep sets

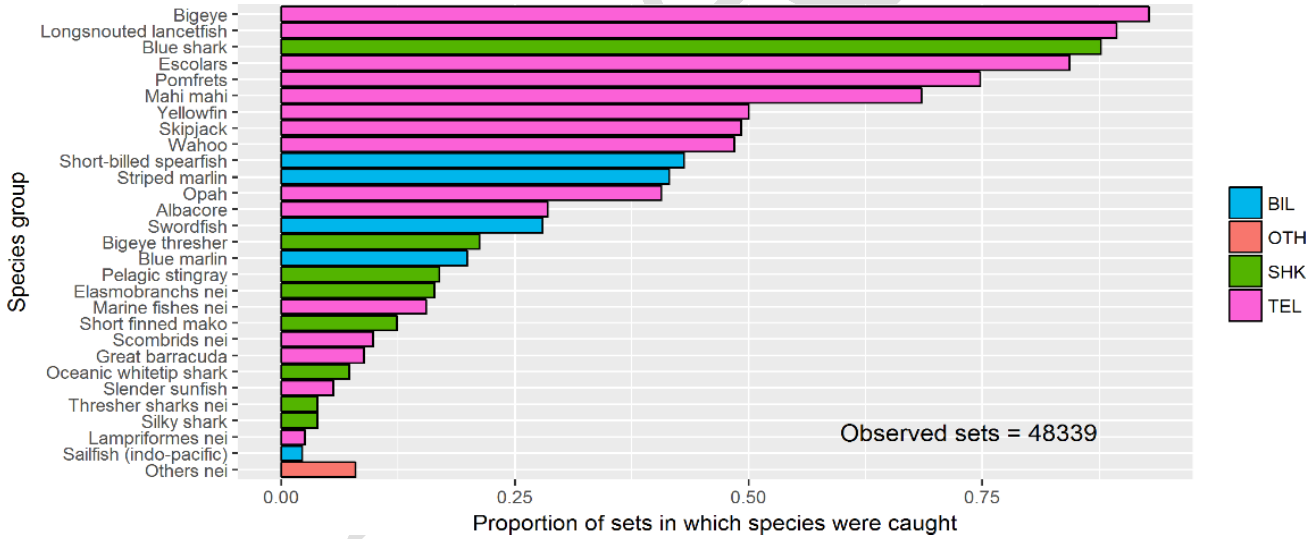


Figure 53. The proportion of longline shallow sets and deep sets in north temperate fisheries with observed catch against species/species group. Rarely observed species have been grouped in to ‘others nei’. Bar color denotes billfish (BIL), marine mammals (MAM), others nei (OTH), shark species (SHK), teleosts or fish (TEL), turtles (TTX) and tuna (TUN). Source: Peatman et al 2018.

Peatman et al. (2018) also provided annual catch estimates at the regional levels. These estimates are based on the limitations of the dataset and the methodology used to obtain these estimates (low percentage of observer coverage and the exclusion of a few fisheries, i.e. former shark fisheries in the EEZ's of Papua New Guinea and Solomon Islands). The median estimated number of interactions with the elasmobranch nei using summarized observer data from 2003 to 2017 at the regional level totals 826 animals. Given that the elasmobranch nei category includes other species than the giant manta ray, we don't expect there were 826 interactions with giant manta rays, however, we can use this number to assess the maximum number of interactions that could have occurred.

Table 46. Median shark and ray catch estimates (number of individuals) by species/species group and region. This region if North of 10N. Source: Peatman et al. 2018.

Year	**Elasmobranchs nei**
2003	43.3
2004	41.3
2005	41.5
2006	43.0
2007	45.6
2008	52.6
2009	66.3
2010	63.9
2011	70.5
2012	47.9
2013	25.9
2014	35.0
2015	57.9
2016	91.4
2017	99.5
Total	825.6

IATTC

Co-operating non-parties, co-operating fishing entities or regional economic integration organizations within the IATTC (collectively "CPCs") with longline vessels over 20 m overall length ("large-scale tuna longline fishery") are required to report annually to the IATTC catch and effort data. However, formats used to report the catch and effort data vary considerably amongst the CPC and through time (Griffiths and Duffy, 2017). In addition, there are inconsistencies in how the species are reported. Some CPC's summarize catch while others aggregate catch by broad taxonomic groups and according to Griffiths and Duffy (2017), the reporting of bycatch by most CPCs has generally been poor, even for species of recent and serious conservation concern such as sharks.

The IATTC database includes catch data on manta rays in the purse-seine fishery, however, it does not have any data regarding longline interactions with manta rays. The purse seine fishery is concentrated far south of the *Action Area*, therefore, it is described in the status section but not included in the environmental baseline.

5.2.3.2 U.S. Fisheries Giant Manta Ray Bycatch

Longline fishing affects giant manta rays primarily by hooking, but also by entanglement and trailing of gear that remains attached to an animal. Table 47 describes all giant manta ray interactions in the DSLL fishery from 2004 to 2018.

Table 47. Observed and estimated annual takes using expansions for giant manta rays in the DSLL, 2004-2018.

Year	**Observed**	**% Observer Coverage**	**Expansion Factor 1**	**Estimated Interactions 2**
2004	1	24.6%	4.07	5
2005	2	26.1%	3.83	8
2006	2	21.2%	4.27	10
2007	2	20.1%	4.98	10
2008	2	21.7%	4.61	10
2009	4	20.6%	4.85	20
2010	17	21.1%	4.74	81
2011	1	20.3%	4.93	5
2012	2	20.4%	4.90	10
2013	1	20.4%	4.90	5

2014	3	20.8%	4.81	15
2015	2	20.6%	4.85	10
2016	4	20.1%	4.98	20
2017	0	20.4%	4.90	0
2018	1	22.0%	4.54	5

[1] 100/observer coverage. For example, for 2017, 100/20.4 = 4.9.

[2] (Observed interactions) x (Expansion factor). For example, for 2016, 4(4.98) = 19.9, rounds to 20.

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

We used the ratio of confirmed giant manta rays and mobula (devil ray) to estimate the potential giant manta rays included in the *Mobulidae* spp. and unidentified ray category. We incorporated the Wilson score method to include confidence intervals. Details regarding the proration using the Wilson score method are discussed in section 6.2 of this biological opinion, *Exposure Analyses*. As a result, we estimate the HI DSLL fishery had 56 interactions with giant manta rays (44 observed plus 12 *Mobulidae* spp. and 4 unidentified rays) between 2004-2018. Using the Wilson score method, we estimate the HI SSLL fishery had 21 interactions with giant manta rays (17 observed plus 4 (unidentified) giant manta rays) between 2004-2018.

Prior to 2004, NMFS' observers were instructed to document interactions with rays at the family level only, *Mobulidae* spp. Based on comments provided by some observers, rays identified as giant manta rays and mobulas were grouped accordingly. Therefore, there were 3 confirmed giant manta rays, 4 confirmed mobula rays, 41 *Mobulidae* spp. and 49 unidentified rays. Using the Wilson score method, we estimate that prior to 2004, the combined Hawaii longline fisheries (deep-set plus shallow-set) captured 42 giant manta rays (3 observed plus 39 *Mobulidae* spp. and unidentified rays). In total, since 1994, we estimate that the Hawaii based fishery has captured 119 giant manta rays.

In U.S. west coast fisheries, the giant manta ray is occasionally observed as bycatch in the California drift gillnet fishery targeting swordfish and threshers, but in low numbers and only during El Niño events. In fact, from 1990 – 2006, only 14 giant manta rays were observed caught, with 36% released alive. Estimated (extrapolated) catch for the entire period was 90 individuals (95% CI: 26 – 182; CV = 0.48) (Larese and Coan 2008). Since 2010, no manta rays have been observed caught in the California drift gillnet fishery (data available from: NOAA Fisheries West Coast Drift Gillnet Fishery Catch Summaries).

5.2.4 Guadalupe Fur Seal

Within the *Action Area*, no other fisheries are known to interact with Guadalupe fur seals at this time. While overlap occurs between the HI SSLL and the HI DSLL fisheries, fishing effort in the DSLL fishery does not extend to the east of 140 W latitude where interactions in the HI SSLL are occurring. No Guadalupe fur seals have been observed entangled in California gillnet fisheries between 1990 and 2014 (Carretta et al. 2004, 2016b, 2017 citing Julian and Beeson 1998), although stranded animals have been found entangled in gillnet of unknown origin. Gillnets have been documented to entangle marine mammals off Baja California (Carretta et al. 2017 citing Sosa-Nishizaki et al. 1993), but no recent bycatch data from Mexico are available. No interactions have been documented with Guadalupe fur seals in any federal West Coast fisheries (Fahy and Villafana, pers. comm. 2019).

5.3 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.

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

Ship strikes were identified as a source of mortality for green sea turtles in Hawaii waters Chaloupka et al. (2008) reported that 2.5% of green turtles found dead on Hawaiian beaches between 1982 and 2003 had been killed by boat strikes. Ship strikes have also been reported as a potentially important threat to sea turtle populations by researches in other parts of the world including the Canary Islands (Oros et al. 2005), Italy (Casale et al. 2010), and the Galápagos Islands (Parra et al. 2011; Denkinger et al. 2013). NMFS estimates that 250 green sea turtles are struck by vessels annually around Hawaii and the mortality for vessel strikes is 95-100% (NMFS 2018). There is potential for the Hawaii longline fisheries to hit green sea turtles on their way to and from the fishing grounds due to the high density of green sea turtles near the main Hawaiian Islands. As noted earlier in the status of green sea turtles, we expect that the Central North Pacific DPS is at greatest risk of collision with vessels. Based on turtle stranding data and the number of vessel involved in both fisheries we estimate that there could be up to one green sea turtle killed annually from a vessel strike from these two fisheries and we expect that potential to continue into the future and could possibly increase due to the growing number of green sea turtles around Hawaii. NMFS estimates that Navy training activities in the Hawaii Range kills 20 green sea turtles from the Central North Pacific DPS per year (or 100 over the 5-year ITS) and harms approximately 7 turtles annually (34 over the 5-year ITS; see Table 48) due to vessel strikes which will continue to occur into the future (NMFS 2018). The Mariana Islands training and testing activities intersects with the baseline in the transit corridor of their action and they estimate that one green sea turtle (unknown DPS) is killed on an annual basis (NMFS 2017).

Table 48. The number of turtles exempt from the prohibitions of take under the most current incidental take statement and ESA section 7 consultation for vessel strikes that may occur during Navy military training operations in the *Action Area*.

Species	**Vessel Strike**	
	Hawaii Range Complex (5 years)	
	Mortality	**Harm (non-lethal injuries)**
Green Sea Turtle- Central North Pacific DPS	100	34
Green Sea Turtle- Eastern Pacific DPS	3	1
Olive Ridley Sea Turtle	2	1

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

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

5.4 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.

5.4.1 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; NOAA 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; NOAA 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).

Aurioles-Gamboa (2015) identified oil spills as a potential threat to the Guadalupe fur seal, due to the requirements to maintain a thick pelage for thermoregulation purposes. We do not have numbers to quantify this threat to the species in the *Action Area*.

5.4.2 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 and can fish unattended for decades (ghost fishing), killing large numbers of marine animals through entanglement. Fisheries observers in the Hawaii longline fisheries document marine debris that is encountered during fishing trips. From 2008-2015 they documented 1,426 debris encounters. Over 75% of the debris encountered consisted of nets and tangled debris. Almost 85% of debris was collected and brought back to the docks; this amounted to over 211,838 pounds of debris (NOAA 2015).

Marine debris is a significant concern for sea turtles, 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). It is thought that some sea turtles eat plastic because it closely resembles jellyfish, a common natural prey item (Schuyler 2014). Ingestion of plastic debris can block the digestive tract which can cause turtle mortality as well as sub-lethal effects including dietary dilution, reduced fitness, and absorption of toxic compounds (Laist et al. 1999; Lutcavage et al. 1997).

Santos et al. (2015) found that a small amount of plastic debris was sufficient to block the digestive tract and cause death. They reported that 10.7% of green turtles in Brazilian waters were killed by plastic ingestion, while 39.4% had ingested enough plastic to have killed them. These results suggest that debris ingestion is a potentially important source of turtle mortality, one that may be masked by other causes of death. Gulko and Eckert (2003) estimated that between one-third and one-half of all sea turtles ingest plastic at some point in their lives. A more recent study by Schuyler et al. (2015) estimates that 52% of sea turtles globally have ingested plastic debris. Schuyler et al. (2015) synthesized the factors influencing debris ingestion by turtles into a global risk model, taking into account the area where turtles are likely to live, their life history stage, the distribution of debris, the time scale, and the distance from stranding location. They found that oceanic life stage turtles are at the highest risk of debris ingestion. Based on this model, olive ridley turtles are the most at-risk species; green, loggerhead, and leatherback turtles were also found to be at a high and increasing risk from plastic ingestion (Schuyler 2014). The regions of highest risk to global turtle populations are off the east coasts of the U.S., Australia, and South Africa; the East Indian Ocean, and Southeast Asia. In addition to ingestion risks, sea turtles can also become entangled in marine debris such as fishing nets, monofilament line, and fish-aggregating devices or FADs (NRC 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 forty years (1970-2010). Ingestion of marine debris can also have fatal consequences for large whales. In 2008, two male sperm whales stranded along the northern California coast with large amounts of fishing net scraps, rope, and other plastic debris in their stomachs. One animal had a ruptured stomach, the other was emaciated, and gastric impaction was suspected as the cause of both deaths (Jacobsen et al. 2010). According to marine debris data compiled in NOAA's 2014 Marine Debris Program Report, three Guadalupe fur seals were reported to have evidence of entanglement; one with net markings, another with a polyfilament line around its neck, and one with hook and line (Hanni et al. 1997). Once entangled, Guadalupe fur seals may drag and swim with gear attached for long distances, ultimately resulting in fatigue, compromised feeding ability, or severe injury, which may lead to reduced reproductive success and death.

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

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

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

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*.

5.4.3 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 2003). 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 activities; active sonar; and pile driving and construction.

5.4.3.1 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 (Hildebrand 2009c; Mckenna et al. 2012a; NRC 2003b). 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.

Pinniped reactions to vessels are variable and reports include a wide spectrum of possibilities from avoidance and alert, to cases where animals in the water are attracted, and cases on land where there is lack of significant reaction suggesting habituation to or tolerance of vessels (Richardson et al. 1995c). There is also evidence that nearby vessels and construction noise affect haulout times during pupping season for some pinniped species (Jansen et al. 2010, Karpovich et al. 2015, Anderwald et al. 2013b). Guadalupe fur seals are found offshore in the *Action Area*, away from their haul out and pupping areas, where vessel noise may have other effects that could potentially disturb feeding and breeding but these affects are more difficult to study and therefore unquantifiable at this time.

Exposure to vessel noise for sea turtles could result in short-term behavioral or physiological responses (e.g., avoidance, stress) but is not quantifiable in the *Action Area* at this time.

Exposure to vessel noise for elasmobranchs could result in short-term behavioral or physiological responses (e.g., avoidance, stress) but is not quantifiable in the *Action Area* at this time.

5.4.4 Seismic Surveys

Offshore seismic surveys involve the use of high energy sound sources operated in the water column to probe below the seafloor. Seismic surveys primarily for scientific research have been conducted in the *Action Area* over the past several decades (NMFS 2018b).

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. 2000b). McCauley et al. (2000b) 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.

In 2018 NMFS performed an ESA section 7 consultation on seismic surveys funded by the National Science Foundation in 2018 and 2019 and they estimated that the action would expose sea turtles to sounds from the airgun arrays during the course of the seismic surveys that will elicit a behavioral response that will constitute harassment. A behavioral response that would constitute harassment is expected to occur at received levels at or above 175 dB re: 1 μPa (rms) for ESA-listed sea turtles. No death or injury is expected for any individual sea turtle exposed to seismic survey activities. NMFS expects that 7 North Central Pacific DPS of green, 65 leatherback, 61 North Pacific Ocean DPS loggerhead, and 32 olive ridley sea turtles will be harassed during the survey around Hawaii (NMFS 2018a).

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 2009). 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 (Casper et al. 2012; Casper et al. 2003; Casper and Mann 2006; Casper and Mann 2009; Ladich and Fay 2013; Myrberg 2001). 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 take. If these behavioral reactions were to occur, we would not expect them to result in fitness impacts such as reduced foraging or reproduction ability. There is no data to quantify what potential impacts there are to elasmobranchs in the *Action Area*.

At this time, no seismic surveys have been conducted within the *Action Area* where Guadalupe fur seals have been documented (near the Californian coast). Furthermore, no ITSs have been issued for the species by NMFS.

5.4.5 Military Training and Testing Activities

The Navy has conducted training and testing activities and other military readiness activities in the *Action Area* and these activities are ongoing and are expected to continue into the future (NMFS 2018). During training, existing and established weapon systems and tactics are used in realistic situations to simulate and prepare for combat. Activities include routine gunnery, missile, surface fire support, amphibious assault and landing, bombing, sinking, torpedo, tracking, and mine exercises. Testing activities are conducted for different purposes and include at-sea research, development, evaluation, and experimentation. The majority of the training and testing activities the Navy conducts in the *Action Area* are similar, if not identical, to activities that have been occurring in the same locations for decades (NMFS 2018).

Navy activities produce sound and visual disturbances to marine mammals and sea turtles throughout the *Action Area*. Impacts from harassment due to Navy 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. Sound produced during Navy training and testing activities also results in instances of TTS and PTS to marine mammals and sea turtles. The Navy training and testing activities constitute a federal action and take of ESA-listed marine mammals and sea turtles considered for these Navy activities have previously undergone ESA section 7 consultations (NMFS 2018). They have an incidental take statement for harassment of Guadalupe fur seas of 15/ 1,442 (TTS/ Behavioral) over a five-year period. They also have an incidental take statement for sea turtles for harassment and harm from noise (Table 49). The oceanic whitetip and the giant manta ray are not expected to be affected from these activities, thus they were not provided an ITS.

Table 49. The number of sea turtles and marine mammals exempt from the prohibitions of take under the most current incidental take statement and ESA section 7 consultation for Navy training activities in the Hawaii Range Complex on an annual basis (NMFS 2018b).

Species	**Impulsive and Non-Impulsive Acoustic Stressors**		
	Harassment (TTS/ Behavioral)	**Harm (PTS)**	**Harm (Slight Lung Injury)**
Guadalupe Fur Seal	15 / 1,442	0	none
Green Sea Turtle- Central North Pacific DPS	20 / 1,831	7	1
Olive Ridley Sea Turtle	0 / 96	none	none
Loggerhead Sea Turtle- North Pacific DPS	0 / 182	none	none
Leatherback Sea Turtle	0 / 193	none	none

5.5 Synthesis of Baseline Impacts

Collectively, the stressors described above have had, and likely continue to have, lasting impacts on the ESA-listed species considered in this consultation. Some of these stressors result 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. Assessing the aggregate impacts of these stressors on species is difficult and, to our knowledge, no such analysis exists. This becomes even more difficult considering that many of the species in this opinion are wide-ranging and subject to stressors in locations throughout the *Action Area* and outside the *Action Area*.

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 entanglements in fishing gear, and overharvests. 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.

The stress regime created by the activities discussed in this *Environmental Baseline* continue to be a threat for loggerhead sea turtles, although it may be less than it once was since there has been increased nesting at some of the key nesting beaches over the last decade. Even 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.

The stress regime created by activities discussed in this *Environmental Baseline* continue to be a threat for the Central North Pacific and East Pacific DPSs of green sea turtles although they may be less than they once were since there has been increased nesting over the last couple of decades. Threats associated with climate change related to loss of nesting habitat pose the greatest risk to the Central North Pacific DPS, along with vessel strikes and nearshore fisheries in the main Hawaiian Islands. Even with increasing populations these DPSs have a risk of extinction due to environmental stochasticity and the threats from fishing gear and vessel strikes contribute to the risk. The other four green sea turtle DPSs could also be impacted by 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*.

As previously discussed in the *Status of Listed Resources*, Guadalupe fur seals are experiencing increased population abundance and are thus expanding their range to compete for resources, which suggests that the stress regime they are experiencing in the baseline is not a threat to recovery of the species. But future climate change impacts may delay recovery due to a larger population with less prey resources. The population has also experienced greater pup mortality from El Nino events and tropical cyclones which may be exacerbated in the future (Fleischer 1987; Trillmich et al. 1991; Gallo-Reynoso 1994; Elorriage-Verplancken et al. 2016a, 2016b; Pablo-Rodriguez 2016; NMFS 2017; Juarez-Ruiz et al. 2018; Aurioles-Gamboa 2015).

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 take in the industrial purse-seine fishery and target take 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.

The stressors discussed in this *Environmental Baseline* are also a threat for the oceanic whitetip shark. Oceanic whitetip sharks face a high probability of becoming extinct as a result of both environmental and demographic stochasticity. Due to their life-history characteristics, oceanic whitetip sharks are more susceptible to the effects of high fishing exploitation. The information available suggests that oceanic whitetip sharks have high a probability of becoming extirpated in the Pacific Ocean unless they are protected from the combined threats of incidental take 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 the species.

6 EFFECTS OF THE ACTION

In *Effects of the Action* sections of biological opinions, NMFS presents the results of its assessment of the probable direct and indirect effects of federal actions that are the subject of a consultation as well as the direct and indirect effects of interrelated, and interdependent actions on threatened and endangered species and designated critical habitat. 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 this biological opinion 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 4.1 of this biological opinion, *Status of Listed Resources Not Considered Further*.

6.1 Potential Stressors

Potential stressors associated with the proposed action include interaction with fishing vessels during transit and active fishing, which may result in vessel collisions, vessel noise, emissions, addition of waste and oily discharges into marine waters, loss of derelict fishing gear (nets, lines, and hooks that have been lost, abandoned or discarded into marine waters, and the capture (hooking, entanglement and mortality) of listed species, their prey or predators, and depredation of hooks by listed species. This list of stressors is not exhaustive; however, it represents the stressors for which some information is available. Stressors on this list are not mutually exclusive because some sources of stressors may produce multiple stressors. For example, vessels represent one stressor because of their weight and speed (risk of potential collisions), a second form of stressor because of the sounds associated with their passage (bow wave and engine noise), a third form of stressor when they release wastes and pollutants, and a fourth form of stressor when they are actively fishing and fishing gear is deployed and retrieved. Fishing gear, lines and hooks, represent one stressor because of their ability to entangle and hook non-target animals, like protected species, as well as a source of derelict fishing gear.

Vessel noise, vessel collisions, introduction of discharges and other wastes, gear loss and vessel emissions were covered earlier in this biological opinion (See section 4.1, Listed Resources Not

Considered Further). As a result, in this section we focus primarily on the stressors created by active fishing, which results in hooking and entanglement of listed species and are thus likely to adversely affect them. We briefly discuss slipped or unobserved catch, and depredation of bait and catch. As outlined in the introductory paragraph of this section, we follow our exposure and response evaluation with an *Integration and Synthesis* (see chapter 7 of this biological opinion), which presents the results of our risk analyses.

6.2 Exposure Analyses

As discussed in the *Approach to the Assessment* section of this opinion, our exposure analyses are designed to determine whether listed resources are likely to co-occur with any direct and indirect beneficial and adverse effects that these actions have on the environment and the nature of that co-occurrence. 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.

Based on the limited empirical information available, we cannot estimate the number of endangered or threatened marine animals that might be exposed to the HI SSLL fishery and the stressors produced by the fishery. The area encompassed by the fishery is so large that it would be almost impossible to identify or estimate the specific number of different marine animals that are actually exposed to the fishery and all of its stressors, or changes in the pattern of exposure over the course of a year. As a result, our effects analysis focuses on observed hooking and entanglement as the best surrogates for evaluating exposure, although this approach would underestimate true exposure to the total stressor regime from this action because the data we need to support comprehensive exposure analyses were not available (e.g., data to evaluate exposure to vessel traffic or data to evaluate unobserved or dropped catch, injury and mortality).

6.2.1 Unobserved Catch

Various factors determine how interactions with target and bycatch species occur. This discussion focuses on those aspects which provide a level of uncertainty to catch data, and unobserved interactions for various species under consideration. As previously discussed, the HI SSLL fishery is required to use size 18/0 circle hooks with a 10 degree offset, and mackerel-type bait. These regulations were implemented to reduce bycatch of non-target species, particularly sea turtles. The use of circle hooks has been extensively studied to reduce bycatch interactions (as reviewed in Parga 2012). Bait selection also influences longline fisheries operations, catch rates, the species that interact with the gear, and bait loss (as reviewed in Kumar et al. 2016). However, due to the technique in which these fisheries operate, observation and monitoring of all hooks set is not possible. As stated in the biological evaluation, the HI SSLL fishery overall catch rates in relation to the number of hooks set are generally low (less than 2%). This leads to some level of uncertainty when catch is lost prior to being boarded or observed.

A number of terms describe how catch can evade capture whether intentional or not. Gilman et al. (2013) summarizes a number of these terms including pre-catch loss, cryptic mortality, slipped catch, collateral mortalities, and post-release mortalities; in the context of longline fishing operations. The literature was extensively reviewed, including all of these terms, in an attempt to determine if any data existed on catch loss rates before boarding occurs. Additional searches included keywords: "unaccounted mortality," "unobservable mortality," "drop off rates," "longline drop-offs," "longline discards," and "catch depredation." Cryptic mortality and post release mortality provided the most relevant results for this discussion. Overall, the literature suggests that quantification of these events is extremely difficult if not impossible. However methods to understand these mechanisms are evolving. For instance, the use of hook timers, electronic (video) monitoring, and side-scan sonars can potentially provide some insight to what is occurring with the gear while it's soaking.

Many studies attempt to determine post release mortality metrics for various species through tagging studies, including sea turtles and sharks (Chaloupka et al. 2004; Hueter 2006; Ryder et al. 2006; Skomal 2007; Campana et al. 2009; Gilman et al. 2013; Gallagher et al. 2014; Swimmer et al. 2014). Most of the literature for sharks are for non ESA-listed species and focus on biological characteristics related to stress response and reflex impairment either via the buildup of lactate or plasma catecholamine levels (Hight et al. 2007; Frick et al. 2010; Gallagher et al. 2014). Whereas Dapp et al. (2016) provided an extensive literature review comparing different commercial fishing gears and respiratory functions in sharks. Davis (2010) discusses reflex impairment for various fish species other than sharks. However, determining post release mortality efforts are currently underway to illuminate metrics relevant specific to the oceanic whitetip shark, and specifically within the Hawaii longline fisheries (Hutchinson, M. pers. comm. 2018).

Gilman et al. (2013) also discusses how models and estimates attempt to account for these potential interactions. Warden and Murray (2011) and Murray (2011) discuss how bycatch estimation may underrepresent actual levels of interactions with protected species in various fisheries. Discussions specific to underrepresentation for the Hawaii based commercial fisheries have been debated in the literature (Moyes et al. 2006; Campana et al. 2009; Musyl et al. 2009).

All species under consideration which could, or are known to depredate bait or catch, have the potential to be affected from hooking and could therefore potentially escape before observation. Those species include loggerhead sea turtles, leatherback sea turtles, green sea turtle sea turtles, hawksbill sea turtles, olive ridley sea turtles, Guadalupe fur seals, Hawaiian monk seals, MHI insular false killer whales, sperm whales, scalloped hammerhead sharks, and the oceanic whitetip sharks. When data was not available for the species under consideration, we searched for information on surrogate species.

We were not able to find literature or observer data specific to any of the listed species considered herein to determine lost catch or rates of interactions with animals that were not boarded or witnessed by observers. Although fishing line damage may be observed when the gear is retrieved, what species is responsible for the damaged the line is generally unknown. Wire leaders and traces are not used in this fishery, therefore sharks of any species can bite through the leader and escape. The MHI insular false killer whale and the sperm whale could potentially break the line due to their size and strength. Lines can also break from other factors like marine debris, other vessels, failure of gear, poor line integrity, etc. Additionally, drop offs can occur after mortality of the individual. While catch rates are determined, disposition of all hooks and bait at retrieval are not.

Even so, this fishery has 100% observer coverage, meaning all observable interactions are noted and documented in the observer's protected species log. Additionally, pre-catch fishing mortality, otherwise known as depredated catch, is tracked when observed. Our anticipated take levels for the HI SSLL fishery accounts for observerable catch and bycatch as defined in the metrics for model computation; however due to limitations in the modeling process, there is no practical way to quantify unobserved interactions and are thus considered a constraint to the analysis (McCracken 2018).

6.2.2 Depredation

Once an animal is hooked on the line, their ability to move is restricted which increases their chances at being depredated on by a predatory species. Additionally, if the animal drowns, the carcass could attract predators. Sharks and cetaceans are well known to depredate catch (Garrison 2007; Gilman et al. 2008, 2013; Mandelman et al. 2008; as reviewed in Hamer et al. 2012, 2015; Kumar et al. 2016; as reviewed in Mitchell et al. 2018) and specifically in the Hawaii longline fisheries (Baird and Gorgone 2005; Gilman et al. 2006; Forney et al. 2011; Baird et al. 2015; Welch 2017).

While portions of damaged catch can be quantified due to 100% observer coverage in the HI SSLL fishery, some interactions may be unobserved. Catch that is fully eaten, or removed from the hook, cannot be quantified and would be considered pre-catch loss. Often times, fish heads or partially eaten catch are retrieved. Examination of the damage can provide insight to the type of predatory species which predated on the bait or catch (Rabearisoa et al. 2012). However, as discussed in the literature, quantification of specific predatory species responsible for bait or catch damage can be difficult (Mandelman et al. 2008). Shark teeth may potentially dislodge in the retrieved animal, thereby suggesting a species, however multiple shark species could depredate on a large fish like a tuna or marlin. Observable interactions could be possible at the vessel rail prior to landing for species specific information. Additionally, observers are required to record damaged catch when landed. Forney et al. (2011) determined that 3% of longline sets in the HI SSLL fishery were depredated on by false killer whales from 2003 through 2009. Even so, there is a potential for some protected species, like turtles to be caught on longline gear and depredated by predatory species without ever being detected by observers during gear retrieval.

The following search terms, or combination of search terms, were assessed during the literature review in order to determine depredation rates, if any, on these species: “depredation,” “depredation of hooked [species],” “depredation of [species],” “shark depredation,” “killer whale depredation,” “white shark depredation,” ”depredation by cetaceans,” “catch de/predation,” “cookie cutter shark,” and “longline depredation.” Mitchell et al. (2018) provided the most comprehensive review of shark depredation in fisheries worldwide by reviewing 61 studies from 1955 to 2018. Results of the literature review provided a large wealth of information particularly regarding bycatch interactions and rates, bycatch mitigation and reduction measures, and literature focused on extraneous species - such as dolphins or other toothed whales. These articles were reviewed in case any pertinent information, in terms of surrogacy, to ESA-listed species under consideration; would be applicable.

After reviewing the observer database on the HI SSLL fishery from 2004 to 2018, we found one record of potential depredation of a hooked protected species. Observers noted a hooked oceanic whitetip shark was depredated upon by a cookiecutter shark (*Isistius brasiliensis*). These sharks bite small circular or oval shaped plugs of flesh typically from larger species. Wounds are typically categorized as minor and heal over time. The literature suggests many fish species, including sharks, are depredated by this ectoparasitic species in Hawaiian waters (Papastamatiou et al. 2010). Dwyer and Visser (2011) provided an extensive overview of the literature in regards to cookie cutter shark depredation on cetaceans worldwide. A total of forty-nine species have been documented thus far with a maximum wound healing time of 150 days in killer whales (*Orcinus orca*) (Dwyer and Visser 2011).

Based on Dwyer and Visser (2011) and others (Gallo-Reynoso and Figueroa-Carranza 1992; Hiruki et al. 1992; Papastamatiou et al. 2010), species under consideration in this consultation that would be most likely to experience cookie cutter depredation include large whales and seals. Furthermore, whales have been documented specifically in Hawaiian waters with cookiecutter wounds (Baird et al. 2006; Aschettino et al. 2012). However, as noted the observer database contained one record of cookie cutter depredation on a protected species, the oceanic whitetip shark. No notes were provided on the extent, magnitude, or disposition of the injury to determine if the wound was healed tissue or a fresh injury. Additionally, photographic evidence is not available for this record. Thus it is unclear if this depredation event occurred after interaction with the longline gear.

6.2.3 Hooking and Entanglement of Listed Species

Interactions in the form of observed hooking and entanglement are the best surrogates for evaluating exposure of listed species, and represent the most significant risk to listed species from the proposed action. The numbers of interactions discussed herein, however, represent the minimum number of the listed species described that are exposed to hooking and entanglement because we cannot account for unobserved interactions that may have occurred before the animal was brought to the surface for the observer to witness. We describe the exposures and release conditions of protected species in detail in the following sections.

6.2.3.1 Sea Turtles

Since 2004 the HI SSLL fishery has interacted with (injured and killed) a total of 302 sea turtles (Table 50). Green and olive ridley sea turtles, each, have comprised 3% of the turtle interactions, whereas leatherback sea turtles have comprised 35% of the turtle interactions, and loggerhead sea turtles account for more than half (59%) of the total observed sea turtle interactions since 2004.

Table 50. Total number and portion of sea turtles by species injured or killed in the Hawaii shallow-set longline fishery since 2004.

Observed turtle status	Green sea turtle	Leatherback sea turtle	Loggerhead sea turtle	Olive Ridley sea turtle	Grand Total
Dead			2		2
Injured	10	105	175	10	293
Grand Total	10	105	177	10	302
% of Total	3	35	59	3	100

NMFS uses an established method for estimating post release mortality of sea turtles in longline gear (Ryder et al. 2006; Summer and Gilman 2011). Post release mortality rates are calculated based on the release condition of the turtle and five injury categories. Release conditions are: (a) released with all gear removed, (b) released with hook and line less than half the length of the carapace length (turtle is not entangled), (c) released with hook and line greater than or equal to half the length of the carapace (turtle is not entangled), and (d) released with hook and entangled (line is not trailing). The six injury categories are:

I. Hooked externally with or without entanglement ("external");

II. Hooked in upper or lower jaw with or without entanglement ("jaw");

III. Hooked in cervical esophagus, glottis, jaw joint, soft palate, tongue, and/or other jaw/mouth tissue parts not categorized elsewhere with or without entanglement. Includes all events where the insertion point of the hook is visible when viewed through the mouth ("insertion visible");

IV. Hooked in esophagus at or below level of the heart with or without entanglement. Includes all events where the insertion point of the hook is not visible when viewed through the mouth "insertion not visible");

V. Entangled only, no hook involved ("entangled"), and

VI. Comatose/resuscitated ("comotase").

We used NMFS' observer data collected in the HI SSLL fishery to assign species to injury categories and release conditions. A number of records in the observer database required hand searches of observer notes to properly code release condition and injury category. A subset of these records required data corrections to account for missing or incomplete data, such as amount of gear remaining relative to turtle length. We also made reasoned decisions to assign appropriate injury categories to a small subset of turtles. For instance, we used the probability of a particular injury to assign injury categories to four loggerhead sea turtles that lacked information. When observers were unable to identify a turtle to species, we used spatial distribution patterns and probabilities to assign the unidentified turtle to one of the four listed species expected in this fishery.

Table 51 describes the number of sea turtles captured alive in the HI SSLL fishery by injury category, and Table 52 presents the proportion of all injuries and mortalities for all sea turtle species. Most turtles are externally hooked (62%), and entanglement occurs relatively infrequently for all species with more loggerhead sea turtles described as entangled than other species. No turtles have been recorded as comatose (injury category IV from Ryder et al. 2006). Only two loggerhead sea turtles have been recorded as dead during the past 14 years, resulting in a probability of 0.01 of an at-vessel death for loggerhead sea turtles. In contrast, the at vessel mortality rate for all sea turtles is 0.007. We discuss exposure scenarios by species in more detail in subsequent sections.

Table 51. Numbers of all sea turtles, by species and injury category, captured alive in the Hawaii shallow-set longline fishery, 2004-2018.

Injury Category	**Green sea turtle**	**Leatherback sea turtle**	**Loggerhead sea turtle**	**Olive Ridley sea turtle**	**Grand Total**
Entangled	2	3	9	1	15
External	4	89	87	5	185
Insertion Not Visible		1	18		19
Insertion Visible	2	8	31	1	42
Jaw	2	4	30	3	39
Total Injured	10	105	175	10	300

Table 52. Proportion of sea turtles, by species and mortality and injury categories in the Hawaii shallow-set longline fishery, 2004-2018.

Injury Category		**Green sea turtle**	**Leatherback sea turtle**	**Loggerhead sea turtle**	**Olive Ridley sea turtle**	**Grand Total**
Dead				0.01		0.01
Injured	Entangled	0.01	0.01	0.03	0.00	0.05
	External	0.01	0.30	0.29	0.02	0.62
	Insertion Not Visible	-	0.00	0.06	-	0.06
	Insertion Visible	0.01	0.03	0.10	0.00	0.14
	Jaw	0.01	0.01	0.10	0.01	0.13
Grand Total		0.03	0.35	0.59	0.03	1.00

Leatherback sea turtle

Leatherback sea turtles are vulnerable to foul hooking in the flipper and shoulder area, possibly due to their morphology (large size, long pectoral flippers, and lack of a hard shell), their attraction to gelatinous organisms and algae that collects on buoys and buoy lines at or near the surface, or some combination of these and/or other reasons. Since the HI SSLL fishery re-opened in 2004 there have been a total of 105 observed interactions with leatherback sea turtles that have resulted in their hooking or entanglement.

Table 53 describes the number of leatherback sea turtles captured in the HI SSLL fishery between 2004 and 2018 by their injury category and release condition, and Table 54 describes the proportion in each category. Overall, external hooking accounts for the largest category of interactions (85%). Thirty-seven percent of interactions result in the animal being externally hooked and released without gear. Whereas, 48% of the animals externally hooked are released with gear (22% hooks and trailing greater than or equal to ½ the turtle's SCL and 26% are released with gear and trailing line that is less than half of the turtle's SCL). There were several cases where the observer could not determine where the animal was hooked because the line parted before the animal was close enough for the observer to see. In these instances we used the proportions calculated for the known interactions with leatherbacks to assign injury categories.

Table 53. Number of leatherback sea turtles captured in the Hawaii shallow-set longline fishery between 2004 and 2018 by their injury and release condition.

	Release Condition			
Injury Category	**Line < 1/2 SCL**	**Line ≥ 1/2 SCL**	**No Gear**	**Grand Total**
Entangled	-		3	3
External	27	23	39	89
Insertion Not Visible		1		1
Insertion Visible	4	3	1	8
Jaw	2	1	1	4
Grand Total	33	28	44	105

Table 54. Proportion of leatherback sea turtles captured in the Hawaii shallow-set longline fishery between 2005 and 2018 by their injury and release condition.

	Release Condition			
Injury Category	**Line < 1/2 SCL**	**Line ≥ 1/2 SCL**	**No Gear**	**Grand Total**
Entangled	-	-	0.03	0.03
External	0.26	0.22	0.37	0.85
Insertion Not Visible	-	0.01	-	0.01
Insertion Visible	0.04	0.03	0.01	0.08
Jaw	0.02	0.01	0.01	0.04
Grand Total	0.31	0.27	0.42	1.00

Demographic Patterns of Exposure

Leatherback sea turtles encountered in the HI SSLL fishery are generally too large for observers to bring aboard; for turtles that cannot be boarded observers estimate the SCL of the turtle whenever possible. When length estimates were provided in standard feet, we converted those estimates to metric measurements. Most estimates recorded by observers fall between 4 and 6 feet; however, these estimates must be regarded with some caution because of the lack of precise measurements available for leatherback sea turtles captured by this fishery. Nevertheless, best estimates suggest that the average size of a leatherback sea turtle captured in the HI SSLL fishery is about 145 centimeters (observer data; Martin et al. 2018).

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 NMFS' observers suggest that the majority of leatherback sea turtles that interact with this fishery with are likely adults or nearly adults (subadult turtles). In contrast, the American Samoa and HI DSLL fisheries interact with leatherback sea turtles of an average size less than 120 cm SCL, suggesting that they are interacting with older juvenile and subadult leatherback sea turtles.

Based on genetic samples from leatherback sea turtles in the HI SSLL, 98.3% are confirmed to be from the Western Pacific regional population comprised of Papua-Barat, Indonesia, Papua New Guinea and Solomon Islands. One haplotype (Dc8.1), found in one animal occurs in low frequency in both Western Pacific and Eastern Pacific nesting stocks. Stock origin will be resolved in this case with ongoing nDNA data analysis (Dutton pers comm. December 12, 2017).

The sex of leatherback sea turtles that this fishery interacts with is unknown because the animals typically cannot be boarded, nor identified by sex during an interaction. However, between 1998 and 2005 the HI DSLL and American Samoa fisheries were responsible for the death of ten smaller leatherback sea turtles (Martin et al. 2018). These turtles were returned to NMFS where they were necropsied. Seven of the leatherback sea turtles were from the HI DSLL fishery and three from the American Samoa longline 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 (Martin 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). In previous consultation, NMFS has assumed that about 65% of the leatherback sea turtles in the HI SSLL fishery are females. This estimate appears reasonable based on the DPS and limited studies of foraging and nesting studies in this DPS (Benson et al. 2011, Tiwari et al. 2013).

Spatial and Temporal Patterns of Exposure

The greatest overlap between the occurrence of leatherback and North Pacific loggerhead sea turtles and the HI SSLL fishery occurs in the zones east of 140°W (Zone 1), a band of sea surface temperature (SST) between 17 – 18.5°C (Zone 2), and a band of SST between 22.4 – 23.4°C (Zone 3) (Howell et al. 2015). Martin et al. (2018) developed a random forest model, which is a collection of decision trees based on an ensemble of learning algorithms, to predict the oceanic variables that most influence the spatial and temporal behavior of leatherback sea turtles. The model included over a dozen environmental attributes including data from Howell et al. 2015. The random forest model (Martin et al. 2018) predicted two areas where leatherback sea turtles had their highest probability of occurrence, northwest of the Hawaiian Islands and just west of California. The model found the ten most important variables influencing leatherback sea turtle occurrence (listed in decreasing order): 1) top of the thermocline depth, 2) isothermal layer depth, 3) distance to current front, 4) current speed, 5) mixed layer depth, 6) temperature at the mixing layer, 7) SST, 8) distance to shore, 9) north current speed, and 10) current divergence.

Martin et al. (2018) then developed risk contour maps of the areas where leatherback sea turtles are at highest risk of interactions with the HI SSLL fishery (Figure 54). See Table 55 for a summary of leatherback sea turtle interactions, interactions and sets by critical area, and total sets in the HI SSLL fishery by quarter from Martin et al. (2018). The highest risk of interactions occurs in the 10% quantile during quarters 2, 4, 1 and then followed by 3.Quarter 2 accounts for 32% of all interactions between 2005 and 2017. Quarter 4 accounts for 30% of all interactions, and quarter 1 accounts for 27%, whereas quarter 3 accounts for 11% of the total interactions. Interestingly, while quarter 4 accounts for the second highest amount of leatherback sea turtle interactions (30%), the same quarter represents only 14% of the total sets in the fishery. In contrast, 32% of the leatherback sea turtle interactions occur during quarter 2 as does 32% of the sets, and 27% of the interactions with leatherback sea turtles occurs during quarter 1 when 50% of the sets occur. The 10% quantile in quarter 3 is located east of 140º W.

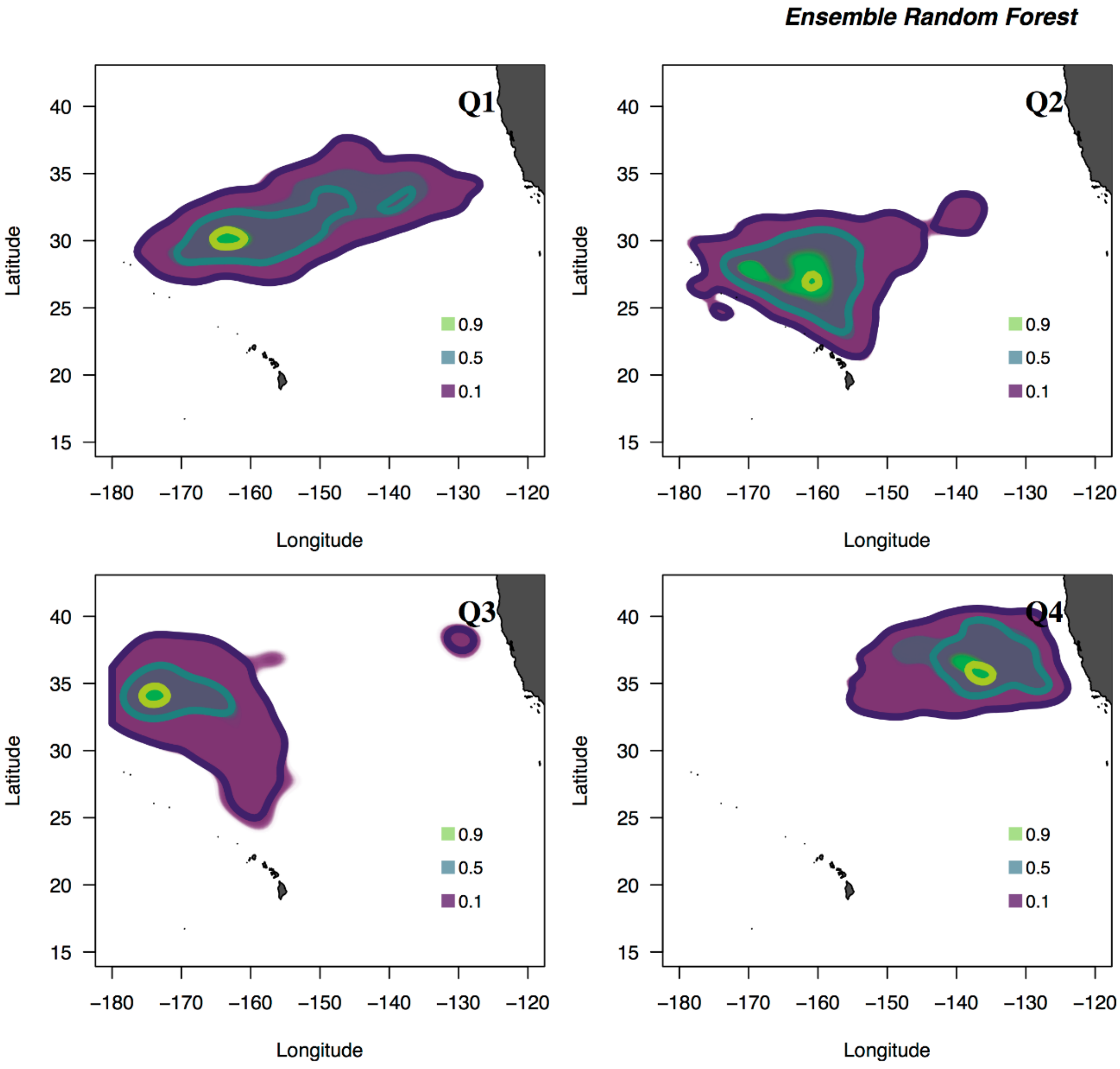


Figure 54. Areas of highest interaction between leatherback sea turtles and the Hawaiian shallow-set longline fishery by quarter. Solid polygons indicate each of the 100 Random Forest critical area predictions from the Ensemble Random Forest model by Martin et al. (2018). The

line indicates the minimal area of overlap between all submodels of the Ensemble Random Forest Model. Figures from Martin et al. (2018).

Table 55. Leatherback sea turtle interactions, interactions and sets by critical area, and total sets in the Hawaii shallow-set longline fishery by quarter (Martin et al. 2018; pers. comm Jones to Garrett, 2019)

Calendar year quarter	**Number of leatherback sea turtle interactions (% of total interactions)**	**Number of interactions in critical area quantile (% of quantile total)**			**Number of Sets (% of total sets)**	**Number of sets in critical area quantile (% of quantile total)**		
		10%	**50%**	**90%**		**10%**	**50%**	**90%**
Q1 (Jan – Mar)	25 (27)	25 (100)	13 (52)	2 (8)	8,581 (50)	8,503 (99)	4,521 (54)	241 (3)
Q2 (Apr – Jun)	30 (32)	28 (97)	24 (80)	1 (3)	5,537 (32)	5,381 (97)	4,177 (76)	117 (2)
Q3 (Jul-Sep)	10 (11)	9 (90)	6 (60)	1 (10)	712 (4)	629 (89)	304 (45)	46 (7)
Q4 (Oct-Dec)	28 (30)	28 (100)	21 (75)	3 (11)	2,302 (13)	2,280 (99)	1,390 (60)	123 (5)

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interaction levels (aka "anticipated take levels"; McCracken 2018) using Bayesian statistical inference techniques as described in the *Approach to the Assessment* section of this biological opinion, and in greater detail in Appendix A (McCracken 2018). The predictions were based on the recorded bycatch of leatherback sea turtles in the HI SSLL fishery. Table 56 contains the resulting predictions of future 1, 2, and 3 year interactions (mean and 95 percentile) for leatherback sea turtles in the HI SSLL fishery (from McCracken 2018).

Table 56. Mean and 95 percentile of leatherback sea turtle posterior estimated exposure in terms of anticipated interactions with fishing gear (i.e., anticipated take level; from McCracken 2018).

Period	**Mean**	**95 percentile**
Annual	10	21
2-year	20	35
3-year	30	48

Loggerhead Sea Turtle

Loggerhead sea turtles represent greater than half of the turtle interactions in the HI SSLL fishery (59%, see Table 50). Loggerhead sea turtles comprise 58% of all injured sea turtles in this fishery, and the only turtle species with observed at-vessel mortalities in the past 14 years. Most loggerhead sea turtles are injured; however, in 2017 and 2018 two loggerhead sea turtles were retrieved dead. Table 57 lists the number of loggerhead sea turtles by injury and release condition in the HI SSLL fishery between 2004 and 2018. Table 58 describes the proportion of live loggerhead sea turtles by injury category and their release condition. Of the loggerhead sea turtles retrieved alive, more are hooked externally than any other hook location. That is, 50% of all injured loggerhead sea turtles that are recorded alive upon haul back are hooked externally, and 49% are released with no gear. "No gear" releases comprise 85% of all loggerhead sea turtles released in the past 14 years in this fishery. The removal of gear, even after hooking and entanglement, increases the probability of sea turtle survival relative to turtles that are released with gear. Only 5% of loggerhead sea turtles are released with trailing line greater than or equal to half the length of the carapace, and 11% are released with trailing line less than half the carapace length. In total, 16% of all loggerhead sea turtles have been released alive with trailing gear.

Table 57. Number of loggerhead sea turtles captured in the Hawaii shallow-set longline fishery between 2004 and 2018 by their injury and release condition.

Injury Category		**Release Condition**			
		Line < 1/2 SCL	**Line ≥ 1/2 SCL**	**No Gear**	**Grand Total**
Dead	External			2	2
Injured	Entangled		1	8	9
	External	1		86	87
	Insertion Not Visible	12	4	2	18
	Insertion Visible	5	3	23	31
	Jaw	1		29	30
Total		19	8	150	177

Since at vessel mortalities are unusual for this fishery, we examined observer data and notes closely to see if anything out of the ordinary may have occurred with the two loggerhead sea turtles recorded dead. We compared the sizes of the animals to other loggerhead sea turtles taken in the fishery and found that the smaller of the two turtles was only slightly smaller than other turtles that were captured alive. The soak time of the gear and the haulback times were average compared to other trips that captured turtles. One of the turtles was captured towards the end of the gear, which is where 30% of turtles are caught and one was caught about mid gear. Observers did not report anything out of the ordinary as far as line parts which can add to the haulback time, and could in turn increase the risk of a turtle drowning. We also investigated to see if it was possible that the turtles had been previously caught since there were higher than usual interactions at this time in the HI SSLL fleet and some observers ran out of tags due to higher than normal interactions per vessel. There were no other turtles captured and released in the area around that time that had the same SCL and all turtles were tagged before release so it does not appear that the HI SSLL fleet had captured either of these two turtles during the same year. On both of these trips the vessels caught several other turtles that were released alive. The smaller turtle was retained for necropsy.

Table 58. Proportion of live loggerhead sea turtles that exhibited different hooking injuries in the Hawaii shallow-set longline fishery between 2004 and 2018.

Injury Category	**Release Condition**			
	Line < 1/2 SCL	**Line ≥ 1/2 SCL**	**No Gear**	**Grand Total**
Entangled	-	0.01	0.05	0.05
External	0.01	-	0.49	0.50
Insertion Not Visible	0.07	0.02	0.01	0.10
Insertion Visible	0.03	0.02	0.13	0.18
Jaw	0.01	-	0.17	0.17
Total	0.11	0.05	0.85	1.00

In 2018 and 2017, interactions with loggerhead sea turtles were the highest recorded in the HI SSLL fishery in the past 14 years. In 2018, 33 loggerhead sea turtles interactions occurred, and in 2017, 21 loggerhead sea turtle interactions occurred (see Figure 55). In contrast, in 2008 the HI SSLL fishery had zero interactions with loggerhead sea turtles, whereas in 2018, the fishery was only open from January 1 and closed on May 4, 2018 and remained closed through December 31, 2018 (50 FR 21939), yet last year marked the highest number of loggerhead sea turtle interactions in the past 14 years. In fact, 2008 and 2018 represent the minimum and maximum number of interactions with loggerhead sea turtles in any calendar year since 2004 in the HI SSLL fishery.

The mean (rounded) and median number of loggerhead sea turtle interactions in the HI SSLL fishery from 2004 through 2018 is 12 turtles per year. The average number of interactions from 2004 to 2016 was 9 loggerhead sea turtles per year. Across the 14 years, largely due to the recent activity, the 3-year moving average depicts an upward trend in interactions with loggerhead sea turtles in this fishery (Figure 55).

Although 2017 and 2018 were particularly high years of interactions, a preliminary analysis of the spatial distribution of those interactions did not reveal apparent changes in fishing effort during these two years (PIFSC as cited in WPRFMC 2018). However, as reviewed by WPRFMC, a small number of vessels are implicated as having five or more interactions with loggerhead turtle (Table 59).

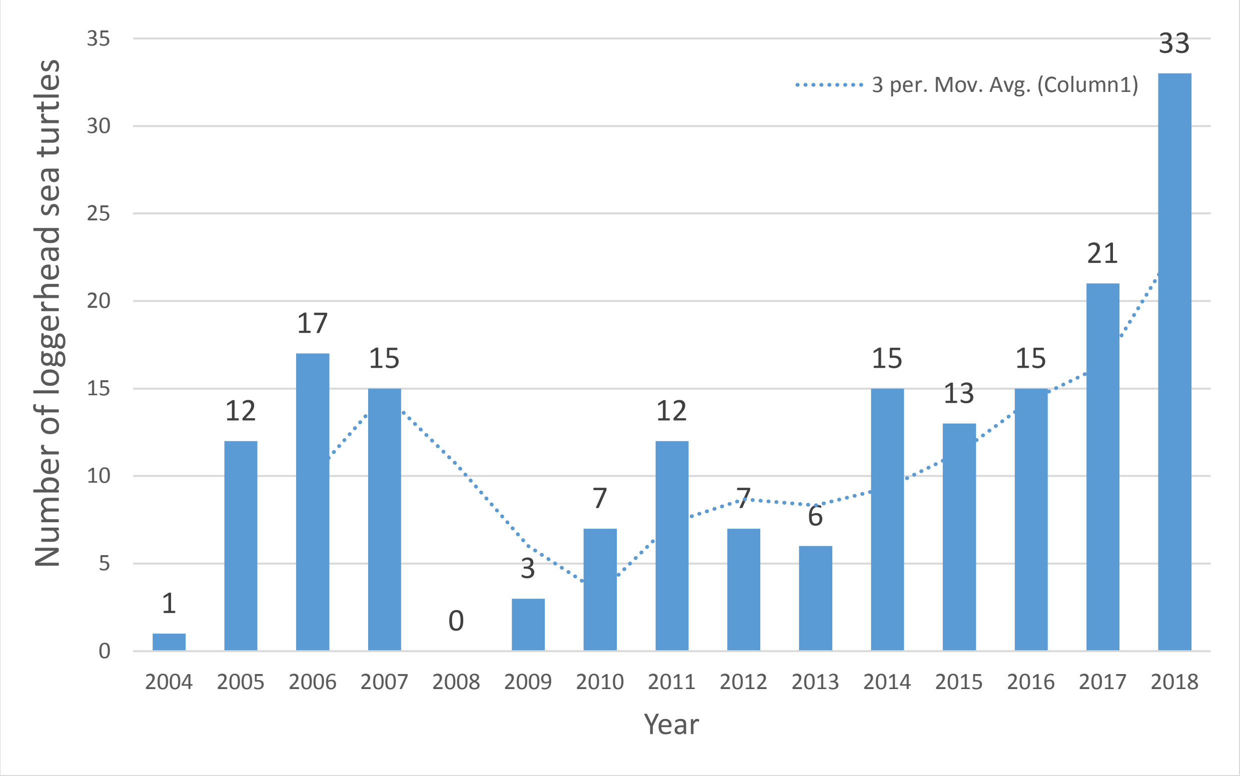


Figure 55. Loggerhead sea turtle interactions by year in the Hawaii shallow-set longline fishery between 2004 through 2018 with 3-year moving average.

Table 59. Number of loggerhead sea turtle interactions per vessel per year, 2004- 2018. Permit numbers were used as a proxy for individual vessels. From WPRFMC 2018.

Loggerhead Sea Turtles		
Number of turtles per vessel per year	**Number of vessels**	**% of vessel years with at least one interaction**
1	66	68
2	16	17
3	8	8
4	3	3
≥5	4	4

Our analysis reveals that during the winter of 2017/18, there were nine vessels responsible for 44 interactions with loggerhead and leatherback sea turtles (Table 60). Four of the nine vessels had four or more sea turtle interactions and in total were responsible for 37 interactions with loggerhead and leatherback sea turtles in the three month period (82% of the interactions). While

NMFS tracks take of sea turtles on a calendar year, the high interaction rate between November 2017 and January 2018 is cause for concern and suggests that a few vessels can significantly adversely affect the listed sea turtles, as well as disproportionately affect the total incidental catch of turtles in the fleet.

Table 60. Vessel and loggerhead and leatherback sea turtle interactions between November 2017 and January 2018.

	Number of Vessels	Number of Loggerhead sea turtle interactions	Number of Leatherback sea turtle interactions	Grand Total Number Sea Turtle Interactions (%)
Vessels with Interactions	9	39	6	45 (100)
Vessels≥4 interactions	4	34	-	37 (82)

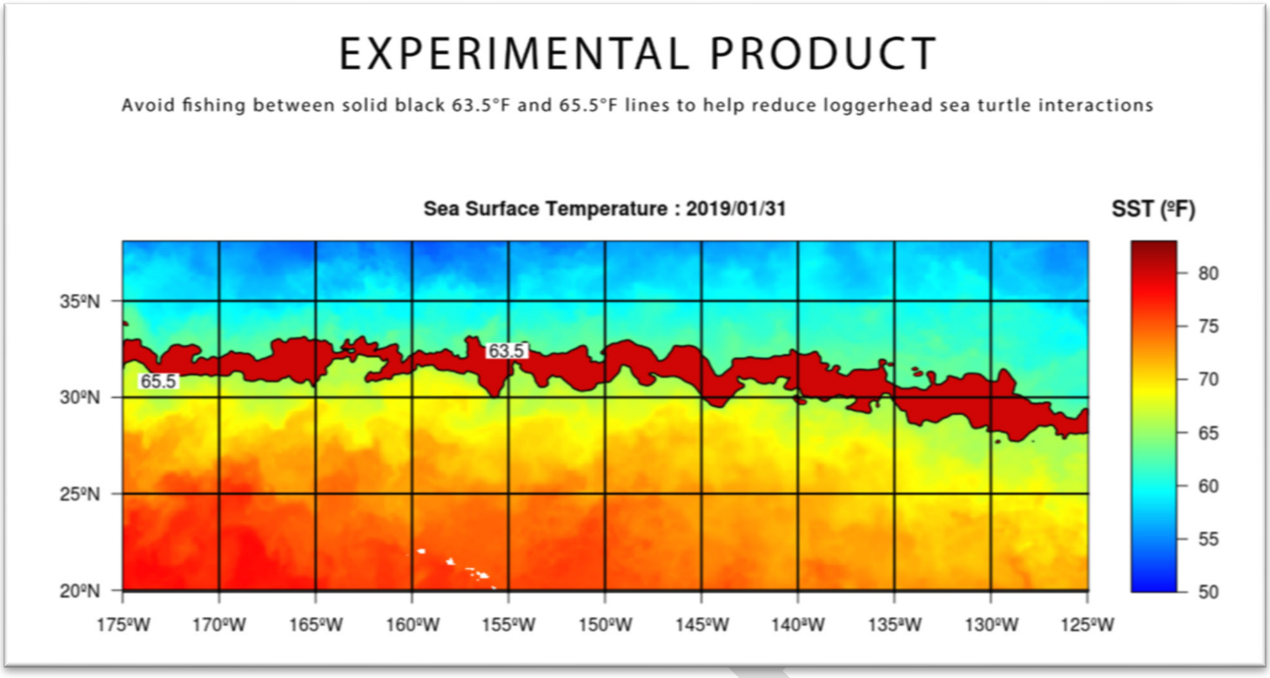
Demographic Patterns of Exposure

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 from the North Pacific DPS. Based on the loggerheads that have been caught thus far, using the relative population size as weighted priors the estimate for each subpopulation are: mean of 50% (CI 0-97%) for the Mainland, 40% (CI 0-99%) for Yakushima, and 9% (CI 0-52%) for Ryuku (Table 15) (Dutton pers comm. May 14, 2018).

Based on NMFS observer data, the average size of loggerhead sea turtle that the HI SSLL fishery interacts with is 60 cm SCL, suggesting that the average loggerhead sea turtle is in the juvenile age class. Because only adult sea turtle show evident external sexual dimorphism, sex ratios are difficult to obtain for the juvenile age class. The sex ratio for North Pacific loggerheads at all ages is unknown and a 50:50 ratio has been used for assessing the population (Conant et al. 2009, Casale and Matsuzawa 2015), therefore we will use the 50:50 ratio in this opinion.

Spatial and Temporal Patterns of Exposure

One of the most important oceanic features that affect loggerhead sea turtle habitat use is sea surface temperatures (Howell et al. 2008, 2015). NMFS issues a composite image of remotely-sensed SST data (the average of the most recent 3-day period) and ocean current vectors (see www.pifsc.noaa.gov/eod/turtlewatch.php) to illustrate the preferred thermal habitat of loggerhead sea turtles and the area where more than 50% of loggerhead sea turtle interactions have occurred during the first quarter of the year (Figure 56). The map is meant to help fishermen reduce interaction with loggerhead sea turtles and is based, in part, on research that indicates that most loggerhead turtles stay in water colder than 18.5º C (Howell, et al. 2008; Polivina et al. 2000, 2001, 2004, 2006). Swimmer et al. (2017) examined the potential impact of restricting fishing in this thermal band and determined that interactions between the HI SSLL and loggerhead sea turtles could have been reduced by 42% and interactions with leatherback sea turtles reduced by 44%.



EXPERIMENTAL PRODUCT

Avoid fishing between solid black 63.5°F and 65.5°F lines to help reduce loggerhead sea turtle interactions

Sea Surface Temperature : 2019/01/31

Figure 56. TurtleWatch map, January 31, 2019 (https://www.pifsc.noaa.gov/turtlewatch/today.png).

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interaction levels (aka “anticipated take levels”; McCracken 2018) using Bayesian statistical inference techniques as described in the *Approach to the Assessment* section of this biological opinion, and in greater detail in Appendix A (McCracken 2018). The predictions were based on the recorded bycatch of loggerhead sea turtles in the HI SSLL fishery). Table 61 contains the resulting predictions of future annual, 2 year, and 3 year interactions (mean and 95 percentile) for loggerhead sea turtles in the HI SSLL fishery (from McCracken 2018).

Table 61. Mean and 95 percentile of loggerhead sea turtle posterior estimated exposures in terms of anticipated interactions with fishing gear (i.e., anticipated take level; from McCracken 2018).

Period	Mean	95 percentile
Annual	16	36
2-year	31	59
3-year	47	81

Olive Ridley Sea Turtle

Hooking is the most significant stressor that olive ridley sea turtles are exposed to directly from the action. Since the shallow-set fishery re-opened in 2004 there have been a total of 10 observed interactions that have resulted in a hooking or entanglement. All 10 of these interactions were recorded with sufficient information to evaluate post interaction mortality risk using the criteria of Ryder et al. (2006). Table 62 describes the injury and release conditions of all olive ridley sea turtles in the HI SSLL fishery. Nine out of the ten olive ridley sea turtles were hooked externally or with the insertion point visible in the mouth or jaw and all gear was removed. One olive ridley was entangled and all gear was removed.

Table 62. Number and proportion of olive ridley sea turtles captured in the HI SSLL fishery between 2004 and 2018 by their injury and release condition.

Injury Category	**Line < 1/2 SCL (%)**	**Line ≥ 1/2 SCL (%)**	**No Gear)**	**Grand Total (%)**
Entangled	-	-	1(10)	1
External	-	-	5(50)	5
Insertion Not Visible	-	-	-	-
Insertion Visible	-	-	1(10)	1
Jaw	-	-	3(30)	3
Grand Total	0	0	10	10

Demographic Patterns of Exposure

As described in the *Status of the Species* section of this biological opinion, there are two listed populations of olive ridley sea turtles that are exposed to the action; the endangered Mexico population and the threatened non-breeding Mexico population. Genetic data on olive ridley sea turtles captured in this fishery can differentiate whether the turtle belongs to the Eastern Pacific or the Western Pacific but it cannot differentiate between turtles from Mexico or other Central American locations. As a result, genetic data on olive ridley sea turtles captured in this fishery suggest that 75% of the turtles are from the Eastern Pacific and 25% from the western/ Indo-Pacific. Since 2004, five of the 10 interactions have produced results and all are from the Eastern Pacific (Dutton pers. comm. December 12, 2017).

As discussed previously, large data gaps still exist in this species' demography, including age and sex distribution; growth, birth, and death rates; immigration, and emigration (Zug et al. 2006; NMFS and FWS 2007, 2014). The olive ridley sea turtles that have interacted with the HI SSLL fishery ranged in size from 48 to 64.5 cm SCL (NMFS unpublished observer data). Based on studies by Work and Balaz (2002, 2010), and Shanker et al. (2003), the olive ridley sea turtles in the HI SSLL fishery are likely immature turtles. At least one animal that was capture was likely an adult. Based on age estimation charts in Zug et al. (2006), olive ridley sea turtles interacting with the HI SSLL fishery are likely 12 to 13 years old based on the size-at-age growth function, and could fit the range of either subadult (60%) or an adult (40%) turtles

Spatial and Temporal Patterns of Exposure

We analyzed the location and dates of the interactions that occurred since 2004 to see if any spatial or temporal patterns emerged. Most of the interactions occurred throughout the action area. However, one cluster of five interactions occurred approximately 500 NM NE of Oahu in the months of March and April. Four of the five interactions occurred within 100 NM of each other and the fifth was approximately 120 NM SE from those four. These five interactions represent 83% (5/6*100) of the interactions documented in the months of March and April across all years for olive ridley sea turtles in the HI SSLL fishery. Due to the spatial distance and small number of interactions, a heat map could not be produced using ArcGIS.

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interaction levels as previously described in the leatherback section above. The predictions were based on the recorded bycatch of olive ridley sea turtles in the HI SSLL fishery. Table 63 contains the resulting predictions of future annual, 2 year, and 3 year interactions (mean and 95 percentile) for olive ridley sea turtles in the HI SSLL fishery (from McCracken 2018).

Table 63. Mean and 95 percentile of olive ridley sea turtle posterior estimated exposure in terms of anticipated interactions with fishing gear (i.e., anticipated take level; from McCracken 2018).

Period	**Mean**	**95 percentile**
Annual	1.4	5
2-year	2.9	8
3-year	4.3	11

Green Sea Turtle

Hooking and entanglement are the most significant stressors that green sea turtles are exposed to directly from the action. Green sea turtles interact with the HI SSLL fishery in low numbers. In 14 years, 10 green sea turtles have been observed in this fishery. All ten of these were recorded with sufficient information to evaluate post interaction mortality risk using the criteria of Ryder et al. (2006).

Table 64 describes the proportion of green sea turtles captured in the HI SSLL fishery between 2004 and 2018 by their injury category and release condition. Forty percent of the turtles were hooked externally and 40% were hooked in the mouth, either in the jaw or other locations of the mouth where the insertion point of the hook was visible and able to be removed. Two of the turtles were entangled. All ten green sea turtles caught in the HI SSLL fishery between 2004 and 2018 were released alive with no gear attached.

Table 64. Number and proportion of green sea turtles captured in the HI SSLL fishery between 2004 and 2018 by their injury and release condition.

Injury Category	**Line < 1/2 SCL (%)**	**Line ≥ 1/2 SCL (%)**	**No Gear (%)**	**Grand Total**
Entangled	-	-	2(20)	2
External	-	-	4(40)	4
Insertion Not Visible	-	-	-	-
Insertion Visible	-	-	2(20)	2
Jaw	-	-	2(20)	2
Grand Total	0	0	10	10

Demographic Patterns of Exposure

As previously discussed in the *Status of Listed Resources*; since 1996, NMFS has evaluated the mtDNA of 19 green sea turtles captured in the SSLL fishery. The majority of the turtles caught in the HI SSLL are predominantly comprised of animals from the Eastern Pacific DPS and the Central North Pacific DPS (Dutton, pers. Comm. 17 December 2018). An estimate of the catch composition to date, using Bayesian mixed stock analysis suggests that a small number (<2%) of the green sea turtles may comprise animals from the East Indian - West Pacific DPS, Central West Pacific DPS, Southwest Pacific DPS, and the Central South Pacific DPS. To date, no animals from these additional four DPS's have been captured. However, the potential exists based on the movement of juvenile green sea turtles among these DPS within the Pacific (see Seminoff et al. 2015 for a discussion of genetics and tagging studies evaluated in determining DPS discreteness).

Looking at the interactions between 2004 and 2018, all ten green sea turtles were measured in the HI SSLL fishery as shown in Table 65. There is not enough information available to identify green sea turtles captured in this fisher to sex, and the size of those captured does not provide information assist to provide us any indication of gender. However, as green sea turtles do not mature in this region until reaching 80 cm (USFWS and NMFS 1998; Zug et al. 2002), we suspect that the largest green sea turtle may have been an adult and the remaining individuals were immature. Two turtles, those in the 65 cm range, may have been sub adults. Table 65 also shows the age estimate based on skeletochronology aging data presented by Zug and Balazs (2000) and Zug et al. (2002). We estimate these turtles ranged between 4 years old for the individual at 26 cm, upwards of 30 years old for the turtle at 88.50 cm.

Spatial and Temporal Patterns of Exposure

Table 65. Straight carapace length measurements and estimated age for all green sea turtles which have interacted with the Hawaii shallow-set longline fishery between 2004 and 2018 (Estimates are based on Zug and Balaz 2000, and Zug et al. 2002).

SCL (cm)	Estimated Age (yrs)
26.00	4
34.50	6
38.00	8
38.00	8
46.00	11
50.00	14
55.50	18
65.00	23
65.50	23
88.50	30

We plotted the 10 green sea turtle interactions that have occurred in the HI SSLL fishery between 2004 and 2018 in ArcGIS. No spatial patterns are apparent from the data due to the small number of data points. Spatial remoteness between the data does not allow for viable density mapping products to be produced. Interactions span a latitudinal distance of approximately 1,197 nautical miles (nm) across the action area. The mean distance between interaction locations (n=10) is approximately 255 nm (range 157 nm to 606 nm).

Temporally, some consistency in interactions occurs between quarters 2 and 3 looking across all years (2004-2018). One interaction occurred in February, 3 in March, 4 in April, and 2 in May. 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.

Predicted Future Exposure to the Fishery

- NMFS developed predictions of future interaction levels as previously described in the leatherback section above. The predictions were based on the recorded bycatch of green sea turtles in the HI SSLL fishery. Table 66 contains the resulting predictions of future annual, 2 year, and 3 year interactions (mean and 95 percentile) for green sea turtles in the HI SSLL fishery (from McCracken 2018).

Table 66. Mean and 95 percentile of green sea turtle posterior estimated exposure in terms of anticipated interactions with fishing gear (i.e., anticipated take level; from McCracken 2018).

Period	**Mean**	**95 percentile**
Annual	1.4	5
2-year	2.8	8
3-year	4.1	10

6.2.3.2 Oceanic Whitetip Shark

Between 2004 and 2018, 875 oceanic whitetip sharks were caught in the HI SSLL fishery (Figure 57; NMFS Observer program unpublished data).

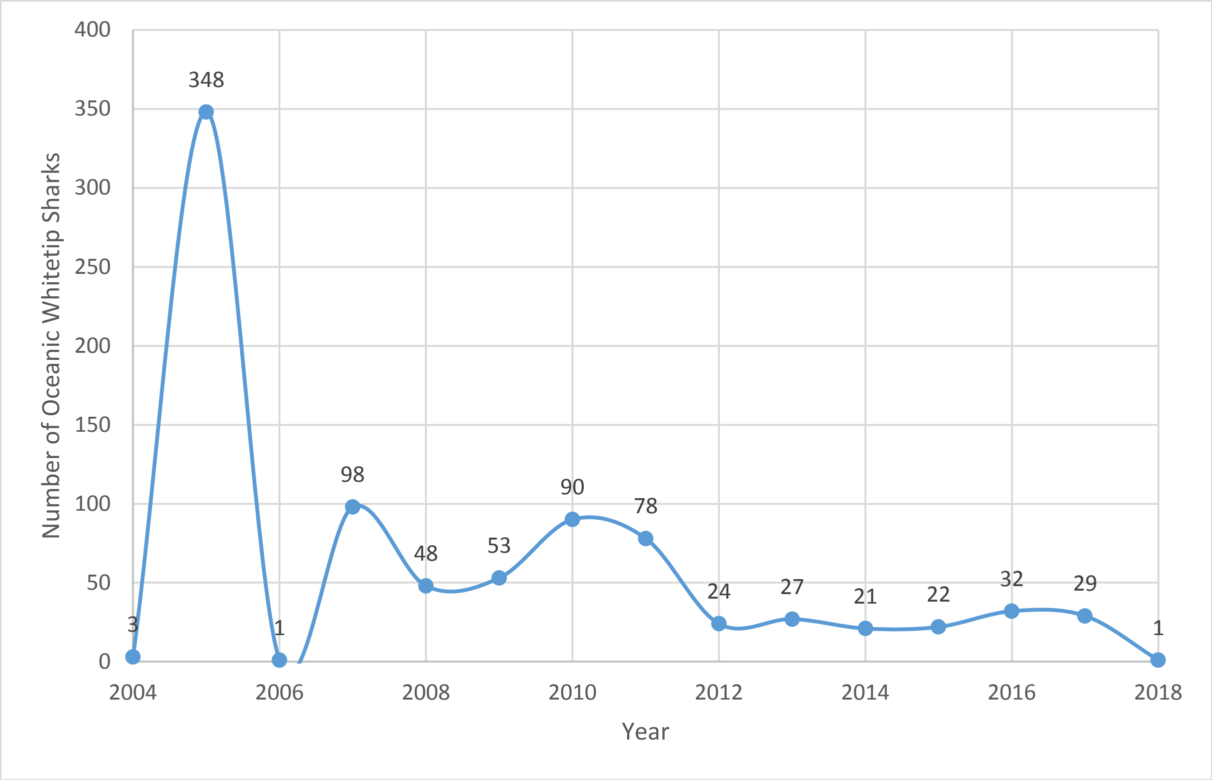


Figure 57. Observed oceanic whitetip shark interactions in the Hawaii shallow-set fishery, 2004-2018.

Most oceanic whitetip sharks are released alive (88%) and the number of individual sharks retained by the HI SSLL fishery appears to have declined in recent years. As previously discussed in the *Status of Listed Resources,* the oceanic whitetip shark is not only experiencing overfishing in the Western and Central Pacific, but the stock is currently in an overfished state (Rice and Harley 2012; Young et al. 2016). HI SSLL fishery data, also incorporated in the Status Review (i.e. Young et al. 2016), supports these conclusions as the number of interactions have decreased substantially in our action area over time as shown in Figure 57, and occurred before the Papahanaumokuakea Marine National Monument (hereinafter referred to as "Monument") declaration in 2016 which shifted effort to the east. Also previously discussed in the *Status of Listed Resources* and *Environmental Baseline*, Young et al. (2016) indicated that the oceanic whitetip shark population in the operational range of the fishery might have stabilized in recent years based on a preliminary analysis of annual standardized CPUE from 1995-2014. Since then, observer data from 2015 and 2016 show nominal CPUE was approximately same or slightly higher than 2014 (NMFS Observer Program, unpubl. data), however these are unstandardized data and should be interpreted with caution. These data were also combined with data from the HI DSLL which operates in a different manner than the shallow set and has a much larger number of interactions with this species. As this fishery is unique from the deep set and considered a separate action, we assess the data as such.

Average at-vessel mortality of oceanic white tip sharks is 12% in the HI SSLL fishery. Supporting data and the condition at the end of each shark interaction can be found in Table 67 (NMFS Observer Program unpublished data). However, it is unknown how many of these sharks died after release due to injuries and trauma sustained during capture and handling. An extensive review of the literature revealed that post release mortality rates are not available for oceanic whitetip sharks in any fishery and is a data gap for the species (Musyl et al. 2011; Young et al. 2016).

As previously discussed in the *Action Area,* a number of exclusion zones are present and include areas once previously fished by the fleet. Four interactions occurred in the boundaries defined by Presidential Proclamation 8031[8]. Two occurred before the designation in 2005, and two afterward in 2009. The Monument was expanded to include the UWS EEZ around the NWHI in 2016. Of the 875 interactions that occurred between 2004 and 2018, 484 interactions (55%) occurred within the newly established Monument boundaries, and 391 were outside (45%). Therefore, as the HI SSLL fishery can no longer fish this area, we assessed all interaction to date as well as any interactions that occurred after the monument declaration went into effect resulting in a shift of fishing effort eastward from the NWHI.

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[8] Through Proclamation 8031 of June 15, 2006, as amended by Proclamation 8112 of February 28, 2007, the President established the Papahānaumokuākea Marine National Monument, to protect and preserve the marine area of the Northwestern Hawaiian Islands and the historic and scientific objects therein.

Table 67. Oceanic whitetip shark interaction data displaying condition at end of interaction to determine percentage released alive in the Hawaii shallow-set longline fishery.

Year	**Number of Interactions**	**NMFS Special Study**[9]	**Finned**	**Retained**	**Released dead**	**Released Alive**
2004	3					3
2005	348			6	26	316
2006	1					1
2007	98			3	4	91
2008	48			6	2	40
2009	53		1	11	2	39
2010	90			14	3	73
2011	78			2	7	69
2012	24			1	1	22
2013	27				2	25
2014	21				3	18
2015	22				2	20
2016	32				3	29
2017	29	6			1	22
2018	1				1	
Totals	875	6	1	43	57	768

Demographic Patterns of Exposure

In the HI SSLL fishery, sharks may not be boarded for a number of reasons, including poor weather, the shark's size, condition at landing, efficiency of maintaining fishery operations, safer and less stressful for the animal, and for the crew's safety. Therefore, observers estimate lengths based on their best professional judgement and with known measurements around the rail of the vessel. Of the 875 observed interactions of oceanic whitetip sharks between 2004 and 2018, length estimates were provided for 530 individuals. The average estimated length was 146 cm (4.8 ft., range 2 ft. to 8 ft.).

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[9] NMFS conducted species specific research in 2017 that contained atypical handling and release procedures to attach scientific instrumentation to sharks.

When feasible, observers collect actual length measurements. Currently the protocol is to measure every third fish, regardless of species (NMFS 2017). The actual number of oceanic whitetip sharks boarded and measured during this time frame was 22 individuals and included six measurements that were out of protocol (not the 3rd fish). These measurements were opportunistically taken by the observers. The average lengths were as follows; pre-caudal 156.33 cm (n=9); fork length of 159 cm (n=16); outside-of-protocol pre-caudal lengths 136.40 cm (n=5); and out-of-protocol fork length of 152 cm (n=6). Average of all pre-caudal lengths resulted in 149 cm (n=14). The total average overall fork length is 157 cm (n=22). The largest recorded specimen had a fork length of 205 cm and a pre-caudal length of 200 cm. When comparing this data to Joung et al. (2016, see also Figure 33), the data shows the average shark that interacts with gear in this fishery is approximately 3 years old with a maximum age of 6 years. However, length data only exists for 2.5% (22/875*100) of the observed specimens. In the North Pacific, females mature at about 168-196 cm TL, and males at 175-189 cm TL, which corresponds to an age of 4 and 5 years, respectively (Seki et al. 1998). However, more recently Joung et al. (2016) determined a later age of maturity in the North Pacific of approximately 8.5-8.8 years for females and 6.8-8.9 years for males. Therefore, 6 of the 22 measured sharks were likely adults and the remainder (n=16) were juveniles.

Of the 875 interactions, 151 oceanic whitetip sharks were identified according to sex, with 58% (87) identified as females and 42% (64) male oceanic whitetip sharks. The remaining (n=724) were not identified (82.75%). While sharks can be visually sexed with ease by looking at the ventral surface (i.e. underside) for claspers, signifying a male shark. As previously discussed, many variables determine whether this data can be collected on an individual shark. Each interaction is unique and the observer cannot always maintain a visual line of sight on the animal through the entire interaction. Poor weather and sea state, the shark's size, condition at landing, whether the crew cuts the line or snaps from pressure, animal and crew safety can all influence an observers ability to identify claspers on a shark. Based on this sample, the HI SSLL fishery interacts with more females than males, at a ratio of approximately 3:2. While the fishery appears to interact with a slightly larger proportion of females, these data should be regarded with some caution. Collection protocols for sharks and shark data have changed in the 14 year period considered, additionally some measurements were collected opportunistically throughout the assessed interval.

Spatial and Temporal Patterns of Exposure

All interaction data with oceanic whitetip sharks from the SSLL fishery were reviewed from 2004 to when the fishery closed on May 4, 2018. Clear spatial and temporal patterns in interactions with oceanic whitetip sharks are evident both before and after the Monument declaration occurred. Comprehensively, most interactions (97%) occur between April and July, peaking in May and June (Figure 58).

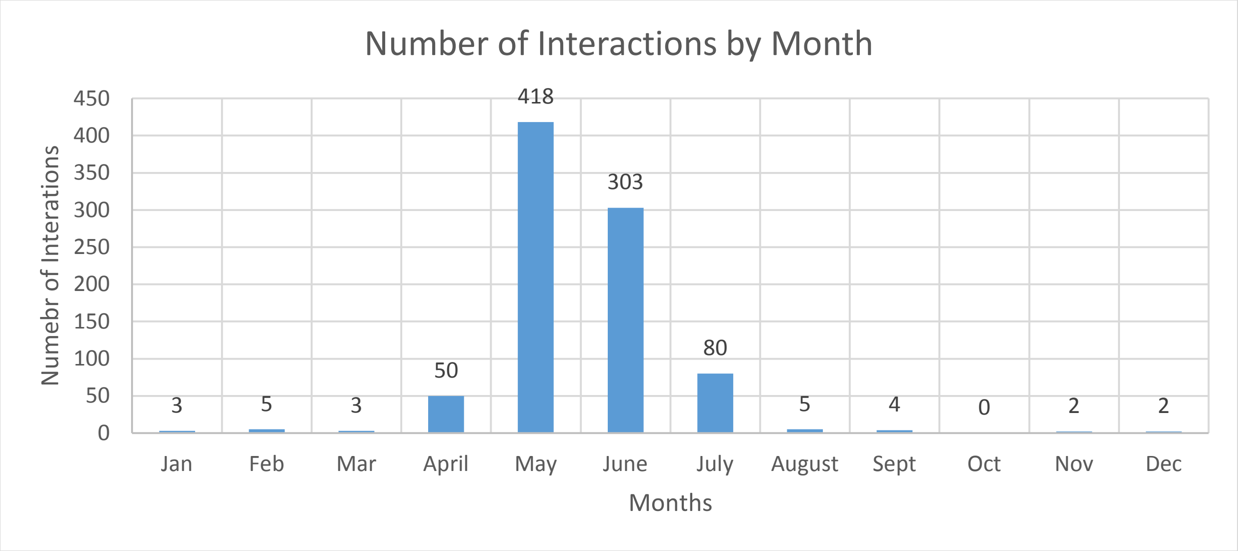

Figure 58. Total number of oceanic whitetip shark interactions by month in the Hawaii shallow-set longline fishery, 2004 through 2018 (n=875).

Using haul back locations, we mapped all sets where interactions occurred using GIS (Figure 59). The majority of the interactions occur on the north side of the Hawaiian Islands in a linear band stretching southeast to northwest within the limits of the EEZ, both inside and outside of the Monument. One hotspot is identified on the south portion of the EEZ far to the west, now within the limits of the Monument. Comparing the data to comprehensive longline effort (Figure 60) we can see the majority of the fishing effort is to the northeast where interactions are not occurring. It should be noted that the maps were sized the same to show relation and comparison of fishing effort versus interaction data. Fishing effort continues to the northeast, and stretches close to the California coast (see *Action Area*).

To provide some additional baseline reference to these figures, both maps contain the same key features. They include the Hawaiian Islands chain, the EEZ shown as a light grey line, the longline prohibited fishing area around the MHI is depicted in green, the longline exclusion zone around the NWHI is in red, and the expanded Monument boundary is shown in bold black line. The island to the south of the NWHI is Johnston Atoll and its EEZ. Some maps may exclude certain features depending on what information is being illustrated.

The ArcMap kernel density tool was used to calculate a density which is based on point features using interaction and haul data along with a range of unique values as displayed in the maps key. 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 more fluid edges. As the density increases, the color shifts from a cooler blue to a more dynamic warmer tone, also known as a color ramp. Therefore, blue areas have lower interaction or effort rates than those areas with orange or reds. The key was adjusted to illustrate low versus high density values.

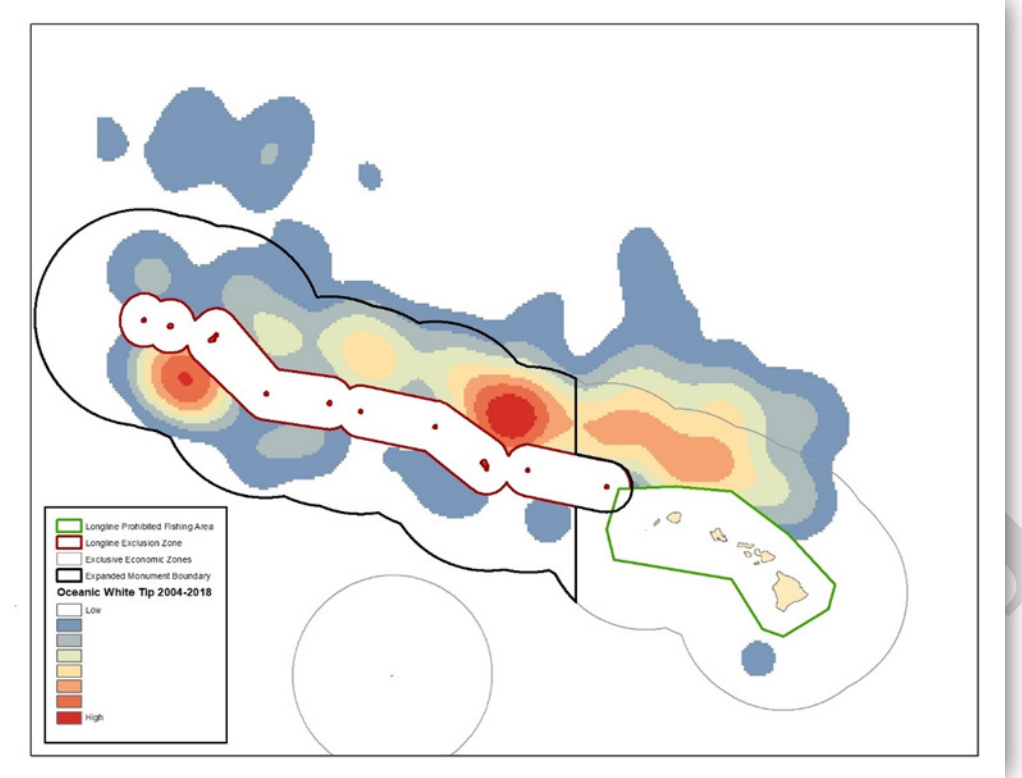

Figure 59. Geographical representation of aggregate oceanic whitetip shark interactions with the Hawaii shallow set longline fishery between 2004 and 2018.

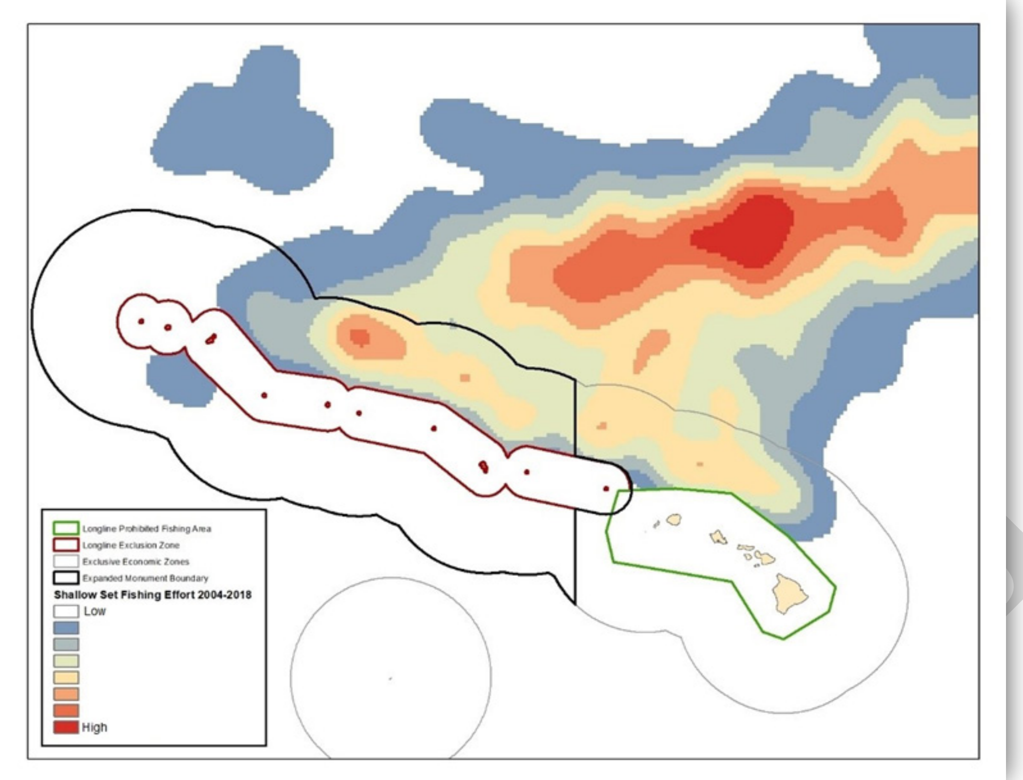

Figure 60. Snapshot of aggregated Hawaii shallow set longline fishing effort between 2004 and 2018.

As Figure 58 showed a clear temporal pattern of interactions with a sharp increase between April and July. We looked at the effort data for April, May, June, and July across all years to compare where sharks interactions are occurring geographically (Figure 59) when compared to aggregate fishing effort (Figure 61). This allowed us to determine whether interactions are within a certain region or distributed throughout the fishing fleets' range during these months. When comparing these data to Figure 59, it is apparent that shark interactions are occurring closer to the islands and not just where fishing effort is greatest. Although effort is higher along the north side of the NWHI, interactions are not occurring to the north where considerable effort was also present. Figure 62 depicts the aggregated effort with the Monument removed, as these areas can no longer be fished by the fleet.

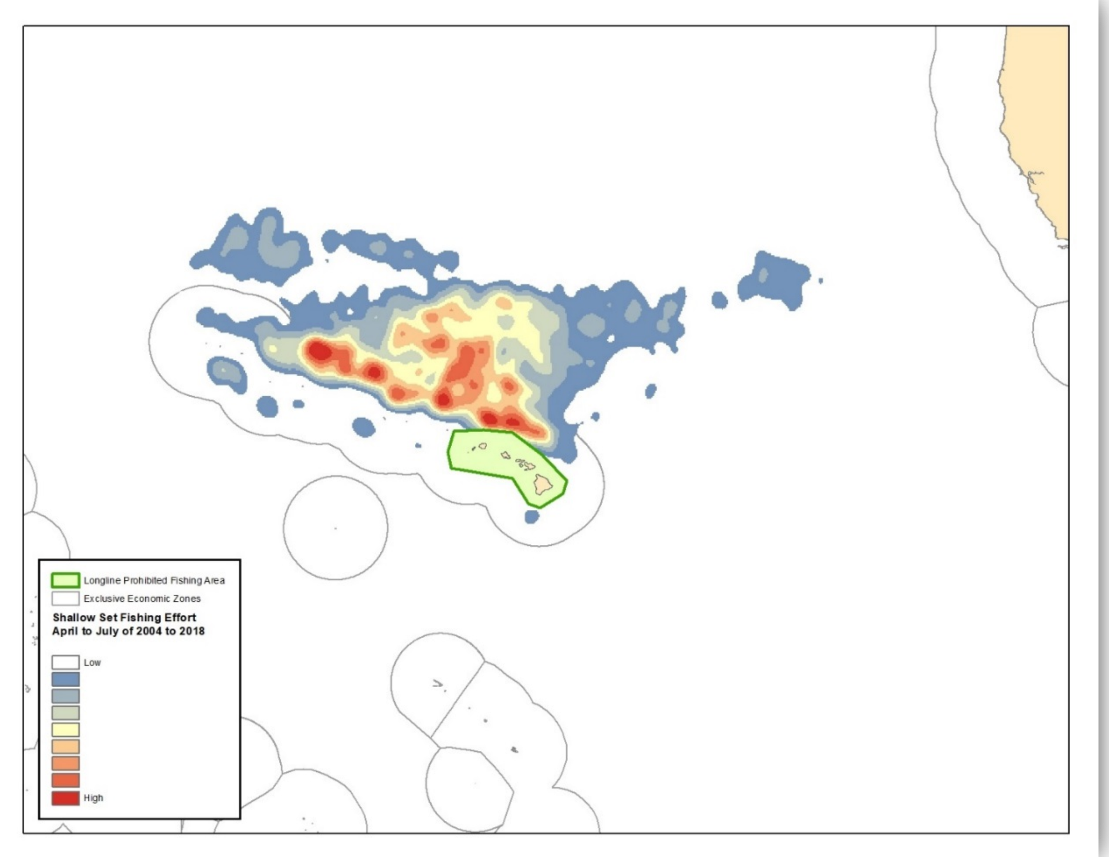

Figure 61. Aggregated fishing effort from 2004 to 2018, during only the months of April, May, June and July.

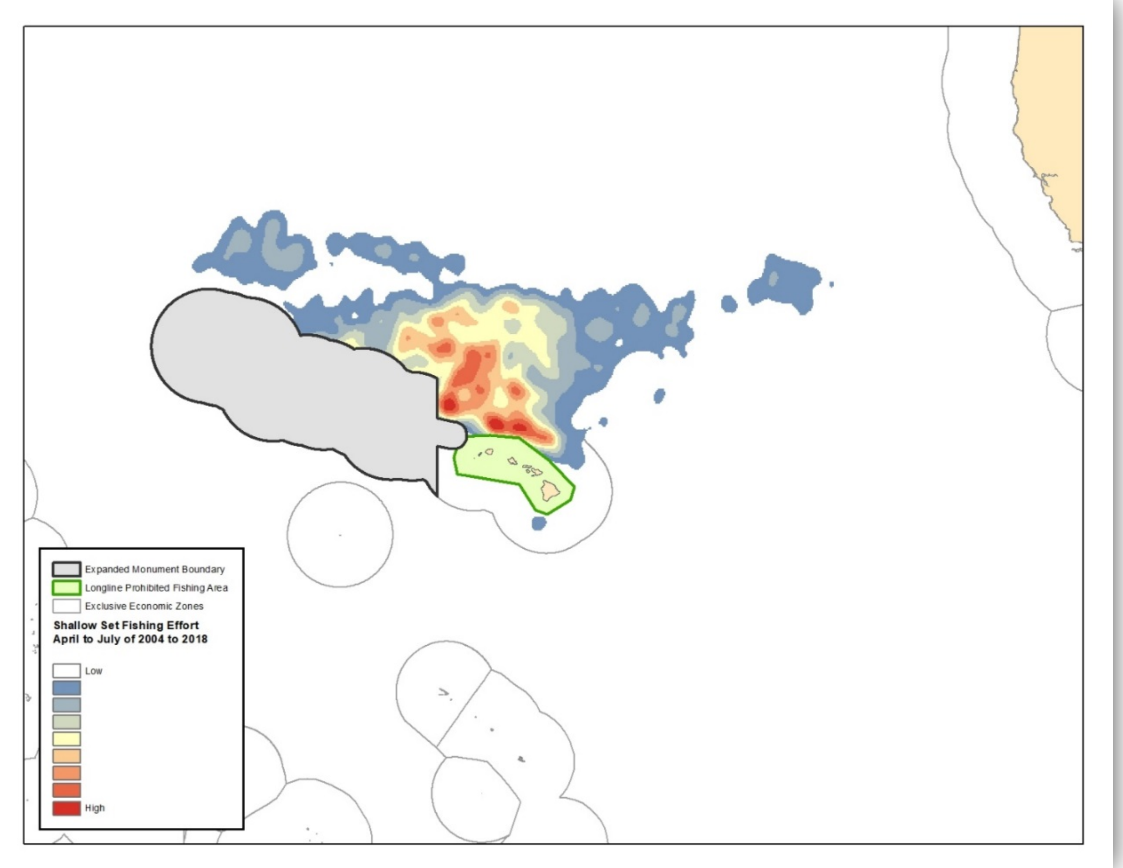

Figure 62. Aggregated fishing effort from 2004 to 2018, during the months of April, May, June and July excluding the Monument.

Between 2004 and 2018, a total of 48 vessels caught oceanic whitetip sharks in the HI SSLL fishery. Table 68 lists interactions (n= 875) by four interaction ranges: 25 vessels have less than 10 interactions, 9 vessels have between 10 and 20 interactions, 11 vessels have between 20 and 50 interactions, and 3 vessels have over 50 interactions (Table 68). The three vessels accounted for 35% (95% CI: 0.41, 0.48) of the total interactions. These data show how a small number of vessels can have a disproportionate effect on a population by fishing in an area where larger concentrations of sharks may be present at a certain time of year.

Table 68. Number of vessels in the Hawaii shallow-set longline fishery and the number of interactions with oceanic whitetip sharks between 2004 and 2018

Number of Vessels	Range (Number of Interactions)
25	<10
9	10-20
11	20-50
3	>50

Table 69 shows the collated total of the three fishing vessels and what percentage of interactions are attributed to the group per year compared to the total number of interactions by the fleet along with a lower and upper confidence interval. These data shows how a small number of vessels can have a disproportionate effect on a population by fishing in an area where larger concentrations of sharks may be present at a certain time of year. These three vessels range between 0 and 66.67% of the interactions any given year. However it should be noted the lower and upper range have a low number of interactions for those years. We draw your attention to the number and percentage of interactions for 2017 and 2018, as this fishing effort will be discussed later.

Table 69. Number of oceanic whitetip shark interactions attributed to the top three vessels, broken down by year and shown with lower and upper 95% confidence intervals.

Year	Top 3 Vessels: Total Number of Interactions	Total Number of Interactions by Year	Percent Interactions	95% CI: Lower	95% CI: Upper
2004	2	3	66.67	0.21	0.94
2005	96	348	27.59	0.23	0.33
2006	0	1	0.00	0.00	0.79
2007	42	98	42.86	0.34	0.53
2008	18	48	37.50	0.25	0.52
2009	9	53	16.98	0.09	0.29
2010	32	90	35.56	0.26	0.46
2011	41	78	52.56	0.42	0.63
2012	12	24	50.00	0.31	0.69
2013	11	27	40.74	0.25	0.59
2014	6	21	28.57	0.14	0.50

Year	Top 3 Vessels: Total Number of Interactions	Total Number of Interactions by Year	Percent Interactions	95% CI: Lower	95% CI: Upper
2015	2	22	9.09	0.03	0.28
2016	25	32	78.13	0.61	0.89
2017	13	29	44.83	0.28	0.62
2018	0	1	0.00	0.00	0.79

Between 2004 and 2018, a total of 391 interactions have occurred outside of the Monument boundaries as shown in Figure 63. A clear geographical trend in interactions is apparent on the north side of the MHI. This is relevant as the fishing fleet can still fish these areas and a larger density of interactions are occurring within this specific portion of the *Action Area* when compared to the fishing effort (see Figures 6 and 61).

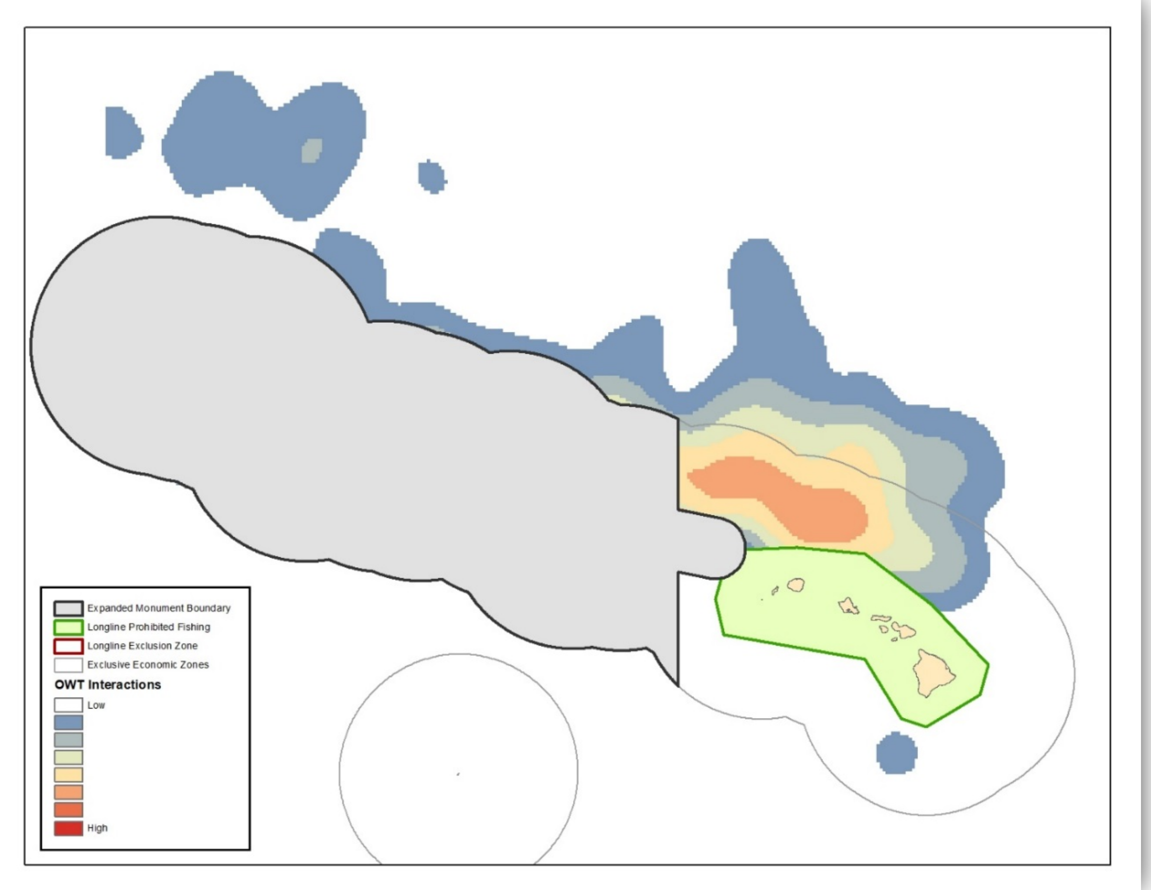

Figure 63. Aggregate oceanic whitetip shark interactions from 2004 to 2018 that occurred outside of the Monument (n=391).

Interaction data was then assessed to look at those interactions which occurred after the Monument was designated (n=30). From August 2016 to May 2018, 96.67% of the interactions (25/30*100) occur in May, shown in Figure 64. Spatial analysis also reveals a hotspot on the north side of the MHI (Figure 65). Of the 30 interactions, 11 vessels interacted with oceanic whitetips during this time frame. The top three vessels account for approximately 45% (95% CI: 0.27, 0.61) of the interactions post Monument designation. It should be noted that no interactions occurred after August 2016, and the fishery was closed in May of 2018, effectively assessing 23 months of fishing effort. Even though fishing effort shifted eastward from the monument, interactions still occurred within the same time frame thorough the calendar year (Figure 64) when compared to aggregated effort (Figure 2). Please note, Figure 58 is the total aggregate of interactions and includes records from the post monument designation.

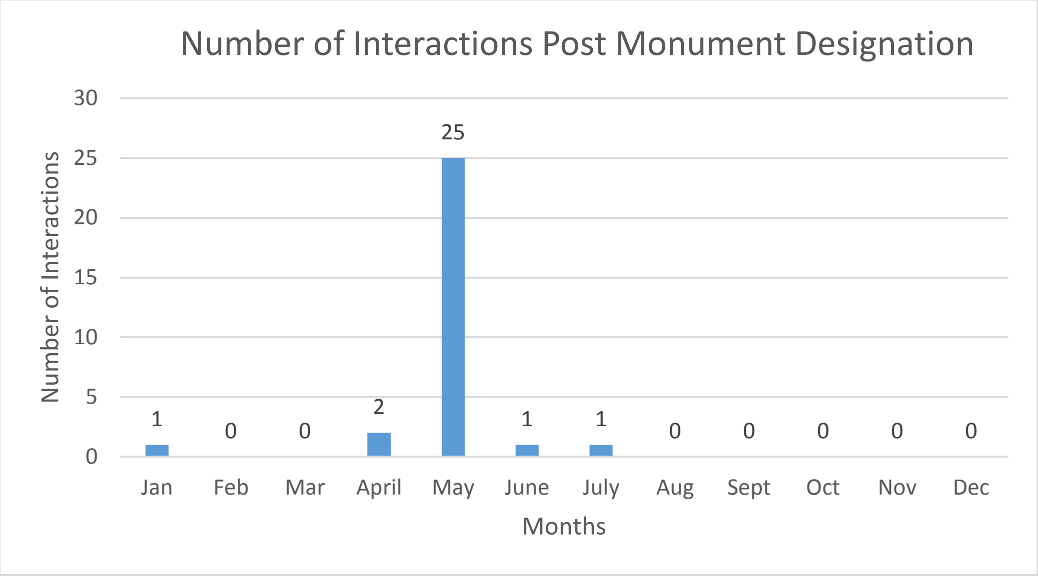

Figure 64. Number of oceanic whitetip shark interactions after the Monument designation; from August 2016 to 2018, displayed by months (n= 30).

Figure 65. Interaction heat map with oceanic whitetip sharks (n=30) post Monument designation (August 2016 to May 2018).

However, since 2016 was a transition year for the fishery, we looked at this year in more detail. All interactions that occurred in 2016 were in the same four month window as all other interactions. However, this was prior to the Monument expansion on August 26, 2016. A total of 32 interactions occurred for the year, 13 interactions occurred in what is now the Monument boundaries, and 19 interactions were outside. Inclusion of the interactions that occurred outside of the Monuments boundaries in 2016 adjusts the hotspot north of the MHI slightly as more interactions occurred in this same area (Figure 66). This shows that this same area is being fished over multiple years during the same time period by multiple vessels.

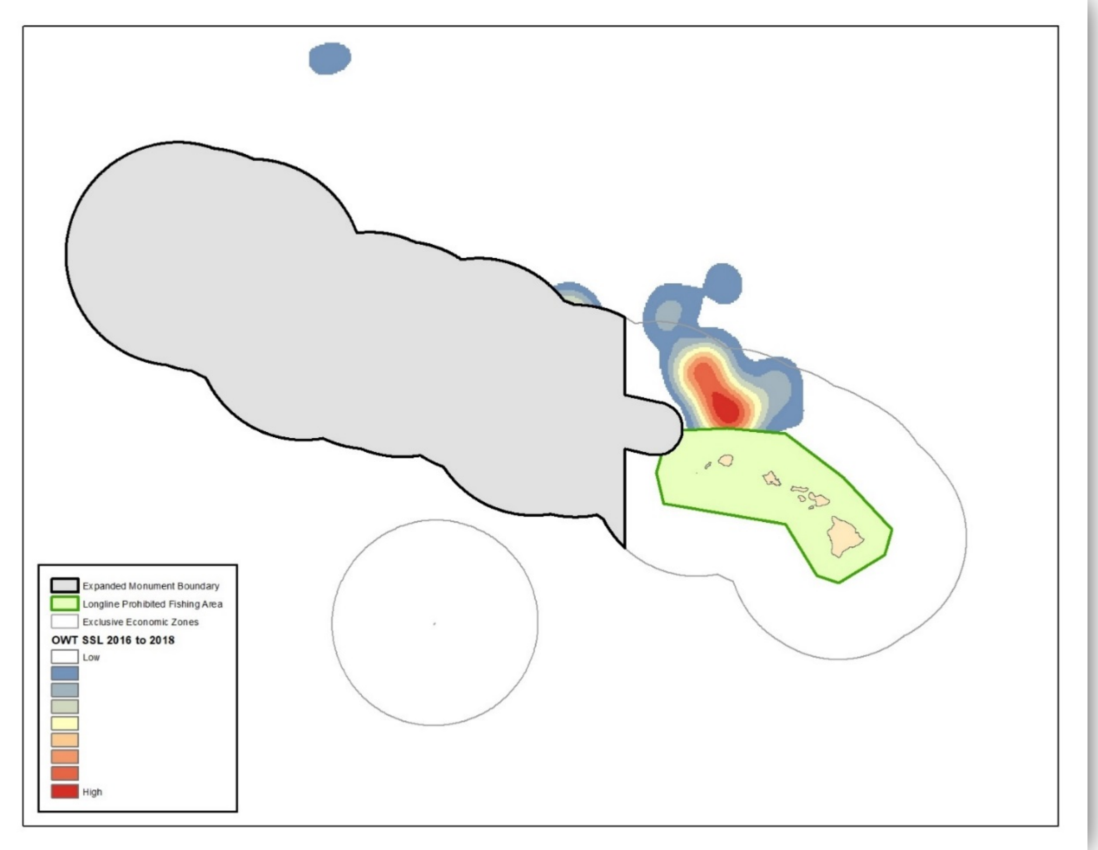

Figure 66. Post monument designation with outside interactions incorporated showing all interactions that occurred outside of the Monument from 2016 to 2018 (n=49).

We then looked at the fishing effort data for pre- and post- Monument expansion (Figure 67 and Figure 68) to compare if fishing effort was focused in this area or distributed elsewhere. Figure 67 does not depict the Monument boundaries, as this data refers to a time period when the fleet was fishing these areas. It is also apparent from the data that fishing effort was focused to the northeast during this 23 months of fishing effort. However, shark interactions are occurring in a specific region of the fishing fleets range and during a specific portion of the calendar year.

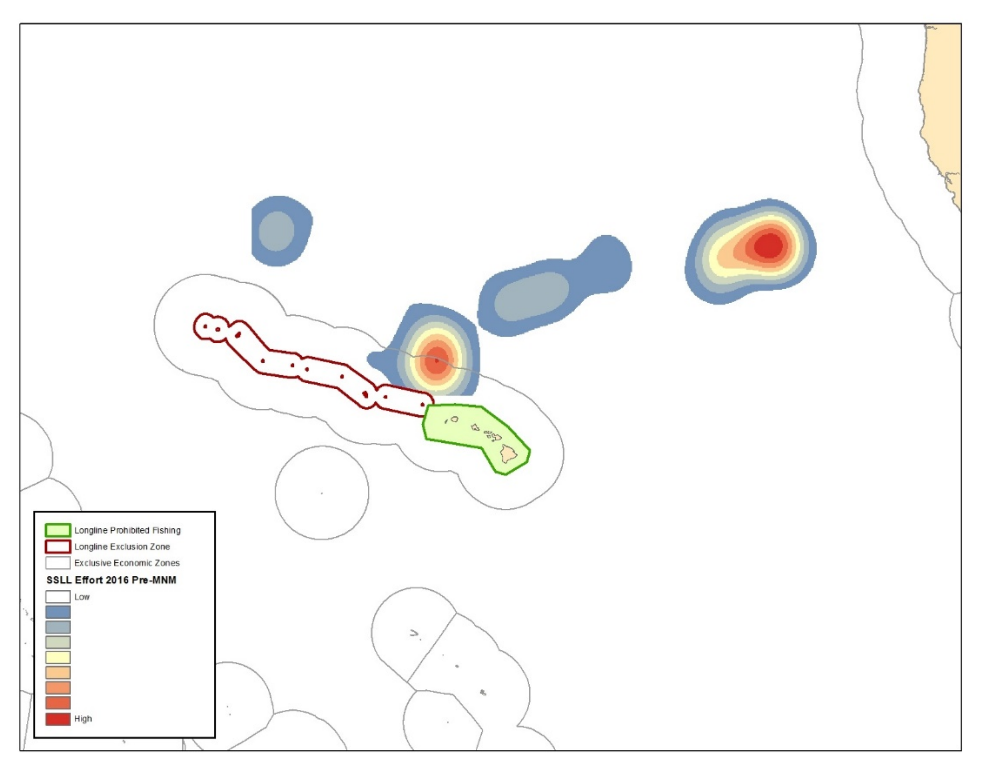

Figure 67. Aggregated fishing effort in 2016 prior to the Monument designation.

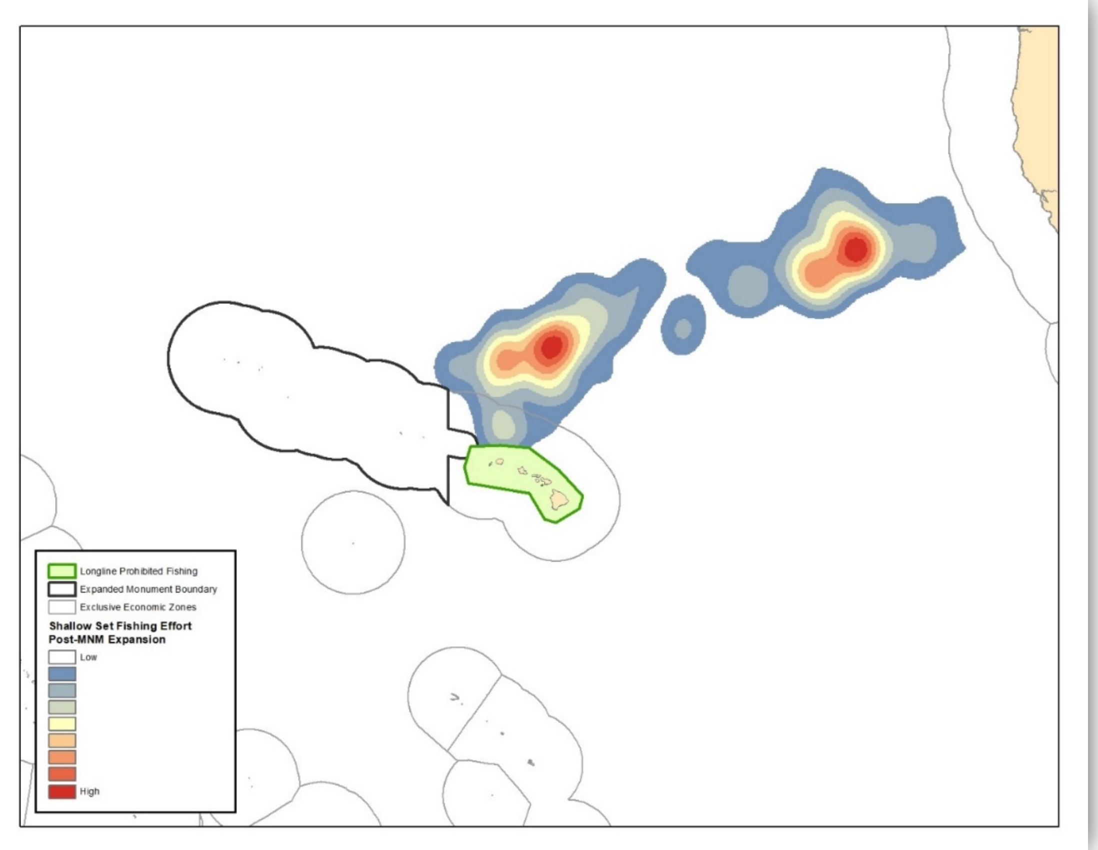

Figure 68. Aggregated fishing effort post Monument expansion; August 26, 2016 to May 2018.

Lastly, a cluster was identified when projecting oceanic whitetip shark and giant manta ray interactions together. Interactions overlap at an area north of the far west end of the EEZ (Liliuokalani Ridge). This area accounted for 31 (out of 80) oceanic whitetip shark interactions and half of the giant manta ray interactions in the month of July across all years. Due to the low number of giant manta ray interactions. This general area is shown in Figure 60 and Figure 69. It appears that the area may be an important aggregation area due to extensive vertical relief in the bathymetrics of the sea floor where upwelling and congregation of prey species would be expected to occur (see also Figure 64).

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interaction levels (aka “anticipated take levels”; McCracken 2018) using Bayesian statistical inference techniques as described in the *Approach to the Assessment* section of this biological opinion, and in greater detail in Appendix A (McCracken 2018). The predictions were based on the recorded bycatch of oceanic whitetip sharks in the HI SSLL fishery. Table 70 contains the resulting predictions of future annual, 2 year, and 3 year interactions (mean and 95 percentile) for oceanic whitetip sharks in the HI SSLL fishery (from McCracken 2018).

Table 70. Mean and 95 percentile of oceanic whitetip shark posterior estimated future exposures in terms of anticipated interactions with fishing gear (i.e., anticipated take level, from McCracken 2018).

Number of years	Mean	95 percentile
1	72	227
2	145	371
3	216	464

We then assessed the number of interactions that occurred within the area now designated as the Monument (n = 484 interactions) which resulted in 55% of the interactions to date. The Biological Evaluation and PIFSC analysis did not take into account the Monument designation. Therefore, we adjusted the projected annual take limits since this area is no longer available to the fishing fleet. We subtracted the interactions that occurred within the geographical boundaries of the Monument from the mean and 95th percentile projected by McCracken (2018) by multiplying the percentage by the value and then subtracting it from the original value. Rounding was not completed until the final step thus accounting for the individual appropriately. Results are shown in Table 71.

Table 71. Mean and 95 percentile of oceanic whitetip shark posterior estimated exposure in terms of anticipated interactions with fishing gear (i.e., anticipated take level) adjusted to reflect changes in fishing effort due to the Monument designation.

Number of years	Mean	95th percentile
1	32	102
2	65	166
3	97	299

6.2.3.3 Giant Manta Ray

Considering the filter-feeding nature of mobulid rays, the relatively low number of interactions in longline fisheries may represent a very low selectivity of this particular type of fishing gear (Mas et al. 2015). In fact, similar to leatherback sea turtles, the giant manta ray tends to be more vulnerable to entanglement and foul hooking as opposed to being hooked in the mouth (Sales et al. 2010; Domingo et al. 2012). In addition, giant manta rays are large animals like leatherback sea turtles, thus, they are seldom landed and brought onboard the vessel due to their size. Therefore, they tend to be released with at least the hook attached and often with trailing gear.

Interactions between ray spp., including giant manta rays, in the HI SSLL fishery are listed in Table 72. NMFS' observers are instructed to document interactions with ray by species when possible, but observations can include giant manta rays, mobula (devil rays), *Mobulidae* spp. and unidentified rays. When the animal breaks the line or is able to come free of the hook before being pulled alongside the vessel, the observer may not be able to distinguish whether the ray species is a giant manta ray. In this situation, observers are instructed to use the classification

Mobulidae spp. Likewise, if the observer is unable to identify the ray species as a manta, they are instructed to use the unidentified ray classification.

Table 72. Number of interactions and caught and released condition of rays captured in the HI SSLL fishery between 2004 and 2018.*The caught and released dispositions are for the giant manta ray, *Mobulidae* spp. and unid. ray only.

Number of interactions with ray species and their return condition									
Year	**Giant Manta Ray**	**Alive**	**Dead**	**Mobulidae spp.**	**Alive**	**Dead**	**Unid. Ray**	**Alive**	**Dead**	**Mobula* (Devil Ray)**
2004										
2005										3
2006										
2007	5	3	2							2
2008				1	1	0				2
2009										3
2010	6	6	0	1	1	0				10
2011	3	1	2	2	2	0				5
2012										2
2013										
2014	1	1		1	1	0				4
2015				2	2	0				7
2016				3	3	0	1	1	0	9
2017	2	2	0	4	4	0				5
2018										
Total	17	13	4	14	14	0	1	1	0	52

SFD combined the unidentified ray with the *Mobulidae* spp. category for the purpose of this analysis as there has been only one interaction in the fishery since 2004 reported as an unidentified ray. In addition SFD used the ratio of confirmed giant manta rays and mobula (devil ray) to estimate the potential giant manta rays included in the *Mobulidae* spp. category. In addition to using SFD's procedure for distribution of the unknowns we also used the Wilson score method (Table 73). Both methods yielded the similar results but the Wilson score method provides us with confidence intervals (Table 74) and we incorporated 2017-2018 data. From 2004 to 2018, the fishery recorded 52 mobula (devil ray) interactions (75% of confirmed mobula and giant manta ray combined) and 17 giant manta ray interactions (25% of confirmed mobula and giant manta ray combined). We used the resulting ratio to estimate the proportion of unidentified *Mobulidae* that may have been giant manta rays. We added the resulting mean estimate of giant manta rays (see Table 74) to the number of observed interactions with giant manta rays to calculate the total number of interactions the fishery likely had with giant manta rays. In total, we estimate the HI SSLL fishery had 21 interactions with giant manta rays (17 observed plus 4 (unidentified) giant manta rays) between 2004-2018.

Table 73. Estimated proportions of the different species of *Mobulidae* rays captured in the shallow-set long line fishery using observer data and the Wilson score method.

Species	**Original Data**			**Proportions**		**Confidence Interval for Proportions**	
	Number of Individuals Assigned to Species	**Number of Individuals Not Assigned to Species**	**Sample Size**	**p (Species Proportion)**	**1 – p (not Species)**	**Wilson LCI (Species)**	**Wilson UCI (Species)**
Giant Manta Rays	17	52	69	0.25	0.75	0.1017	0.1447
Mobulas (Devil Rays)	52	17	69	0.75	0.25	0.1017	0.6519

Table 74. Mean estimate (with lower and upper confidence intervals) of the unknown Mobulidae spp that were likely giant mantas or mobulas (estimates generated from data in Table 73).

Unknown *Mobulidae* spp. 15			
Species	**Mean Estimate**	**Lower 95% Confidence Interval**	**Upper 95% Confidence Interval**
Giant Manta Rays	4	2	5
Mobulas (Devil Rays)	11	10	13

Demographic Patterns of Exposure

The sex of giant manta rays that this fishery interacts with is unknown because the animals typically cannot be boarded, nor identified by sex during an interaction. However, one giant manta ray captured in 2007 was identified as a male. Currently, observers are not instructed to measure or estimate the size of giant manta rays.

Spatial and Temporal Patterns of Exposure

Our analysis of giant manta ray and other *Mobulidae* ray interaction data revealed both spatial and temporal trends. Spatially, interactions with all ray classifications occurred in three discrete areas, the Liliuokalani Ridge, Musicians Seamounts, and the southern region of Musicians Seamounts north of the Hawaiian ridge (Figure 69). Giant manta rays are known to aggregate around bathymetric features such as seamounts and ridges (Kashiwagi et al. 2011).

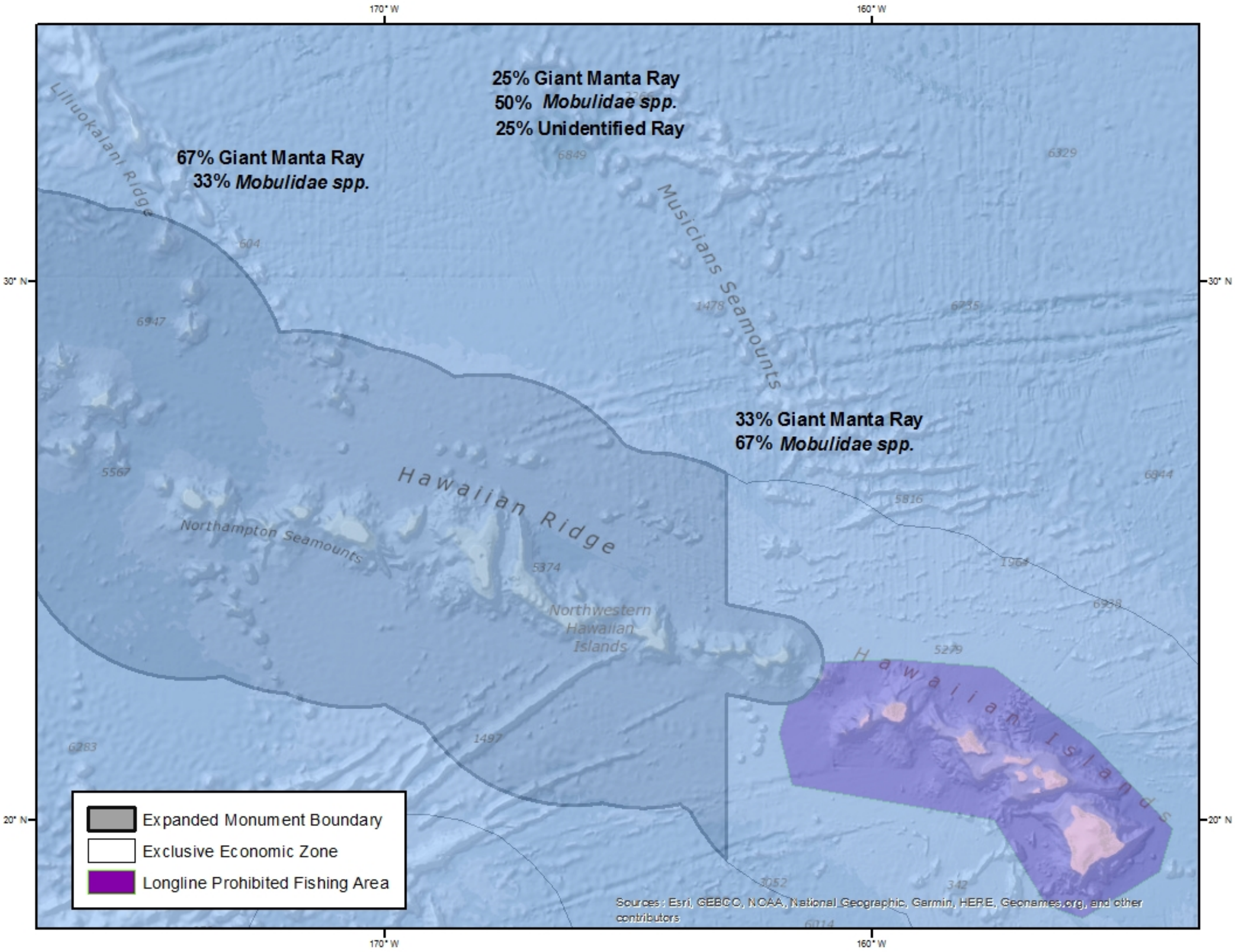

Figure 69. General location of interactions with giant manta ray, manta/mobula and unidentified ray and the percentage of each classification in each area.

The percentages were calculated per area; the majority of giant manta rays were observed in the Liliuokalani Ridge while most of the rays in *Mobulidae* spp. classification occurred in the region north of the Hawaiian ridge. Temporal trends were also evident and occurred over the second and third quarters; April, May, June, and July and August and September (Figure 70). Overall, the third quarter (July-September) is when most interactions occurred . Specifically, 41% of all interactions occur during the month of July. During these two quarters, fishing effort in the HI SSLL fishery tends to be low and the effort shifts closer to the Hawaiian Islands.

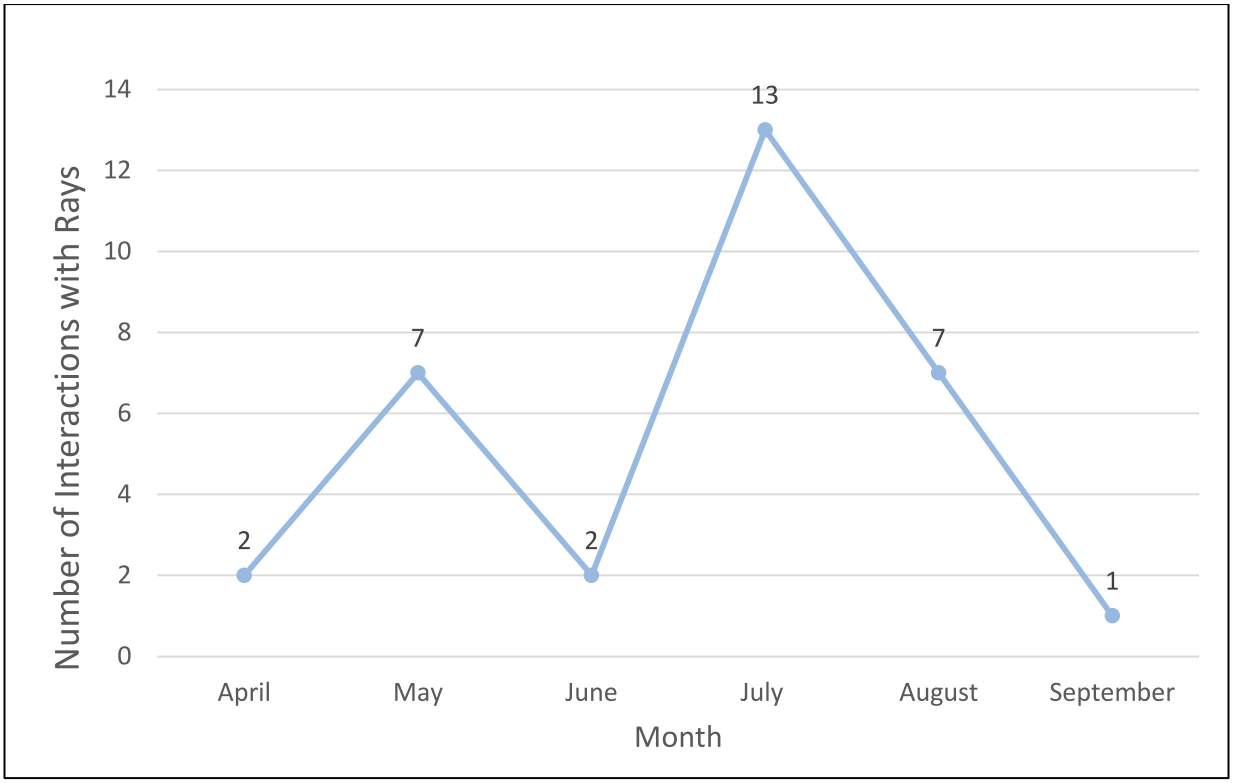

Figure 70. Giant manta ray and *Mobulidae* spp. interactions in the HI SSLL fishery by month, combined data from 2004 through 2018

Upon further analysis, we discovered a large portion of ray interactions can be attributed to a small number of vessels during the summer. In addition and as noted earlier, 39% of oceanic whitetip shark interactions also occurred during the month of July and in the Liliuokalani Ridge area where 67% of giant manta rays were captured.

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interaction levels (McCracken 2018) using Bayesian inference techniques as described in the *Approach to the Assessment* section of this biological opinion, and in greater detail in Appendix A (McCracken 2018). The predictions were based on the recorded bycatch of giant manta ray and the *Mobulidae* spp. classification in the HI SSLL fishery). Table 75 contains the resulting predictions of future annual, 2 year, and 3 year interactions (mean and 95 percentile) for giant manta ray and Table 76 contains the predictions for the *Mobulidae* spp classification in the HI SSLL fishery (from McCracken 2018).

Table 75. Mean and 95 percentile of giant manta ray posterior estimated exposure in terms of anticipated interactions with fishing gear (i.e., anticipated take level from McCracken 2018).

Number of years	Mean	95th percentile
1	3	8
2	5	13
3	8	17

Table 76. Mean and 95 percentile of *Mobulidae* spp. classification posterior estimated exposure in terms of anticipated interactions with fishing gear (i.e., anticipated take level from McCracken 2018).

Number of years	Mean	95th percentile
1	2	5
2	4	9
3	5	11

As noted previously, we expect that a portion of the animals assigned to the *Mobulidae* spp. classification are giant manta rays that could not be identified to species in the field. Using the Wilson score method, as discussed previously, we proportioned the 15 unidentified *Mobulidae* spp. (see Table 74) into a mean of 4 and an upper estimate of 5 (95% CI = 2, 5). The resulting future anticipated exposure estimates are listed in Table 77. For the remainder of this evaluation, we will focus on using the upper adjusted estimates of giant manta ray likely to be exposed to the HI SSLL in our jeopardy analysis.

Table 77. Anticipated future exposure levels for giant manta rays in the Hawaii shallow-set longline fishery. Values are derived by combining estimates from Table 74 and Table 75.

	Mean Estimates of giant manta rays exposures			Upper Estimates of giant manta ray exposures		
No. of Years	**Bayes estimate**	**Wilson score estimate**	**Adjusted mean estimate**	**Bayes estimate**	**Wilson score**	**Adjusted upper estimate**
1	3	4	7	8	5	13
2	5	4	9	13	5	18
3	8	4	12	17	5	22

6.2.3.4 Guadalupe Fur Seal

A total of four confirmed interactions with Guadalupe fur seals have occurred in the HI SSLL fishery. No observed interactions of pinnipeds were recorded prior to 2013, and between 2013 and 2018 there have been 12 interactions with animals in the suborder *Pinnipedia.* In addition to positive identification of Guadalupe fur seals, there have been two documented interactions with Northern elephant seals, and six unidentified pinnipeds.

Injury determinations are made in accordance with MMPA Mortality and Serious Injury determination (MSI) standards. Two interactions in the HI SSLL fishery resulted in non-serious injury determinations as all gear was removed from the animals and the individuals appeared healthy upon release (Carretta pers. comm. 2018). The other two interactions resulted in serious injuries (SI) in which mortality would be expected after release as substantial amounts of gear (more than a body length of line) were left on the individuals (Carretta pers. comm. 2018). The 2016 SAR is the most recently revised SAR for this species and only includes one interaction. NMFS expects information on the additional three interactions to be provided in the 2019 SAR update (Carretta pers. comm. 2018).

Guadalupe fur seal that are seriously injured are expected to have a high risk of latent mortality following release from the gear. Although the current at-vessel mortality rate is 0, and the overall mortality rate taking into account post release mortality for observed Guadalupe fur seals in the HI SSLL fishery is 50%. Caveats apply regarding a total mortality rate as each individual is assessed and provided a unique determination. These assessments are affected by various factors such as hook location, amount of gear left on the individual, handling, health at release, etc. For further information on MSI and SI determinations required by the MMPA, see Wade (1998) or the synopsis provided by Carretta et al. (2018). All six unidentified interactions of the suborder *Pinnipedia* resulted in SI determinations as large amounts of gear were left on the individuals (Carretta et al. 2018). The two interactions with Northern elephant seals resulted in one SI (2013) and one NSI determination (2014) (Bradford and Forney 2017).

A portion of the unidentified animals may be Guadalupe fur seals. We used the Wilson score method, as described earlier in this biological opinion, to estimate the proportion of unidentified *Pinnipedia* that may have been unidentified Guadalupe fur seals (67%, see Table 78 and Table 79). We added the resulting mean estimate of Guadalupe fur seals to the number of observed interactions the fishery had with Guadalupe fur seals to calculate the total number of interactions the fishery had with Guadalupe fur seals. In total, we estimate the HI SSLL fishery had 8 interactions with Guadalupe fur seals (4 observed plus 4 unidentified fur seals) between 2013 and 2018.

Table 78. Estimated proportions of the different species of *Pinnipedia* captured in the shallow-set long line fishery using observer data and the Wilson score method.

Species	**Original Data**			**Proportions**		**Confidence Interval for Proportions**	
	Number of Individuals Assigned to Species	**Number of Individuals Not Assigned to Species**	**Sample Size**	**p (Species Proportion)**	**1 – p (not Species)**	**Wilson LCI (Species)**	**Wilson UCI (Species)**
Guadalupe Fur Seal	4	2	6	0.67	0.33	0.3000	0.9032
Elephant Seal	2	4	6	0.33	0.67	0.0968	0.7000

Table 79. Mean estimate (with lower and upper confidence intervals) of the unknown Pinnipedia spp that were likely unrecorded Guadalupe fur seals based or Northern elephant seals (estimates generated from data in Table 78).

Unknown *Pinnipedia* spp. 6			
Species	**Mean Estimate**	**Lower 95% Confidence Interval**	**Upper 95% Confidence Interval**
Guadalupe fur seal	4	2	5
Northern Elephant seal	2	1	4

Demographic Patterns of Exposure

Four Guadalupe fur seals interactions have been confirmed in the HI SSLL fishery, and a total of 12 from the suborder *Pinnipedia*. Of those, no animals were boarded on any vessel. Per the data collection protocols, fishery observers use their best professional judgement to gauge the approximate length of the animal and record this data in their marine mammal log. Looking at all 10 records, the animals range approximately 3 to 5 ft (NMFS Observer Program unpublished data). Considering only the four confirmed records, three of the four were estimated at 4 ft in length, and the remaining seal was reported at approximately 3 ft in length.

It is unclear what age group these animals may have belonged to. We suspect that the animals may have been subadult or adult females or juvenile or sub-adult males. Females can reach up to approximately 5.4 ft and males upward of approximately 8 ft in length (Gallo-Reynoso and Figuerosa-Carranza 1996). The Guadalupe fur seal measured at 3 ft in length could be either a male or female based on the metrics provided by Gallo-Reynoso and Esperon-Rodriguez (2013), although it would be large enough to be considered weaned.

Spatial and Temporal Patterns of Exposure

Due to the low number of interactions, it is difficult to surmise strong spatial or temporal patterns in the interactions. However, of the four observed Guadalupe fur seal interactions, two occurred in November and two in the first week of December. In combining the unidentified pinnipeds with Guadalupe fur seal interactions, we note that all interactions during the same 3 month period across all years. One interaction occurred in the middle of October, two interactions in November, and seven interactions occurred in the month of December, six of which occurred in the first week of December. These interactions do not coincide with the two unusual mortality events which have been declared for this species off the U.S. West coast as previously discussed in the *Status of Listed Resources*. More interactions maybe occurring in this time frame due to a shift in fishing effort towards the Californian coast. All interactions between the HI SSLL fishery and all pinnipeds, including Guadalupe fur seals, have occurred off the West coast of California outside of the U.S EEZ. Nine interactions occur within 433 km (233 nmi) of each other with one outlier approximately 900 km (483 nmi) from the other interactions.

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interaction levels (aka “anticipated take levels”; McCracken 2018) using Bayesian statistical inference techniques as described in the *Approach to the Assessment* section of this biological opinion, and in greater detail in Appendix A (McCracken 2018). The predictions were based on the recorded bycatch of Guadalupe fur seals, unidentified pinniped and *Otariidae*, in the HI SSLL fishery. Table 80 contains the resulting predictions of future annual, 2 year, and 3 year interactions (mean and 95 percentile) for Guadalupe fur seals in the HI SSLL fishery (from McCracken 2018). Table 81 lists values determined for unidentified pinnipeds and Table 82 displays values for unidentified *Otairidae*.

Table 80. Mean and 95 percentile of Guadalupe fur seal posterior estimated exposure in terms of anticipated interactions with fishing gear (i.e., anticipated take level, from McCracken 2018).

Period of ATL	Mean	95 percentile
annual	2	6
2-year	4	9
3-year	5	12

Table 81. Mean and 95 percentile of unidentified pinniped posterior estimated exposure in terms of anticipated interactions with fishing gear (i.e., anticipated take level, from McCracken 2018).

Period of ATL	Mean	95th percentile
annual	2	5
2-year	3	8
3-year	4	10

Table 82. Mean and 95 percentile of unidentified Otariidae posterior estimated exposure in terms of anticipated interactions with fishing gear (i.e., anticipated take level, from McCracken 2018).

Period of ATL	Mean	95th percentile
annual	1	4
2-year	2	6
3-year	3	8

As noted previously, we expect that a portion of the animals assigned to the unidentified pinniped classifications are Guadalupe fur seals that could not be identified to species in the field. Using the Wilson score method, as discussed previously, we proportioned the 6 unidentified animals of the *Pinnipedia* suborder (see Table 79) into a mean of 4 (95% CI = 2, 5). The resulting future anticipated exposure estimates are listed in Table 83. For the remainder of this evaluation, we will focus our evaluation by using the upper adjusted estimates of Guadalupe fur seals likely to be exposed to the HI SSLL fishery in our jeopardy analysis.

Table 83. Anticipated future exposure levels for Guadalupe fur seals in the Hawaii shallow-set longline fishery. Values are derived by combining estimates from Table 79 and Table 80.

	Mean Estimates of Guadalupe fur seal exposures			**Upper Estimates of Guadalupe fur seal exposures**		
No. of Years	**Bayes estimate**	**Wilson score estimate**	**Adjusted mean estimate**	**Bayes estimate**	**Wilson score**	**Adjusted upper estimate**
1	2	4	6	6	5	11
2	4	4	8	9	5	14
3	5	4	9	12	5	17

6.3 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, 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 HI SSLL fishery presents to listed species results from hooking and entanglement by gear which can injure or kill sea turtles, Guadalupe fur seals, Oceanic whitetip sharks and giant manta rays. If hooked or entangled, air-breathing species can drown after being prevented from surfacing for air; alternatively, all listed species that are hooked or entangled, but do not immediately die from their wounds can suffer impaired swimming or foraging abilities, altered migratory behavior, and altered breeding or reproductive patterns, and latent mortality from their interactions.

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

6.3.1 Entanglement in Longline Gear

6.3.1.1 Sea Turtles

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

Sea turtles have been found entangled in branchlines, mainlines and float lines. Longline gear is fluid and can move according to oceanographic conditions determined by wind and waves, surface and subsurface currents, etc.; therefore, depending on both sea turtle behavior, environmental conditions, and location of the set, turtles could be entangled in longline gear. Entanglement in monofilament line or polypropylene (float line) could result in substantial wounds, including cuts, constriction, or bleeding on any body part. In addition entanglement could directly or indirectly interfere with mobility, causing impairment in feeding, breeding, or migration. Sea turtles entangled by longline gear are most often entangled around their neck and foreflippers, and, often in the case of leatherback entanglements, turtles have been found snarled in the mainline, floatline, and the branchline (e.g. Hoey 2000).

6.3.1.2 Oceanic Whitetip Shark

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

6.3.1.3 Giant Manta Ray

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

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

6.3.1.4 Guadalupe Fur Seal

We have very limited data on the incidental bycatch of Guadalupe fur seals in pelagic longlines, and even less data on entanglement in longline gear. However, according to marine debris data compiled in NOAA's 2014 Marine Debris Program Report, three Guadalupe fur seals were reported to have evidence of entanglement; one with net markings, another with a polyfilament line around its neck, and one with hook and line (Hanni et al. 1997). Once entangled, Guadalupe fur seals may drag and swim with gear attached for long distances, ultimately resulting in fatigue, compromised feeding ability, or severe injury, which may lead to reduced reproductive success and death.

6.3.2 Hooking

6.3.2.1 Sea Turtles

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

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

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

6.3.2.2 Oceanic Whitetip Shark

Sharks are incidentally captured when they attack baited hooks or depredate on catch. Sharks are a common bycatch in the HI SSLL fishery and according to Gilman et al. (2008) the use of lightsticks and setting hooks in shallow waters contribute to high shark catch rates. Injuries to sharks from longline hooks can be external-generally in the mouth, jaw, gills, roof of mouth, tail and fin or ingested internally, considered deeply-hooked or gut-hooked.

Circle hooks, which are required in the HI SSLL fishery, tend to become hooked in the mouth or jaw, as opposed to the gut or esophagus, and are intended to limit injury and be more easily removed (Cooke and Suski 2004). As with other marine species, even if the hook is removed, which is often possible with a lightly hooked shark, the hooking interaction is believed to be a significant event. As previously mentioned, capture on a longline is a stressful experience that can last an average 8-10 hours in the HI SSLL fishery, and longer. During capture, the amount of water flow over the gills is limited and biochemical recovery can take up to 2 to 7 days, and even longer for injured sharks (Campana et al. 2009). In addition, sharks are vulnerable to predation while being captured due to their restricted mobility, after their release due to exhaustion and injury. Furthermore, handling procedures can cause additional damage (e.g. cutting the jaw, tail, gaffing, etc.), stress, or death.

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

6.3.2.3 Giant Manta Ray

The giant manta ray primarily feeds on planktonic organisms such as euphausiids, copepods, mysids, decapod larvae and shrimp, but some studies have noted their consumption of small and moderate sized fishes (Bertolini 1933; Bigelow and Schroeder 1953; Carpenter and Niem 2001; The Hawaii Association for Marine Education and Research Inc. 2005). Although there have been no studies specifically on the association of giant manta ray to lightsticks, we do know that planktonic marine life are attracted to light; therefore, it seems plausible that foraging mantas may also be drawn to the lightsticks and therefore at risk of becoming hooked. Due to its foraging behavior the giant manta ray tends to be more vulnerable to foul hooking as opposed to being hooked in the mouth (Mas et al. 2010).

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

6.3.2.4 Guadalupe Fur Seal

Although the preferred prey species of Guadalupe fur seals are vertically migrating squid, pelagic fish also constitute a small percentage of their diet (Gallo-Reynoso and Esperon-Rodrguez 2013). According to Gallo-Reynoso et al. (2008) the Guadalupe fur seal's foraging strategy tends to be limited to shallow dives in the range of 0-200 m, which start at sunset and end at dawn. This range overlaps with the depth and time longline gear is set in the HI SSLL fishery which targets an optimum fishing depth of 100 m and sets gear at night and begins to haul gear at sunrise. Therefore, the risk of incidental hooking of Guadalupe fur seals is twofold; a seal could concurrently be foraging in or migrating through an area where longline gear is set and soaking, or actively feeding on mackerel-type bait. To date, the Guadalupe fur seals and unidentified pinnipeds that have interacted with HI SSLL fishery have been hooked externally, in the flippers, head, or mouth. Even if the hook is removed, which is often possible with a lightly hooked seal, the hooking interaction is believed to be a significant event. In general, capture on a longline is a stressful experience that can last on average 8-10 hours in the HI SSLL fishery, and even longer if gear breaks or there is a mechanical issue on the vessel. The mainline is shallow enough that the seal would be able to surface for air, however, depending how long it is hooked and how hard it fights to break free, the seal is more vulnerable to predation and can be exhausted upon release requiring extended time for behavioral recovery and tissue repair.

If a Guadalupe fur seal should ingest a hook, it would be at risk of perforation of the esophagus, stomach, or intestines which can cause serious infection which may result in the death of the animal. Similar to sea turtles, when a hook is ingested, the process of movement, either by the seal's attempt to get free of the hook or by being hauled in by the vessel, can traumatize the internal organs of the seal either by piercing the esophagus, stomach, or other organs, or by pulling the organs from their connective tissue. Alternatively, as documented in the Hawaiian monk seal and the harbor seal, ingested hooks have been found in the stomachs of these animals, presumably without piercing an organ (Osinga and Hart 2006; NOAA 2018).

6.3.3 Trailing Gear (Line)

6.3.3.1 Sea Turtles

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

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

6.3.3.2 Oceanic Whitetip Shark

Members of the Western and Central Pacific Fisheries Commission are required to regulate their vessels consistent with the conservation and management measures for the oceanic whitetip shark. Pursuant to Conservation and Management Measure (CMM 2011-04), NMFS has implemented regulations (50 CFR 300.226) requiring vessels to release any oceanic whitetip shark that is caught as soon as possible after the shark is brought alongside the vessel, and to do so in a manner that results in as little harm to the shark as possible. In accordance to this measure, the amount of trailing gear shall be minimal as to cause as little harm as possible. Excessive trailing gear could directly or indirectly interfere with the shark's mobility, causing impairment in feeding, breeding, or migration. Further, trailing line can also become snagged on a floating or fixed object, further entangling the shark or the drag from the float can cause the line to constrict around the body of the shark or its fins.

6.3.3.3 Giant Manta Ray

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

6.3.3.4 Guadalupe Fur Seal

Similar to sea turtles, trailing line has the potential to be ingested by Guadalupe fur seals, which may prevent or hamper the seal when it feeds. As a result, trailing line can eventually kill a seal shortly after it is released or it may take a while for the seal to die. Alternatively, trailing line can become a constricting wrap on the animal, become snagged on something in the environment, anchoring the animal or accumulate drag. Crew that have hooked a seal are directed to remove the hook if it is external. However if the hooking is internal and would cause more damage to remove, crew are instructed to cut the line as close to the hook as possible in order to minimize the amount of trailing gear. Occasionally, the branchline breaks during the interaction and a majority of the branchline may remain attached to the animal.

6.3.4 Post Interaction Survival

6.3.4.1 Sea Turtles

In order to assess the post-hooking survival and mortality of turtles that came up hooked or entangled and were released alive we used the NMFS criteria (Ryder 2006) that was described briefly in the exposure section above. The criteria was 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 was last assessed by experts in 2011 (NMFS 2012) and no changes were made to the mortality categories which are provided in at the beginning of the Exposure section of this biological opinion.

Since it has been several years since the criteria has been reviewed we conducted a systematic search for new literature that may warrant updating the mortality rates in Ryder et al. (2006). Our general search parameters are discussed in the *Approach to the Assessment* section of this biological opinion. We found several studies that yielded mortality rates for sea turtles and they are summarized in Table 84.

Table 84. Estimates of sea turtle mortality from the literature. (Adapted from Swimmer et al. 2013).

Author year	Mortality rate	Comments	Methods	Sample size
Aguilar et al. 1995	20-30%	All deep hooked, studied in captivity	captivity	38
Casale et al. 2008	65-82%		captivity	-
Hays et al. 2005	31%		PTT	-
Parker et al. 2005	20-40%	Depends on hooks status deep vs light	PTT	-
Chaloupka et al. 2004	8-34%	Shallow-hook vs deep. (within one week)	PTT	40
Swimmer et al. 2006	0%	Low rates in shallow gear	PSAT	-
Sasso and Epperly, 2007	19%	Depends on how much gear is removed	PSAT	-
Quevedo et al. 2013	31-38%	Found no difference between light and deep hooking.	PSAT	26
Swimmer et al. 2013	7-55%	28% overall 55% deep hooking 7% shallow hooking	PSAT	29

Most of these studies were used in the development of the 2006 criteria or in the evaluation done in 2011 (Ryder et al. 2006, Swimmer and Gilman 2012). Much of the literature in the last ten years has focused on studies that compare circle hooks to J-hooks and the difference between light hooking vs deep for turtles, or the impacts to target species and other bycatch. Based on our review of the literature we 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 HI SSLL fishery. The criteria establishes 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.

As previously described in the *Exposure* section of this biological opinion, we assigned each turtle caught in the HI SSLL fishery to one of six injury categories based on the description of the hook location and the amount of gear remaining when the animal was released (see for instance Table 58). We applied the mortality coefficient from Ryder et al. (2006) for each injury category to the total number of turtles in the injury category to estimate the cumulative number of latent sea turtle mortalities for the fishery. We also calculated the fishery mortality rate of each sea turtle species that interacted with the HI SSLL fishery between 2004 and 2018, and we calculated the confidence interval on the proportion without a correction for continuity as described by Newcombe (1998). We applied the same injury category mortality coefficients to the anticipated number of animals exposed to the action (using the mean and 95 percentile of the 1 year, 2 year, and 3 year predictions) to estimate future mortality of each species (see also Appendix XX McCracken 2018).

Table 85. Criteria for assessing marine turtle post-interaction mortality after release from longline gear. Percentages are shown for hardshell turtles (i.e., loggerhead, Kemp's ridley, olive ridley, hawksbill, and green turtle), followed by percentages for leatherbacks (in parentheses; Table from Ryder et al. 2006).

Injury Category	Release Condition			
	Hook and with trailing line ≥ to half the length of the carapace (line is trailing, turtle is not entangled)	Hook and trailing line < half the length of the carapace (line is trailing, turtle is not entangled)	Hook and entangled (line is not trailing, turtle is entangled[1])	All gear removed
	Hardshell (Leatherback)	Hardshell (Leatherback)	Hardshell (Leatherback)	Hardshell (Leatherback)
I. Hooked externally w/wo entanglement	20 (30)	10 (15)	55 (65)	5 (10)
II. Hooked in upper or lower jaw w/wo entanglement. Includes ramphotheca, but not any other jaw/mouth tissue parts (see Category III).	30 (40)	20 (30)	65 (75)	10 (15)
III. Hooked in cervical esophagus, glottis, jaw joint, soft palate, tongue, and/or other jaw/mouth tissue parts not categorized elsewhere, w/wo entanglement. Includes all events where the **insertion point of the hook is visible** when viewed through the mouth.	45 (55)	35 (45)	75 (85)	25 (35)
IV. Hooked in esophagus at or below level of the heart w/wo entanglement. Includes all events where the insertion point of the **hook is not visible** when viewed through the mouth.	60 (70)	50 (60)	85 (95)	n/a[2]
V. Entangled only, no hook involved.	Released Entangled 50 (60)			Fully disentangled 1 (2)
VI. Comatose/resuscitated	n/a[3]	70 (80)	n/a[3]	60 (70)

[1] Length of line is not relevant as turtle remains entangled at release.

[2] Per veterinary recommendation hooks would not be removed if the insertion point of the hook is not visible when viewed through the open mouth.

[3] Assumes that a resuscitated turtle will always have the line cut to a length less than half the length of the carapace, even if the hook remains.

Leatherback sea turtle

Between 2004 and 2018 there were a total of 105 interactions with leatherback sea turtles in the HI SSLL fishery (see Table 54 and Table 55). Using the mortality coefficients of Ryder et al. (2006), these 105 interactions likely resulted in the death of 21 leatherback sea turtles over 14 years in the HI SSLL fishery (Table 86). The corresponding mortality rate for leatherback sea turtle interactions in this fishery is 20% (95% CI: 0.135, 0.287).

Table 86. Cumulative number of leatherback sea turtle mortalities by injury category for the Hawaii shallow-set longline fishery, 2004 through 2018. Numbers are based on the total number of interactions and injury mortality rates of Ryder et al. 2006. We calculated the confidence interval on the proportion without a correction for continuity as described by Newcombe (1998).

	Number of leatherback sea turtle mortalities			
Injury Category	**Line < 1/2 SCL**	**Line ≥ 1/2 SCL**	**No Gear**	**Grand Total**
Entangled	-	-	0	0
External	4	7	4	15
Insertion Not Visible	-	1	-	1
Insertion Visible	2	2	0	4
Jaw	1	0	0	1
Grand Total	7	10	4	21
Fishery mortality rate (95% CI)				0.20 (0.14, 0.29)

We applied the same injury category mortality coefficients to the anticipated exposure numbers for leatherback sea turtles (see Table 56). We calculated the mortality for the mean and 95 percentile for the 1 year, 2 year, and 3 year predictions for leatherback sea turtles (Table 87). The predicted level of interactions with leatherback sea turtles is a mean of 10 each year with an upper estimate of 21 leatherback sea turtles each year. Our model accounts for annual variability and consequently predicts that over a three-year time period the mean interaction rate is 30 leatherback sea turtles with an upper estimate of 48 interactions with leatherback sea turtles (see Table 56). As a result, the upper estimated number of leatherback sea turtles that would likely die from their interactions in this fishery would be three per year, or 9 over a 3 year period (Table 87).

Table 87. Anticipated mortalities of leatherback sea turtles based on predicted exposure values for the mean and 95 percentile for 1-3 years (see Table 56)

Period of ATL	Mortality estimate for the mean	Upper mortality estimate
annual	1	3
2	3	6
3	5	9

Loggerhead sea turtle

Between 2004 and 2018 there were a total of 177 interactions with loggerhead sea turtles in the HI SSLL fishery (see Table 57). We calculated loggerhead sea turtle mortality resulting from injuries sustained in the fishery using the injury mortality coefficients of Ryder et al. (2006). We combined the estimated mortalities from injuries with the at-vessel (immediate) mortalities to derive the total estimated number of loggerhead mortalities in the HI SSLL fishery between 2004 and 2018. In the past 14 years, an estimated 26 loggerhead sea turtles died from injuries sustained during interactions plus 2 died immediately, for a combined total of 28 loggerhead sea turtle mortalities out of 177 (Table 88). The corresponding mortality rate for loggerhead sea turtle interactions in this fishery is 16% (95% CI: 0.11, 0.22).

Table 88. Cumulative number of loggerhead sea turtle mortalities by injury category for the Hawaii shallow-set longline fishery, 2004 through 2018. Numbers are based on the total number of interactions and injury mortality rates of Ryder et al. (2006) and at-sea (immediate) deaths of loggerhead sea turtles. We calculated the confidence interval on the proportion without a correction for continuity as described by Newcombe (1998).

	Number of loggerhead sea turtle mortalities			
Injury Category	**Line < 1/2 SCL**	**Line ≥ 1/2 SCL**	**No Gear**	**Grand Total**
Entangled	-	1	0	1
External	0	-	4	4
Insertion Not Visible	6	2	-	8
Insertion Visible	2	1	6	9
Jaw	0	-	3	3
Subtotal	8	4	13	26
At vessel mortality		0		2
Grand Total				28
Fishery mortality rate (95% CI)				0.16 (0.11, 0.22)

We applied the same injury category mortality coefficients to the anticipated exposure numbers for loggerhead sea turtles (see Table 61) and added the at-vessel mortality coefficient, which is unique to loggerhead sea turtles in this fishery to derive the future mortality estimates. Table 89 provides the mortality estimates for loggerhead sea turtles for the mean and 95 percentile for 1, 2, and 3 years. The predicted level of interactions with loggerhead sea turtles is a mean of 16 each year with an upper estimate of 36 leatherback sea turtles each year. Our model accounts for annual variability and consequently predicts that over a three-year time period the mean interaction rate is 47 loggerhead sea turtles with an upper estimate of 81 interactions with loggerhead sea turtles (see Table 61). As a result, the upper estimated number of loggerhead sea turtles that would likely die from their interactions in this fishery would be 6 per year, or 13 over a 3-year period.

Table 89. Anticipated mortalities of loggerhead sea turtles based on predicted exposure values for the mean and 95 percentile for 1-3 years and the unique loggerhead sea turtle mortality coefficient for at-vessel mortality.

Period of ATL	Mortality estimate for the mean	Upper mortality estimate
annual	2	6
2	4	10
3	9	13

Olive ridley sea turtle

Between 2004 and 2018 there were a total of 10 interactions with olive ridley sea turtles in the HI SSLL fishery (see Table 62). We calculated olive ridley sea turtle mortality using the injury mortality coefficients of Ryder et al. (2006). In the past 14 years, there were likely no olive ridley sea turtles that died from injuries sustained during interactions out of 10 interactions (Table 90). The confidence intervals for a mortality rate of zero olive ridley sea turtles in this fishery is 95% CI: 0.0, 0.28.

Table 90. Cumulative number of olive ridley sea turtle mortalities by injury category for the Hawaii shallow-set longline fishery, 2004 through 2018. Numbers are based on the total number of interactions and injury mortality rates of Ryder et al. (2006). We calculated the confidence interval on the proportion without a correction for continuity as described by Newcombe (1998).

	Number of olive ridley sea turtle mortalities			
Injury Category	**Line < 1/2 SCL**	**Line ≥ 1/2 SCL**	**No Gear**	**Grand Total**
Entangled	-	0	0	0
External	0	-	0	0
Insertion Not Visible	0	0	-	0
Insertion Visible	0	0	0	0
Jaw	0	-	0	0
Grand Total	0	0	0	0
Fishery mortality rate (95% CI)				0.00 (0.0, 0.28)

We applied the same injury category mortality coefficients to the anticipated exposure numbers for olive ridley sea turtles to derive the future mortality estimates. Table 91 provides the mortality estimates for olive ridley sea turtles for the mean and 95 percentile for 1, 2, and 3 years. The predicted level of interactions with olive ridley sea turtles is a mean of 1 each year with an upper estimate of four olive ridley sea turtles each year (see Table 63). The three-year time estimate of interactions is a mean of 4 and an upper estimate of 11 interactions with olive ridley sea turtles. As a result, the upper estimated number of olive ridley sea turtles that would likely die from their interactions in this fishery would be zero over a three-year period (Table 91). Based on the distribution of hooking injuries recorded for this species, and the mortality coefficients of Ryder et al. (2006), an estimated 15 interactions would have to occur before we expected to see a mortality. However, our sample size of olive ridley sea turtles is very small and small changes in the number and type of injury (i.e., hook location and depth) recorded by observers with subsequent interactions could significantly affect the predicted number of mortalities.

Table 91. Anticipated mortalities of olive ridley sea turtles based on predicted exposure values for the mean and 95 percentile for 1-3 years.

Period of ATL	**Mortality estimate for the mean**	**Upper mortality estimate**
annual	0	0
2	0	0
3	0	0

Green sea turtle

Between 2004 and 2018 there were a total of 10 interactions with green sea turtles in the HI SSLL fishery (see Table 64). We calculated green sea turtle mortality using the injury mortality coefficients of Ryder et al. (2006). In the past 14 years, there were likely no green sea turtles that died from injuries sustained during interactions out of 10 interactions (Table 92). The confidence intervals for a mortality rate of zero green sea turtles in this fishery is 95% CI: 0.0, 0.28.

Table 92. Cumulative number of green sea turtle mortalities by injury category for the Hawaii shallow-set longline fishery, 2004 through 2018. Numbers are based on the total number of interactions and injury mortality rates of Ryder et al. (2006). We calculated the confidence interval on the proportion without a correction for continuity as described by Newcombe (1998).

	Number of green sea turtle mortalities			
Injury Category	**Line < 1/2 SCL**	**Line ≥ 1/2 SCL**	**No Gear**	**Grand Total**
Entangled	-	0	0	0
External	0	-	0	0
Insertion Not Visible	0	0	-	0
Insertion Visible	0	0	0	0
Jaw	0	-	0	0
Grand Total	0	0	0	0
Fishery mortality rate (95% CI)			0.0 (0.0, 0.28)	

We applied the same injury category mortality coefficients to the anticipated exposure numbers for green sea turtles to derive the future mortality estimates. Table 93 provides the mortality estimates for green sea turtles for the mean and 95 percentile for 1, 2, and 3 years. The predicted level of interactions with green sea turtles is a mean of 1 each year with an upper estimate of 5 green sea turtles each year (see Table 66). The three-year time estimate of interactions is a mean of 4 and an upper estimate of 10 interactions with green sea turtles (see Table 66). As a result, the upper estimated number of green sea turtles that would likely die from their interactions in this fishery would be 1 over a 3 year period (Table 93). Like the olive ridley sea turtle, the small number of interactions, and the corresponding small number of injury categories recorded by observers would be expected to have an important influence on our the mortality prediction. Our sample size of green sea turtles is very small and small changes in the number and type of injury (i.e., hook location and depth) recorded by observers with subsequent interactions could significantly affect the predicted number of mortalities. The influence of a turtle or two in different injury categories is apparent in the injury category distribution seen between olive ridley and green sea turtles (see Table 52), and is expressed in predicted mortality estimates (we expect no olive ridley sea turtle mortalities from injuries whereas 1 green sea turtle mortality is expected).

Table 93. Anticipated mortalities of green sea turtles based on predicted exposure values for the mean and 95 percentile for 1-3 years.

Period of ATL	Mortality estimate for the mean	Upper mortality estimate
annual	0	0
2	0	0
3	0	1

Oceanic Whitetip Shark

Between 2004 and 2018 there were a total of 875 interactions with oceanic whitetip sharks in the HI SSLL fishery (Table 67). We calculated overall mortality rates using a known at-vessel mortality rate based on NMFS observer data from 2004 to 2018 in the HI SSLL fishery; with a low and high range of post release mortality metrics determined by a systematic review of the literature.

At this time, metrics related to post release mortality has not been determined for oceanic whitetip sharks in any fishery. This was further substantiated by Musyl et al. (2011), the only stock assessment completed for the species to date (Rice and Harley 2013), and the recent status review conducted in 2016 (Young et al. 2016). At-vessel mortality in the HI SSLL fishery based on all interactions (875) between 2004 and 2018, results in a rate of 11.54% (95% CI: 0.095; 0.13). Reviewing the scientific literature for surrogate species and the fisheries that best exemplify the same operational methods that are practiced in the HI SSLL fishery; post release mortality ranges from 6.3 to 19%, resulting in overall mortality estimates between 23 and 59% for those particular species (i.e. shortfin mako, blue shark, porbeagle; Musyl et al. (2006) also assessed oceanic whitetip, bigeye thresher, and silky sharks but did not determine post release mortality values for these species).

However, caution should be taken in the assessment of the data as post release survival rates are species-specific with much of the research focusing on underlying physiology pertaining to how sharks handle stress, these studies include species in other locations of the world, and some species may not be physiologically similar in resilience. Even so, they provide the best comparable representation in the literature according to how the HI SSLL fishery operates in the action area. The only stock assessment completed for the species in the Western Pacific predicted an estimated fishing mortality of approximately 30% for non-target longline operations (Rice and Harley 2013; WCPFC 2018). This stock assessment may not be a good surrogate or comparision because it does not quantify total mortality rates, is based on three estimated fixed mortality rates (0.05, 0.1, and 0.2), and the model only extends to 2009. Therefore, a cautionary approach was taken using the literature for our analysis using the known at-vessel mortality rate. The literature also suggests that hooking location is a predictor of fate. Sharks that swallow hooks, compared to those that are hooked in the jaw, have an increased likelihood of dying (Campana et al. 2009).

Therefore, using our at-vessel mortality rate of 11.54% and the best scientific data available for post release mortality using the above range, we calculated two mortality rates for oceanic whitetip sharks based on the range we found in the literature (18 and 31%). We then applied these rates to the anticipated exposure level at the 95th percentile value (Table 71) for anticipated exposures as presented in Table 94.

Table 94. Number of expected oceanic whitetip shark mortalities based on the 95th percentile estimate of anticipated future exposures in the HI SSLL fishery (see also Table 71) , and the lower and upper estimated post release mortality values from the literature.

Number of years	Upper exposure estimate	Lower estimate of mortality (95% CI)	Upper estimate of mortality (95% CI)
1	102	19 (0.12, 0.27)	32 (0.23, 0.40)
2	166	30 (0.12, 0.25)	51 (0.24, 0.38)
3	299	54 (0.14, 0.23)	92 (0.26, 0.36)

Thus mortality ranges exist for the projected number of takes for up to the projected three years. For instance, for the first year, the 95th percentile projects 102 interactions (exposures) will occur. Applying the 18% overall mortality rate results in 19 individuals, and the 31% mortality rate results in 32 individuals. Therefore at these rates for the first year, we would expect a total mortality of 19 to 32 of the 102 sharks, or approximately one out of every three to five sharks, to die from interactions with the HI SSLL fishery.

Giant Manta Ray

An extensive review of the literature for post-release survivorship for *Mobulidae* spp. has determined that there are no studies specific to longline fisheries that assess the effect of remaining gear on manta and mobulid species (Mas et al. 2015). In our search we did find a study by Mas et al. (2015) that analyzed fishery observer data on at-vessel mortality of mobulid rays captured in the pelagic longline fishery in the south-western Atlantic. While this study did not assess post-hooking mortality, we found some similarities between the HI SSLL fishery and the Uruguayan pelagic longline fishery; both fisheries target swordfish and set gear in the epipelagic zone generally at depths no greater than 100 m. The percentage of at-vessel mortality rates were low for the mobulids incidentally captured in this study. Similarly, in the HI SSLL fishery, a high proportion of giant manta rays that are hooked are released alive. Coelho et al. (2011) also found similar low at-vessel mortalities for mobulids in the Portuguese longline fishery in the Atlantic and Indian Oceans. Likewise, Beerkircher et al. (2008) found 97% of mobulid and dasyatids (stingrays) captured in the US pelagic longline fishery were released alive.

However, as the name implies, the giant manta ray is large and seldom boarded. Therefore, the animals hooked on the longline may be released alive but are released with the hook and some trailing gear attached. Further, Mas et al. (2015) found that in attempt to remove some of the gear from the mobulids, fishers used pike poles (long handled poles with hooks) to hold the animal in place, sometimes causing severe injuries that may decrease post-release survival. Handling procedures can cause additional damage, stress, or death. Consequently, incorporating safe-release guidelines is essential to increasing post-hooking survival rates (Hutchinson et al. 2017).

Mas et al. (2015) noted that although the observers in their study seldom recorded where the animal was hooked, no mobulids were hooked in the mouth. Similar to the leatherback sea turtle, they tend to be foul hooked externally or entangled in the branchline (Sales et al. 2010; Domingo et al. 2012). Comparable to leatherback sea turtles, if the giant manta ray becomes entangled in trailing gear, it is at risk of severe injuries that can lead to a reduction in feeding efficiency and even death. Trailing line can become snagged on a floating or fixed object, further entangling the giant manta ray, or the drag from the float can cause the line to constrict around a manta's cephalic fin until the line cuts through the appendage.

At this time, the HI SSLL fishery observers do not record the hooking location when giant manta rays are captured. However, the literature suggests there is a similarity between leatherback sea turtles and mobulids in regards to the hooking location. Since neither of these species are actively predating on the bait on longline gear, but attracted to marine life that collects on buoys and buoy lines at or near the surface, or some combination of these and/or other reasons, they tend to be foul hooked in the flipper and pectoral fin area as opposed to the mouth or in the esophagus. Moreover, both of these species are large in size; the giant manta ray is considered to be one of the largest fishes in the world where as the leatherback sea turtle is the largest marine turtle and of the largest reptiles in the world. Further, both animals propel through the ocean environment using similar movements; the leatherback sea turtle employs its long front flippers while the giant manta ray relies on its large pectoral fins as wings to "fly" through the water. Due to these similarities, and the absence of species (or genus) specific mortality rates, we used the leatherback sea turtle as a proxy to assess post-release mortality and considered the Ryder et al. (2006) injury criteria for leatherback sea turtles as a possible proxy for determining post-hooking survival of giant manta rays (see Table 58).

Since giant manta ray do not have a carapace, we used the length of the animal when assessing how much trailing gear remained. Additionally, based on anecdotal data from the PIRO Observer Program, giant manta rays are not dehooked by the observer or the crew. The crew tends to cut the line and in some cases the line breaks. Therefore, we did not include the "No Gear" column in our giant manta ray comparison.

Between 2004 and 2018 there were a total of 21 interactions with giant manta rays (17 observed plus 4 (unidentified) giant manta rays) in the HI SSLL fishery (see Table 77). The corresponding at vessel mortality rate for giant manta ray interactions in this fishery is 18% (95% CI: 0.07, 0.39). Since four giant manta rays were dead upon retrieval, we added the at-vessel mortality coefficient to derive the future mortality estimates. Using the leatherback sea turtle mortality coefficients (Ryder et al. 2006), these 21 interactions likely resulted in the death of nine giant manta ray over 14 years in the HI SSLL fishery (Table 95).

Table 95. Cumulative number of giant manta ray mortalities by injury category for the Hawaii shallow-set longline fishery, 2004 through 2018. Numbers are based on the total number of interactions and injury mortality rates of Ryder et al. (2006). We calculated the confidence interval on the proportion without a correction for continuity as described by Newcombe (1998).

Injury Category	**Line < 1/2 SCL**	**Line ≥ 1/2 SCL**	**Grand Total**
External	1	2	3
Insertion Not Visible	-	0	0
Insertion Visible	1	1	2
Jaw	0	0	0
Sub Total	2	**3**	**5**
Special Hazard Rate	0	-	4
Total mortalities	-	**-**	**9**
Fishery mortality rate (95% CI)			.41 (0.23, 0.61)

We used the same mortality rates to estimate the total number of future mortalities from anticipated adusted upper exposure estimate for the annual, 2 year, and three year estimates which are (13,18, and 22 respectively,) for giant manta rays (Table 77). The upper estimated number of giant manta rays that would likely die from their interactions in this fishery would be four in one year or 9 over a 3 year period.

Table 96. Anticipated mortalities of giant manta rays based on predicted exposure values for the mean and adjusted upper estimate for 1-3 years (see Table 77).

Period of ATL	**Mortality estimate for the mean**	**Adjusted upper estimate**
annual	2	4
2	4	7
3	4	9

Guadalupe Fur Seal

Regarding Guadalupe fur seals, four interactions to date resulted in 50% SI and NSI determinations respectively (16 U.S.C. §§ 1361 et seq.; Carretta pers. comm. 2018). Those with a SI determinations would be expected to die following release from the vessel (later in time). Therefore, the current at-vessel mortality rate is zero, and the overall mortality rate taking into account post release mortality for Guadalupe fur seals in the HI SSLL fishery is 50%. Caveats apply as each determination is affected by various factors such as hook location, amount of gear left on the individual, handling, health at release, etc. For further information on NSI and SI determinations required by the MMPA, please see Wade (1998) or the synopsis provided by Carretta et al. (2018).

Additionally, three unidentified otariids (1 in 2014, and 2 in 2015) and three unidentified pinnipeds (all in 2015) have interacted with the fishery (Carretta et al. 2018). All six unidentified interactions resulted in SI determinations as large amounts of gear were left on the individuals (Carretta et al. 2018). When including all interactions of Guadalupe fur seals and unidentified otariids and pinnipeds, a total of 10 interactions have occurred with an 80% mortality rate.

Therefore, to quantify the number of expected mortalities, we projected the upper exposure estimate using the Wilson score method. This method allows for the appropriation of unidentified animals to the listed species and accounts for all individuals we expect to interact with the fishery. We then applied the 80% mortality rate to these values because this is the estimated mortality for pinniped interactions in this fishery, and while it is not specific to only the Guadalupe fur seal, it represents the best estimate we have given the small number of interactions to date. Using the mortality rate of 80%, we recognize this is likely the maximum number of individual Guadalupe fur seals that would die from interactions with the HI SSLL fishery (Table 97).

Table 97. Number of expected mortalities based on the adjusted upper exposure estimate using the Wilsons score method for Guadalupe fur seals in the Hawaii shallow-set longline fishery, with an 80% mortality estimate.

Number of years	Adjusted Upper Exposure Estimate	Upper Mortality Estimate – based on rate of 80% (95% CI)
1	11	9 (0.52, 0.95)
2	14	12 (0.60, 0.96)
3	17	14 (0.59, 0.94)

Thus mortality ranges exist for the projected number of takes for up to the projected three years. For instance for one year, the 95th percentile projects upwards of 11 takes will occur for Guadalupe fur seals. Applying the 80% overall mortality rate results in 9 individuals. We would therefore expect 9 of the 11 total projected Guadalupe fur seal interactions in the first year to perish from interactions with the fishery.

6.4 Indirect Effects

NMFS' 2018 BE notes that "there is no information to conclude that there would be indirect effects from the operation of the fishery that could affect Guadalupe fur seals, oceanic whitetip sharks, and giant manta rays." We concur with this conclusion. The BE also notes a 2012 technical memorandum by Chan and Pan (2012) that describes indirect beneficial effects from the HI SSLL fishery on sea turtles, but does not describe how these beneficial indirect effects influence the total number of predicted interactions, or the survival and recovery, of loggerhead sea turtle, leatherback sea turtle, green sea turtle, or olive ridley sea turtle populations considered herein. The basic premise of the argument is that the HI SSLL fishery causes beneficial indirect effects on sea turtles is that this domestic fishery takes (interacts with and kills) fewer turtles than foreign fleets. In essence, the argument supposes "but for" the HI SSLL fishery the overall (global, or at least Pacific wide), take of sea turtles would be higher than they otherwise are when the HI SSLL fishery is operating because there would be more turtle interactions from a "market transfer" effect when foreign fisheries increase production of swordfish to meet market demands that were voided by the closure of the HI SSLL fishery. Since foreign fisheries do not abide by the same turtle-friendly protective measures as the HI SSLL fishery, interactions with sea turtles would be expected to increase.

We conducted a systematic review to critically appraise whether market transfer (spillover) effects resulting from the HI SSLL fishery meets the definition of indirect effects — "those that are *caused by the proposed action* and are later in time, but still are *reasonably certain to occur* (50 CFR 402.02)." The objective was to systematically review and summarize the best available scientific and commercial information on the occurrence and persistence of the indirect effects of market transfer effects from the HI SSLL fishery on threatened and endangered species. We conducted detailed evaluations of several sources of data or descriptive information and found that there are data or studies that suggest a market transfer effect is possible (Bartram 2010; Chan and Pan 2012, 2016; Helvey et al. 2017; Komoroske and Lewison 2015: Mukherjee 2015; Rausser and Kovach, unpublished 2018; Rausser et al. 2009; Sarmiento 2006; Scorse et al. 2018) . However, the available data focus on the occurrence of a market transfer effect from the 2001 – 2004 closure of the HI SSLL fishery, but does not establish that such an effect persists today. Most available studies relied on changes in import data to reach a conclusion that a market transfer effect occurred during the HI SSLL fishery closure period. However, those studies are not sufficient to "indicate an increase [in (sic)] sea turtle bycatch if there is no increase in corresponding swordfish production by foreign vessels having a higher sea turtle bycatch rate (Chan and Pan 2016)." Chan and Pan (2016) noted that earlier studies, including their own, "*did not* demonstrate whether the changes in US imports due to the regulations of the Hawaii longline fishery *caused* foreign production to increase." Several authors developed econometric and other models to evaluate the market transfer effect in the HI SSLL fishery, however, most relied on assumptions that cannot be verified empirically, which weakens their assertion that a market transfer effect occurred and, another set of assumptions might drastically change the outcome of those models. Nevertheless, we found no evidence to suggest that a market transfer effect occurs today.

As noted by Pan and Chan (2016), among other things, one condition that must be met for a spillover to occur is that sea turtle bycatch rates of the foreign longline fisheries must be higher than the sea turtle bycatch rate of the HI SSLL fishery, both of which target swordfish. It appears that the management region for non-domestic fisheries has changed, such that this may no longer be true. In 2004, the Convention on the Conservation and Management of Highly Migratory Fish Stocks in the Western and Central Pacific Ocean went into effect. Most of the non-domestic fleets considered as having weaker or no regulations either ratified or acceded to this Convention, which established the Western and Central Pacific Fisheries Commission (WCPFC).

Since 2008, the WCPFC has adopted conservation and management measures (CMM 2008-03, 2018-04) that requires, among other measures, that members requires longline vessels fishing for swordfish in the shallow set manner to employ large circle hooks, use only whole finfish bait, and use additional measures to conserve sea turtles (see FAO 2010 for additional description of the 2008 measure). Recent improvements in foreign fishing sea turtle mitigation measures call into question core assumptions that reductions in effort in the U.S. longline fisheries result in higher interactions of sea turtles in foreign fisheries, such that effects are “transferred” to fleets controlled by weaker or no protective regulations.

The data WCPFC collect appear to support this. Specifically, the WCPFC collects data on bycatch rates and mortality rates of turtles, sharks, and seabirds. NMFS retrieved the WCPFC bycatch data that are publicly available (for the years 2013 to 2017). These data do not support the central assumptions made by Rausser et al. (2009) and the other authors who concluded that non-domestic longline fleets catch and kill turtles at substantially higher rates than domestic fleets.

We compared interaction rates in the HI SSLL fishery and the WCPFC shallow-set longline fisheries of north of 10° N between 2013 and 2017, and found that there does not appear to be a large difference between interaction rates may have been the case during the time of the 2001 - 2004 closure. We compared interaction rates between the HI SSLL fishery and WCPFC countries and found no appreciable difference in loggerhead sea turtle interaction rates (see Table 98 median estimates for 2016 and 2017). Similarly, we found no appreciable difference in interaction rates with leatherback sea turtles (see Table 99).

Table 98. Interaction rates compared between Hawaii shallow-set longline fishery and non-domestic (foreign) WCPFC shallow-set fisheries (excludes Hawaii) for loggerhead sea turtles, 2013 and 2017 (LHST = loggerhead sea turtle; estimates for the total foreign fisheries are reported as the median predicted interaction rate, and the lower and upper 95% confidence intervals (LCI and UCI), calculated with data reported by Peatman et al. 2018. Numbers in italics highlight instances when the foreign interaction rate is less than the Hawaii shallow-set longline fishery.

Year	**HSLL Intxn Rate LHST**	**Foreign Fisheries Interaction Rate**		
		N of 10N Median LHST	**LCI No 10N - LHST**	**UCI No. 10N LHST**
2013	0.00566	0.01133	*0.00454*	0.02801
2014	0.01011	0.01838	*0.00788*	0.04292
2015	0.01052	0.02640	0.01222	0.05907
2016	0.02085	*0.02052*	*0.00881*	0.04730
2017	0.02045	*0.00819*	*0.00207*	0.03214
average	0.01352	0.01696	0.00710	

Table 99. Interaction rates compared between Hawaii shallow-set longline fishery and non-domestic (foreign) WCPFC shallow-set fisheries (excludes Hawaii) for loggerhead sea turtles, 2013 and 2017 (LEST = leatherback sea turtle; estimates for the total foreign fisheries are reported as the median predicted interaction rate; calculated with data from Peatman et al. 2018. Numbers in bold highlight instances when the foreign interaction rate is less than the Hawaii shallow-set longline fishery.

Year	**HSLL Intxn Rate LEST**	**Foreign Fisheries Interaction Rate N of 10N Median LEST**
2013	0.01037	*0.00799*
2014	0.00944	*0.00765*
2015	0.00405	0.00857
2016	0.00695	*0.00652*
2017	0.00389	0.00435
average	0.00694	0.00702

The interaction rates described in Table 98 and Table 99 suggest that available studies that examine the 2001 -2004 closure to assert a market transfer effect and subsequently, a beneficial effect from the existence of the HI SSLL fishery are no longer relevant. There appears to have been a significant change in interaction rates in foreign longline fisheries, such that shifting fishing effort from U.S. fleet to foreign fleets would not likely affect sea turtles as those studies claim (positively or negatively). Even if we agree that the “but for” test of causality was met by evidence presented by the Sarmiento (2006), Rausser et al. (2009), Chan and Pan (2012, 2016), and others, it is important to note that the relationship only necessarily holds true for the period of the closure. That is, these authors have demonstrated an association between the closure and a possible increase in turtle interactions during the closure period. We cannot conclude from these studies there is a beneficial effect from the continued operation of the HI SSLL fishery to sea turtles, generally, or to the specific sea turtle species adversely affected by the HI SSLL fishery.

Importantly, the purpose and obligation of section 7(a)(2) of the ESA is that Federal agencies are required to insure their proposed actions are not likely to jeopardize listed species or destroy or adversely modify designated critical habitat. NMFS is required to assess the direct and indirect effects of the authorization of the HI SSLL fishery and indirect effects must be both *caused by* the proposed action a*nd reasonably certain to occur*. Given our analysis, instead of treating putative increases in the number of sea turtles captured and killed in foreign longline fleets as an “indirect effect” of the proposed action, we evaluate the effects of other fisheries, including foreign fisheries in the action area, on threatened and endangered species in the *Environmental Baseline*. Specifically, we treat foreign fisheries that share the action area with the domestic HI SSLL as “other human activities in the action area” that may affect the status of listed species in that action area. At a larger scale, we evaluate the positive and negative past, present, and future effects of those fisheries in the *Status of Listed Resources* to the extent we have information available.

6.5 .Cumulative Effects

"Cumulative effects", as defined in the ESA implementing regulations, are limited to the effects of future state, tribal, local, or private actions that are reasonably certain to occur in the action area considered in this opinion (50 CFR 402.02). 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. Because the action area is primarily a swath of the North Pacific Ocean and transit routes to Honolulu and California (Figure 6) and cumulative effects, as defined in the ESA, do not include the continuation of actions described under the Environmental Baseline, few actions within the action area are expected to result in cumulative effects.

Cumulative effects on the species addressed by this opinion are likely to occur as a result of worsening climate change, and any increase in fishing, ship traffic, and other actions described in the Environmental Baseline section. Such effects could include worsening of the climate change effects, and also could result in corresponding increases in fishing gear entanglements and ship strikes for ESA listed species. In addition, any increases in marine debris could also increase entanglements of all five species.

Global anthropogenic climate change is expected to continue and to therefore continue to impact sea turtles and their habitats. Rising temperatures at nesting beaches may continue to exacerbate a female bias in hatchling sex ratio and could also increase embryonic mortality if beaches are already at the high end of thermal tolerance for sea turtle nests (Matsuzawa et al. 2002). In addition, the number of severe storms is expected to increase with warming ocean temperatures which is expected to change the shape of nesting beaches and to wipe out nests. This has been documented in the Atlantic; comparisons were made between loggerhead and green turtle nesting and cyclone intensity and they found that hatching success declines with increased cyclone intensity (Van Houtan and Bass 2009). Only low-level nesting of greens takes place inside the action area. However, turtles that occur in the action area come from nesting aggregations that may be affected by impacts at their nesting beaches of origin throughout the Pacific. The best available demonstrations of the potential effects of sea level rise indicate that some sea turtle nesting beaches will lose a percentage of their current area by 2100 (Fish et al. 2005; Baker et al. 2006; Fuentes et al. 2009), however these were modeled on static systems and did not account for geomorphological dynamics, such as the natural sinking of islands or the natural growth of coral reefs to keep up with sea level rise. A quantitative analysis of physical changes in 27 atoll islands in the central Pacific over a 19 to 61 year period that corresponds with a rate of sea level rise of 2.0 mm/y shows that 86% of islands remained stable (43%) or increased in area (43%) while only 14% of study islands exhibited a net reduction in island area (Webb and Kench 2010, Van Houtan 2010), evidence that changes will not be uniform or predictable and sea level rise may or may not result in beach loss.

Alterations to foraging habitats and prey resources, changes in phenology and reproductive capacity that correlate with fluctuations in SST, and potential changes in migratory pathways and range expansion (all discussed previously in Environmental Baseline) are additional ways in which sea turtles may continue to be impacted by climate change. Many marine species, including the pelagic life stages of sea turtle species in the action area, forage in areas of nutrient rich oceanic upwelling, the strength, location, and predictability of which may change with increasing global temperatures (Harwood 2001).

Although there is much speculation about potential impacts of climate change to species and ecosystems, there are multiple layers of uncertainty associated with these analyses making it impossible to accurately predict the most likely scenario that will result and consequently what impacts species and ecosystems will face, particularly in Pacific Island countries (Barnett 2001). Effects of climate change will not be globally uniform (Walther et al. 2002) and information regarding the magnitude of future climate change is speculative and fraught with uncertainties (Nicholls and Mimura 1998). In particular, there is no comprehensive assessment of the potential impacts of climate change within the action area or specific to sea turtles, Guadalupe fur seals, giant manta rays, or oceanic whitetip sharks that may be within the action area.

In addition to the uncertainty of the rate, magnitude, and distribution of future climate change and its associated impacts on temporal and spatial scales, the adaptability of species and ecosystems are also unknown. Impact assessment models that include adaptation often base assumptions on when, how, and to what adaptations occur on theoretical principles, inference from observations, and arbitrary selection, speculation, or hypothesis (see review in Smit 2000). Impacts of climate change and hence its ‘seriousness’ can be modified by adaptations of various kinds (Tol et al. 1998). Ecological systems evolve in an ongoing fashion in response to stimuli of all kinds, including climatic stimuli (Smit et al. 2000). Sea turtles may exhibit a variety of adaptations to cope with climate change-related impacts, although it will likely take decades to centuries for both climate-related impacts and associated adaptations to occur (Limpus 2006) making it increasingly difficult to predict future impacts of climate change on these species in the action area. For example, sea turtles are known to be highly mobile and in the past have shown the ability to adapt to changes in their environment and relocate to more suitable foraging and nesting sites over the course of multiple generations. Implications of climate change at the population level are a key area of uncertainty and one of active research (e.g. Jonzén et al. 2007) and cannot currently be reliably quantified in terms of actual mortalities resulting from climate change impacts over any time scale, nor can they be qualitatively described or predicted in such a way as could be more meaningfully evaluated in the context of this biological opinion.

7 INTEGRATION AND SYNTHESIS OF EFFECTS

This section provides an integration and synthesis of the information presented in the *Status of the Listed Resources*, *Environmental Baseline*, *Cumulative Effects*, and *Effects of the Action* sections of this biological opinion. The intent of the following discussion is to summarize our evaluation of the risk that continuation of the proposed action, the HI SSLL fishery, poses endangered leatherback sea turtles, endangered North Pacific DPS of loggerhead sea turtles, Eastern Pacific DPS of green sea turtles, Central North Pacific DPS of green sea turtles, East Indian – West Pacific DPS of green sea turtles, Central West Pacific DPS of green sea turtles, Southwest Pacific DPS of green sea turtles, Central South Pacific DPS of 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 endangered Guadalupe fur seals in the Pacific Ocean. The *Status of the Listed Resources*, *Environmental Baseline* and *Cumulative Effects* discussions describes how these listed species affected by the proposed action have been adversely affected by human-induced factors such as commercial fisheries, direct harvest, and modification or degradation of habitat including marine debris and climate change. These sections lay the foundation for our evaluation.

The *Effects of the Action* section of this biological opinion, describes the direct and indirect effects of the continued authorization of the HI SSLL fishery, which we expect would continue in perpetuity since longlining has a history of more than 100 years in Hawaii (Kaneko 2015). NMFS approved the Pelagic FMP in 1987, and established the Federal longline permit and logbook reporting requirements in 1991.

In 2004, the Hawaii longline fishery was split into a shallow and deep set sector via a Regulatory Amendment. Since 2004, the HI SSLL fishery has had 100% observer coverage, which makes for a strong data set on observed interactions (i.e. hooking and entanglement) with threatened and endangered species over 15 years. Data on interactions with listed species are available for a longer period, however, interaction rates dropped considerably for sea turtles following the implementation of gear changes adopted with the reopening of the HI SSLL fishery in 2004. Since then, interactions have declined by 84% for leatherback sea turtles and 95% for loggerhead sea turtles (Swimmer et al. 2017). Interactions with green sea turtles and olive ridley sea turtles also dropped as a result of the change in operations. In contrast, interactions with Guadalupe fur seals are a relatively new phenomenon, likely owing to range expansion of this species. Interactions with oceanic whitetip sharks and giant manta rays may have a long history in this fishery but proper identification and record-keeping on interactions with these species has resulted in a highly variable and sometimes suspect data set particularly in earlier years for these two threatened species. For all of these reasons, our effect analysis is focused largely on the interaction data available from the past 14 – 15 years. These data formed the backbone to our effects analysis, and although we acknowledge the action may have effects that we cannot observe or quantify, our analysis describes that the effects of other stressors, aside from hooking and entanglement, are likely diffuse or otherwise rare, and probably do not pose high risks to listed species.

In the *Approach to the Assessment* section of this biological opinion, we stated that 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 animals 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 (Anderson 2000; Mills and Beatty 1979; Stearns 1977, 1992). As a result, if we conclude that listed animals are *not* likely to experience reductions in their fitness, we would conclude our assessment. If our analysis shows that the effects of the action are likely to reduce the viability of one or more populations, and that reduction in turn is likely to be sufficient to reduce the viability of the species those populations comprise, then we would conclude that the action appreciably reduces the likelihood of survival and recovery of the species.

An appreciable reduction in a species' likelihood of surviving and recovering in the wild is not necessarily a quantitative calculation; in fact we rarely have adequate information on the abundance and trend, growth rate, spatial structure or other indicators of population and species diversity to place numeric values on a species chances of persistence. This means, even if we have strong data on an action's effects, our predictions of the long-term impact to the species are uncertain at best. Typically, available data can only provide short or patchy time series, which do not allow for detailed quantitative analyses of a species' decline to extinction (Fagan and Homes 2006).

This is the case for every listed species addressed in this biological opinion – time series data, or other metrics of population risk such as genetic and spatial diversity and demographic rates are poorly studied and understood, and only available as a patchy time-series, at best. Nevertheless, we must reach a conclusion at the close of every consultation as to whether the action is not (or, conversely, is) *likely to jeopardize the continued existence* of the threatened and endangered species, or result in the destruction or adverse modification of their designated critical habitat.

Rarely would we expect a federal action to be the singular cause of extinction, so the section 7 regulations expand our understanding of this idea by defining to *jeopardize the continued existence of* means as engaging 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* (emphasis added). In other words, in addition to other considerations, we must reach a conclusion as to size and severity of the additive effect the proposed federal action has on the species' risk of extinction.

We acknowledge that, as described by Brosi and Biber (2009), there is an inherent asymmetrical risk with threatened and endangered species because extinction is irreversible. For this reason, instead of using the mean number of incidental interactions in this biological opinion as the basis of our jeopardy analysis, which we have done in past consultations, we report the mean incidental intraction and mean mortality predictions, but we used the highest likely predicted interaction number and the highest estimate of mortality for our jeopardy evaluation. The flaw with averages is that years of highest interaction rates are canceled out by the years of lowest interaction rates; averages tell you nothing about the actual incidents (interactions) and they often present a misleading picture. This issue was discussed at length at the meeting the WPFMC convened in 2017 on Rare Events, at which time the statement was made more than once that using the average rate of interaction events (or other such metrics) means the agency must be willing to accept that its predicted interactions and mortality estimates are wrong 50% of the time. This means a biological opinion with an ITS based on averages will be wrong part of the time because it will under forecast the interaction rate with the listed species

Therefore, while we used statistical tests and models in our analysis, as mentioned in the *Approach to the Assessment* section of this biological opinion, our assessment focused on clinical (or biotic) significance of the effects of the action on threatened and endangered species. Clinical significance is a notional term from the medical community that recognizes the practical importance of a treatment — a treatment can be real, palpable, and noticeable, without being statistically significant. We used the causal loop diagrams to capture key interrelationships and build our understanding of the life cycle and the threats affecting the species to interpret the proposed action's effect on the survival and recovery of the species. If a proposed action has an adverse effect an endangered species with sufficient magnitude and severity, it could further reduce the species chances of survival and recovery.

This type of shift in the species liklihood of persistence may be considered a tipping point — the point at which a threatened or endangered species is pushed into a new systems state that specifically marks the threshold between adverse effects and jeopardy. We explore the notional idea of a tipping point in this context. The concept of a tipping point and the need for the Services to consider the tipping point between survival has arisen in the courts a few times (e.g. Oceana, Inc. v. Nat'l Marine Fisheries Serv., 705 F. App'x 577, 580 (9th Cir. 2017); Wild Fish Conservancy v. Salazar, 628 F.3d 513, 527 (9th Cir. 2010); Nat'l Wildlife Fed'n v. Nat'l Marine

Fisheries Serv., 524 F.3d 917, 936 (9th Cir. 2008)). In the Wild Fish Conservancy case, the Ninth Circuit 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 at 527. 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. The 9th Circuit rejected this argument because "a species can often cling to survival even when recovery is far out of reach."

If survival were defined to mean "not currently extinct," or "not likely to become extinct in the short-term," we can understand the concern. The most obvious point of no return is "simply at a population size of one (Scheffer 2010) and some species have increased in abundance with fewer than 10 individuals remaining (Fagen and Holmes 2010). In fact, prior to the passage of the ESA and also since its passage, we have witnessed the extinction of hundreds of species when their populations became too small to survive or recover. Booming Ben, the last heath hen, a unique eastern subspecies of the greater prairie chicken went extinct in the 1930s. By 1927, 13 heath hens existed, and by 1928 there were only two. By 1929, only "Booming Ben" survived. Named Booming Ben for the elaborate calls of the hens during mating, Ben remained silently alone, only to disappear by 1932 (Heisman 2016). The Po'ouli, or black-faced honeycreeper of Maui, once inhabited the slopes of the Haleakala volcano. Discovered in the 1970s, by 1997 only three Po'ouli were left. By 2004, the species was declared extinct. There are many more such examples: "Lonesome George" (the last Pinta Island tortoise), "Orange" (the last Dusky Seaside Sparrow), Laysan honeycreeper, the ivory-billed woodpecker, Lake Sammamish early-run Kokanee salmon, Bramble Cay melomys (a small mammal that recently listed as extinct), and the list goes on. In the decades that preceded these species' extinctions, their populations "survived," if we define that term to mean they were "extant" rather than "extinct." At some point, the extinction of these species was already certain and their chances of recovery had been foreclosed.

It is an established fact that species have reached and continue to reach a point where recovery is truly or effectively impossible (or would require extraordinary effort as in the cases of the black-footed ferret, California condor and New Zealand black robins). Species can also reach a point where their survival (persistence over 5, 10, 50, or 100 years of multiple generations) is impossible. Studies suggest there is a "transcritical bifurcation" or a "catastrophic bifurcation," also known as a tipping point, beyond which imperiled populations cannot be expected to recover (Fagan and Homes 2005, Drake and Griffen 2010, Scheffer 2010). Empirical evidence from lab experiments on zooplankton (*Daphnia*) showed the experimental population showed early warning signs of an upcoming bifurcation in the underlying growth equations (Drake and Griffen 2010). However, this was a controlled study; for wild animals we almost never know where these "tipping points" lie.

Arguably, however, under the section 7 of the ESA, there is little difference between the tipping points of survival and recovery. First, the regulation requires that we look at survival *and* recovery, so it is not enough that a species merely *survive* the action. Importantly, the action cannot also appreciably reduce the species chance of recovery. Neither does the ESA purport that holding a species at the level of merely surviving just above extinction is sufficient. The ESA Section 7 Handbook (1986), explicitly for determination of jeopardy, defines survival as:

> *the species' persistence* as listed or as a recovery unit, *beyond the conditions leading to its endangerment, with sufficient resilience to allow for the potential recovery from endangerment*. Said another way, *survival is the condition in which a species continues to exist into the future while retaining the potential for recovery*. This condition is characterized by a species with a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring, which exists in an environment providing all requirements for completion of the species' entire life cycle, including reproduction, sustenance, and shelter (page xix).

Accordingly, our analysis hinges on the effect of the proposed action and whether it would be expected, directly or indirectly to reduce appreciably the likelihood of the recovery of a listed species in the wild by reducing the reproduction, numbers or distribution. While we cannot know that precise tipping point between survival and recovery, there are some early warning signals of small populations that signal near certitude of extinction (and no likelihood of recovery) in one or two generations (without the influence of an unknown factor, extraordinary luck, or a last-ditch intervention that usually consists of removing the entire population from the wild). Factors we reviewed that are designed to help us evaluate extinction risk include: population size, trends, abundance and growth rate; quantitative models of risk such as population viability models; habitat fragmentation and threats; demographic traits: age at maturity, size, fecundity, remigration intervals, death rates, birth rates, sex ratios, etc.; spatial diversity; and genetic diversity, among others. One of the more important metrics, and often the simplest for us to obtain, is population size or some estimation of population size. Often the larger the population size, the larger the buffer the population has against extinction and the greater its chances of recovery.

To satisfy the purpose of the ESA, it is necessary to prevent species from declining to the point where their populations exhibit small population dynamics. For example, the record of animal species extinctions suggests that when a species' is reduced to a single population and that 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 (although work conducted by Fagan and Holmes (2010) suggests that populations numbering 300 individuals are not necessarily protected from these phenomena). If even one of those individuals dies prematurely or misses an opportunity to reproduce, we would anticipate a bifurcation that might preclude the species persistence over 1 or 2 generations (= "survival") and its recovery. None of the species affected by the proposed action are quite so small in numbers as these examples, as far as we know, although some populations that comprise the species may be quite small. Some of the species effected by the proposed action, however, exhibit traits that are cause for concern like declining abundance. One way the Services could address the issue of a tipping point is to identify a quasi-extinction threshold for each species that is high enough to avoid small population dynamics (Allee effects). However, such a number is not identified (it would necessarily need to be unique to each threatened and endangered species), and in fact doing as much could cause unintended problems if the Services or federal agencies were to manage the species at a quasi-extinction threshold. In any case, the Services have declined to create such a metric.

In recent years, the idea of a limit reference point (LRP) has been a hotly debated issue. In 2015, Curtis et al. (2015) published a paper that introduced the notion of LRPs and used the West Pacific Ocean leatherback sea turtle as their case study. The notional idea of LRPs is similar to the fishery concept of optimum sustainable yield (OSY), that there is an acceptable level of risk tolerance that can be identified numerically by creating an estimate of the maximum number of individuals that could be removed from a population without exceeding a particular risk of falling below a population threshold. While LRPs may also provide a way to look at impacts with a common standard or “currency,” they differ in that they are meant to help inform the development of operational objectives that correspond to attaining or maintaining a specified population threshold. While such metrics can help inform management actions, they cannot satisfy the statutory or regulatory requirements of the ESA. The ESA generally prohibits the taking of listed species in order to ensure that the effects of human activities are not expected to appreciably reduce the likelihood of both survival and recovery of those species in the wild. Although incidental take may be permitted under the ESA, any such take must be mitigated or minimized. For Federal actions such as the one analyzed in this biological opinion, section 7 of the ESA requires the minimization of the level of take on the individuals affected through reasonable and prudent measures to reduce the impact of the take and terms and conditions that are used to implement the reasonable and prudent measures.

In contrast, the objectives of the Magnuson-Stevens Fishery Conservation and Management Act (MSA) are to: 1) prevent overfishing; 2) rebuild overfished stocks; 3) increase long-term economic and social benefits; and 4) ensure a safe and sustainable supply of seafood. NMFS seeks to achieve these objectives through maintaining an “optimum sustainable yield” of fish stocks. The term "optimum” in this context means the amount of fish that will provide the greatest overall benefit to the Nation while taking into account the protection of marine ecosystems among other concerns, while also providing for the rebuilding of overfished stocks to a level consistent with producing the “maximum sustainable yield” of harvested fish. Maximum sustainable yield in this context is the greatest number of fish that can be caught each year without impacting the long-term productivity of the stock, whereas optimum sustainable yield is the target. Thus, the LRP under the MSA is maximum sustainable yield.

In the case of the MMPA and the MSA, marine mammal and fish stocks, respectively, are offered a range of conservation strategies depending on the relative condition of the stock. The requirements of the ESA, however, are predicated on the assumption that all ESA-listed species populations are at low or unhealthy levels and are therefore in need of recovery. The distinction between maintaining healthy populations and recovering unhealthy populations is important as it provides key insights into how the information obtained from the proposed methods should be viewed under the respective statutes.

The purpose of the ESA is to protect and recover imperiled species and the ecosystems upon which they depend. The ESA also has a requirement to minimize the effects of any authorized actions, and to minimize the impacts of any incidental take expected from those actions in order to protect and recover those species. Under the ESA, any authorized activity must not appreciably reduce the likelihood of both survival and recovery of the species. The focus is not simply to maintain the species at a predetermined level but to recover the species to the point at which protections pursuant to the ESA are no longer necessary (definition of conserve”, and section 4(f)(1)(g) of the ESA of 1973). For this reason, and because the 1986 Handbook defines survival as "*the species' persistence as listed or as a recovery unit, beyond the conditions leading to its endangerment, with sufficient resilience to allow for the potential recovery from endangerment"* we relied on population metrics and our understanding of the threats on the species to evaluate whether the proposed action, the continued authorization of the HI SSLL fishery, 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*

Our analyses find that the proposed action, while it does contribute to the mortality of endangered leatherback sea turtles, endangered North Pacific DPS of loggerhead sea turtles, Eastern Pacific DPS of green sea turtles, Central North Pacific DPS of green sea turtles, East Indian – West Pacific DPS of green sea turtles, Central West Pacific DPS of green sea turtles, Southwest Pacific DPS of green sea turtles, Central South Pacific DPS of 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 endangered Guadalupe fur seals, the action has a very small effect on these species, such that we believe it does not appreciably reduce their chances of survival and recovery in the wild. We explain the basis for this conclusion 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 the Species* and *Environmental Baseline* sections of this biological opinion to assess the potential risks that the HI SSLL 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.

7.1 Leatherback Sea Turtle

NMFS has managed the HI SSLL fishery for more than 34 years, and has been consulting on the effects of the Hawaii pelagic longlines for at least that long. See Table 100 for a snapshot of this history, as it relates to leatherback sea turtles. Between 2001 and 2004, there were significant changes made in the operation of the longline fisheries. In 2001, NMFS issued a jeopardy biological opinion on the operation of this fishery (at the time consultations were on the mixed fisheries) for leatherback sea turtles, and a few other species. In 2001, the fishery was estimated to interact with between 75 to 157 adult and subadult leatherback sea turtles per year, and between 28 and 57 were estimated to die as a result of those interactions. Significant changes in the fishery led to a significant reduction in interactions for all sea turtles with the reopening of the fishery in 2004. In 2004, that reduction led NMFS to conclude that the action was not likely to jeopardize the leatherback sea turtle. In 2004, NMFS expected that the fishery would interact with 16 leatherback sea turtles and two would die from their interactions. In the 2008 biological opinion NMFS anticipated the fishery would interact with 19 leatherback sea turtles, and five would die as a result of their interactions. However, the 2008 biological opinion included a term and condition stipulating an annual interaction limit of 16. Two years later, in 2011, the cap of 16 was reached and the fishery closed. In 2012, NMFS reissued a new opinion that estimated the fishery would interact with 26 leatherback sea turtles and kill up to 6 each year. See Table 1. In the past fourteen years, from 2004 through 2018, the HI SSLL fishery has interacted with 105 adult and subadult leatherback sea turtles. As discussed in earlier sections of this opinion, these 105 interactions have likely led to the mortality of 21 leatherback sea turtles. Based on the observer data from 1994-1999, about 85% of the leatherback sea turtle interactions in the mixed fishery were attributable the shallow-set fishery, then over the past 24 years, the HI SSLL fishery has interacted with more than 650 leatherback sea turtles, most of which have been adult animals or nearly so.

Importantly, changes in the fishery in 2004 significantly changed the past interaction rate for this species and other turtles. So we rely upon data from the past 14 years to evaluate the present trend in interactions, and not all years are equally informative because the fishery closed early during some years because the cap for loggerhead sea turtles or leatherback sea turtles was met (2006 and 2011). Because of closures in 2006 and 2011, observed numbers of interactions are for a partial year of fishing. Therefore, to develop our Bayesian predictions of future interactions we intentionally excluded some of the interactions to achieve model convergence. Importantly, our data set is small, and as a result there remains much uncertainty in our predictions on the numbers of leatherback sea turtles that the HI SSLL fishery is likely to interact with in the future. Our best estimate is that the HI SSLL fishery will hook and entangle up to 21 leatherback sea turtles each year, and from these interactions up to 3 will die. Assuming this fishery continues for at least another 25 years, we would expect that the HI SSLL fishery would hook or entangle up to 525 more leatherback sea turtles, and up to 75 of these animals would die from their interactions.

Given the small data set of observed interactions, and the inherent weakness in the mortality estimates for leatherback sea turtles, our analytical approach is conservative to account for uncertainty in the data and ensure that where appropriate we provided the benefit of the doubt to the listed species. First, post release mortality rates from Ryder et al. (2004), which we applied for all sea turtle species addressed in this biological opinion, are higher by 5 to 10% in each injury category for leatherback sea turtles to account for their physiological differences from that of hardshell turtles and evidence to suggest that this affects injury severity. We also, as discussed earlier in the *Approach to the Assessment*, purposefully used the 95 percentile (the upper estimate) from our Bayesian analysis of the number predicted interactions in a 1, 2, and 3 year time frame. We used this upper estimate of the predicted number of individual leatherback sea turtles to calculate estimated mortality, and we did not apply a discounting method (adult nesting equivalents) but instead assumed that each leatherback sea turtle that interacts with the fishery has an equal opportunity of reaching adulthood and contributing to the population as the next. Because the West Pacific Ocean leatherback sea turtle species is doing particularly poorly, we weighed the action against the lower estimates of the population's status. Finally, we also considered the effect of the action on one unique life history type, summer nesters, which is also somewhat cautious because we do not yet know the degree to which there phenotypic plasticity in these two life history strategies. We assume that the summer nester phenotype is unique, and important to the individual nesting female leatherback sea turtle but is important to maintaining diversity and resiliency of the population and the species. Each of one of these decisions, reasonably incorporates scientific uncertainty while also providing the benefit of the doubt to the species. This is important given the long-term declining trend of West Pacific Ocean leatherback sea turtles.

Table 100. Biological opinions, conclusions, estimated interactions and actual interactions of leatherback sea turtles, 2001-2018.

Year	Biological Opinion Conclusion	Estimated Interactions		Estimated mortality	Actual (or observed) interactions
1994-1999*		673		56	40
2001*	Jeopardy	75-157		28-57	
2002					
2003					
2004	No Jeopardy	16		2	1
2005					8
2006					2
2007					5
2008	No Jeopardy	19 (RPM limited to 16		5	2
2009		16			9
2010					8
2011	reached cap	16			16
2012	No Jeopardy	26		6	7
2013					11
2014					16
2015					5
2016					5
2017					4
2018					6

As discussed in the *Status of Listed Species* 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 effects only the West Pacific Ocean population. As discussed in the *Approach to the Assessment* section of this biological opinion, two of the primary variables that inform the status of the species are the number of demographically-independent units that comprise the species, and the performance of those independent units over time. The West Pacific Ocean population with the East Pacific Ocean populations exhibit a species that in the Pacific Ocean is in exponential decline (see Figure 5). The West Pacific Ocean population, is genetically and spatially distinct from other populations including the East Pacific Ocean population.

Leatherback sea turtles in the West Pacific Ocean population have declined 83% during the past three generations, and presently the population is composed of an estimated 1,438 individuals (mature adults). The IUCN predicts the population is likely to decline to about 572 nests, and 260 adult females by 2040 (Tiwari et al. 2013, Wallace et al. 2013). Based on NMFS' PVA model, leatherback sea turtles in the West Pacific Ocean population are declining at about 5%

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 three years (range 3-86 years, mean 26 years). PVA modeled estimates suggest the population presently consists of about 1180 adult female leather back sea turtles (median sum of total reproductive females, LCI = 949). Assuming a 3:1 ratio, females to males (reproductive) and based on NMFS' PVA, the total number of adult leatherback sea turtles in the West Pacific Ocean population is larger than that estimated by IUCN (about 1,770 mature leatherback sea turtles). Notably the IUCN did not rely upon a PVA for their estimate; however, their estimate is also about 5 years older than NMFS' PVA. While we would expect differences in estimates produced under different methodologies, we think that the long term declining trend in this particular population warrants an abundance of caution. Therefore, for the purposes of this assessment we think it prudent to rely on NMFS' lower estimate of female abundance (949), which when we assume a 3:1 ratio, females to males, we would expect that the population is composed of only about 1,424 adults (949 females and 475 males).

West Pacific Ocean 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, summer nesters are more likely to interact with the HI SSLL fishery because these individuals migrate across the Pacific Ocean and forage off the West Coast during the summer and fall. According to Benson et al. (2011), a large portion of summer nesters, between 38 and 57%, forage in waters of the US West Coast. Although, the HI SSLL fishery may disproportionately affect the summer nester life history strategy evidence suggest that the winter nester life history strategy may be at greater risk based NMFS' PVA analyses by Jones et al. (pers comm). Because we poorly understand the relationship between the two life history types, we did not rely on a PVA for summer nesters only. The effect of the pooled data set is that estimates were driven down by the inclusion of winter nesters (more conservative). This is not surprising as the winter nesters may be at greater risk of direct harvest in areas like the South China Seas.

We recognize that although we believe the data capture the largest of the nesting aggregations, Jamursba-Medi and Wermon, monitoring has been incomplete among other beaches and years, and yet these other areas likely have independent trends. Moreover, our data and our PVA cannot help us understand the status of male leatherback sea turtles, which are important to the persistence of the species. Absent some analysis of the entire West Pacific ocean meta-population, and all demographically independent units that comprise the metapopulation, with some assessment of the probable trends of the total population, we cannot know the true status of the species. For these reasons, using the lower estimate of the population size seems appropriate.

As described in the *Environmental Baseline*, effects from global climate change, marine debris, and international and U.S. 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 Ocean 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, an estimated 600 leatherback sea turtles are taken in North Pacific longline fisheries, which includes the HI SSLL fishery, each year (Peatman et al. 2018; BDEP data tables). If this is correct, then some leatherback sea turtles are likely exposed to more than one fishery interaction each year. Assuming a mortality rate similar to the estimated mortality rate in the HI SSLL (0.20), then we can expect about 120 leatherback sea turtles die from their exposure to fisheries each year on average in the North Pacific. 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 35% of the turtles that the HI SSLL fishery interacts with, and most of the leatherback sea turtles are subadults or adults. A higher percentage of leatherback sea turtles are released with trailing gear, which increases their risk of mortality. Most leatherback sea turtles in this fishery cannot be boarded onto the vessel because of their size, and thus gear is often left on the animal after an interaction. In thirteen years, from 2005 through 2018, the average number of annual interactions in this fishery is 8 leatherback sea turtles, which means the HI SSLL fishery exceeded 8 interactions half of the time with the highest number of interactions on record as 16 leatherback sea turtles.

Our Bayesian analysis predicts that the mean interaction rate will increase slightly (to 10 each year), and our upper prediction estimates that the HI SSLL fishery would interact with 21 animals or less each year. The hooking or entanglement of 21 leatherback sea turtles per year is expected to result in the mortality of up to 3 leatherback sea turtles per year from their exposure as described in detail in the *Exposure and Response* section of this biological opinion. Assuming there are currently 1,424 adult leatherback sea turtles in the West Pacific Ocean population, the loss of 5% of these individuals means 71 adult leatherbacks would be expected to die from the cumulative stressors that they are exposed to between their natal beaches and migratory routes to and from feeding areas. The contribution of the HI SSLL fishery to this total mortality, using a point estimate for one year only, is up to 4% (3/71 = 4%). Assuming the mortality of males and females is proportionate to their occurrence in the population, we would expect about 47 of the 71 animals that die in a given year would be female. If the fishery disproportionately affected female leatherback sea turtles then the HI SSLL fishery may contribute 6% of the annual mortality to this population. The loss of 3 individual leatherback sea turtles from a population comprised of 1,424 adult animals represents 0.20% of the population. If we assume that all three are females, then the loss of 3 adult nesting females from a population of 949 females is 0.3% of the adult nester population.

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 Ocean 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 Ocean 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 Ocean population, which is the more robust of the two populations in the Pacific Ocean. A species vulnerability is related to the vulnerability of its populations; in other words, the extinction of populations influences the probability of the survival and recovery of the entire species (Ehrlich and Daily 1993).

While the continuation of the HI SSLL fishery is expected to interact with up to 21 individual adult leatherback sea turtles and result in the mortality of up to 3 leatherback sea turtles annually, in many years the total mortality is likely to be less than 3. This number is sufficiently small that we would not expect it to reduce appreciably the species chances of survival and recovery in the wild. Because of baseline conditions that continue to act on the species, this population is already at high risk of extinction, and the continuing declining trend is of particular concern. Our task in a section 7 consultation is to consider whether the nature and magnitude of the proposed action's effects, when considered together with the species status and all other threats acting on it, are consequential enough to appreciably reduce the species' likelihood of survival and recovery. Our analysis suggests that the proposed action's effects, which include interacting with 21 leatherback sea turtles and killing up to three leatherback sea turtles each year, would have a minor or inconsequential impact on the species' overall reproduction, numbers and distribution in the wild. Based on our analysis, we expect little consequential change in the species' chances of survival and recovery with or without the effects of the proposed action. The cumulative effect of other stressors, including other fisheries must be removed or abated or this species will reach a catastrophic bifurcation (tipping point) where recovery is no longer possible. Today, based on the present population abundance, we expect that there is time yet to ensure that the chances for the recovery of the West Pacific Ocean population are not yet foreclosed.

Importantly, Congress envisioned that the ESA would provide a program for the conservation of threatened and endangered species and they ecosystems that they depend upon, and that federal agencies would use their authorities to conserve threated and endangered species. Over the last 13 years, NMFS has gained valuable information from sea turtle studies like Howell et al. (2008, 2015), which led to the development of TurtleWatch (released in 2006). NMFS also supports research and grants to monitor nesting beaches and conduct outreach in Indonesia. These efforts continue to provide important information on leatherback sea turtles, and improve our understanding of their biology and population dynamics, as well as the threats they face and their status. Nevertheless more must be done to ensure the recovery of leatherback sea turtles. NMFS has not released all of the information it has available for the TurtleWatch program, which could be used to assist the fishery in minimizing its effects on leatherback sea turtles. Further, NMFS has not investigated the survivability of adult leatherback sea turtles in this fishery, and still knows very little about the long term prognosis of those individual adult leatherback sea turtles that interact with this fishery. Given our concern for the West Pacific Ocean leatherback sea turtle's status, immediate additional steps to help mitigate the effect of the HI SSLL fishery on leatherback sea turtles are necessary.

This analysis suggests that the continued authorization of the HI SSLL fishery would be expected to kill a small number of leatherback sea turtles each year, but is not likely to reduce the viability of the endangered leatherback sea turtle species. In other words, when considered in the context of the species' status and other baseline threats, we would not expect the proposed action to have consequential effects at the species level that appreciably reduce the likelihood of survival and recovery in the wild. Even so, given the species precarious status, the effects of the proposed action, the section 7 requirement to minimize the effect of the action and the unique requirements on NMFS to further the purpose of the ESA, we present NMFS with reasonable and prudent measures and implementing terms and conditions designed to further mitigate the impacts of incidental take on endangered leatherback sea turtles that are exposed to the proposed action for another 25 or more years.

7.2 North Pacific Loggerhead Sea Turtle DPS

Loggerheads from the North Pacific DPS are listed as endangered and have faced declines of up to 90% since the 1950s and over the last two decades have had an oscillating trend of nester abundance as described in the *Status of Listed Resources.* There are no estimates for the historical abundance but recent abundance estimates have estimated that there are 43,226 loggerhead turtles (CV = 0.51, 95% CI range = 15, 017 to 100,444) in the DPS based on surveys conducted in the Eastern Pacific (Seminoff et al. 2014). The current median female abundance estimate for 2013-2015 is 3,652 (95% CI, 2976 to 4468) (Jones et al. 2018). Over the last decade one subpopulation of the DPS has been on a slight upward trend based on nest counts and a recent PVA which provides an index of population growth derived from nest counts (Jones et al. 2018). The mean log growth rate for the subpopulation is 0.024 (95% CI, -0.108 to 0.156) from the PVA. Under stable environmental conditions, this growth rate would be constant and a population would increase geometrically (λ >1), decrease geometrically (λ <1), or remain the same (λ =1). The λ mean is 1.024 (95% CI, 0.897 to 1.168), which suggest that most trajectories of this subpopulation will increase slightly (Jones et al. 2018). Based on NMFS' PVA model, loggerhead sea turtles in the North Pacific DPS are increasing at about 2% per year, and the population as indicated by the index beaches has 25% probability of falling to less than half of its current abundance in the next 50 years (range 2-99 years, mean 49 years) (Jones et al. 2018). The model used nesting data from three index beaches in Yakushima, Japan – Inakahama, Maehama, and Yotsusehama; which represent approximately 52% of loggerhead nesting in Japan as described in the *Status of Listed Resources*. The other two populations have not had consistent monitoring and/or the data was not available to do a PVA and have not shown similar growth trends in the information that is available.

Loggerhead sea turtles are exposed to the stressors of hooking and entanglement in the *Action Area* and since 2004, approximately 177 have been hooked or entangled during HI SSLL fishing. Since 2004, two loggerhead sea turtles came up dead and another 26 loggerhead sea turtles died at a later time from injuries caused by their hooking or entanglement based on the NMFS post-hooking moratlity criteria. NMFS predicts that the ongoing HI SSLL fishery will hook or entangle up to 36 loggerhead sea turtles annually and up to six may to die each year from their exposure as described in the *Exposure* and *Response* sections of this biological opinion.

Based on genetic analysis, all of the individual loggerhead sea turtles caught in this fishery are part of the North Pacific DPS; however, we do not know with certainty what portion of the loggerhead sea turtles that the fishery interacts with would be from the three different subpopulations. Dutton performed a mixed stock analysis from loggerheads caught in the HI SSLL fishery and estimated that 26% (CI 0-94%) from mixed stock analysis and 40% (CI 0-99%) from weighted priors of population size could be from the Yakushima subpopulation. For the mainland subpopulation he calculated that 55% (CI 0-98%) from mixed stock analysis and 50% (CI 0-98%) for weighted priors could be from this subpopulation. For the Ryuku subpopulation Dutton calculated that 18% (CI 0-67%) from MSA and 9% (CI 0-52%) from the weighted priors could be from this subpopulation. We would expect that the HI SSLL fishery would likely interact with individuals from any of the subpopulations, and the chance of an interaction is likely proportional to the size of the subpopulation. We have no information to expect otherwise.

Fisheries bycatch from pelagic longlining and artisanal coastal fisheries are the greatest threat to individual fitness and to the DPS as described in the *Status of Listed Species* section. There is a great deal of uncertainty regarding which subpopulations face the greatest threat from fisheries. While there has been an increase on Yakushima nesting beaches over the last decade, some beaches on the mainland have increased slightly while others have decreased (Matsuzawa et al. 2016). The growth rate of a strong population, like the Yakushima subpopulation, could mask declines in other (smaller) subpopulations . The lack of information on the demographic units that comprise the DPS is cause for concern because we cannot know their abundance, growth rates, or extinction risk.

As described in the *Environmental Baseline*, effects from international and U.S. fisheries have resulted in interactions with the loggerhead sea turtle in the *Action Area*. These activities are reasonably likely to continue, and may increase over time due to the effects of increased human population, and increased human consumption of fish products.

Climate change may be impacting the species already but will likely increase in the future. Global average sea levels are expected to continue to rise by at least several inches in the next 15 years, and by one to four feet by 2100 (Wuebbles et al. 2017). This could affect migrations and feeding patterns by changing ocean circulation but will also reduce the beach access for nesting due to significant beach armoring in Japan. Increased sand temperatures can also cause decreased egg survival and an increase in female turtles skewing sex ratios and impacting the reproductive capacity of the population.

Based on the 177 loggerhead sea turtles from the North Pacific DPS that were caught by the SSLL fishery between 2004-2018 as described in the *Effects Analysis* section, NMFS predicts that the HI SSLL fishery will interact with up to 36 loggerhead sea turtles per year, (or up to 81 over a three-year period). From these we expect that up to 6 loggerhead sea turtles could die in any one year period (or up to 13 over a three-year period).

Therefore, the action is expected to reduce the abundance of individuals in the population, which may consequently affect the population's viability. Up to six mortalities on an annual basis plus the loss of their potential offspring from a DPS that has over 40,000 individuals, is a small proportion of the overall population and not expected to reduce the fitness of the species. However, if those six individuals and all of their potential offspring are from the smaller declining subpopulation, the risk to the DPS may be higher. We calculated what the potential impacts to each subpopulation would be based on the number of expected mortalities on an annual basis using the relative population size as weighted priors in the genetic analysis from Dutton as described in the *Status of Listed Resources* and *Effects Analysis*. We estimate that 14 individuals from the Yakushima subpopulation could be hooked or entangled and that two of those could result in mortalities, which accounts for 0.01 % of the subpopulation. We estimate that 18 individuals from the Mainland subpopulation could be hooked or entangled and that three of those could result in mortalities, which accounts for 0.01 % of the subpopulation. We estimate that three individuals from the Ryuku subpopulation could be hooked or entangled and that one of those could result in a mortality, which accounts for 0.03 % of the subpopulation. We also calculated what the potential impact would be for Ryuku subpopulation based on the minimum population size of 15,017 for the entire DPS and if all mortalities from the action were from this smaller subpopulation; this results in 0.4% of the subpopulation.

The NMFS and USFWS (1998) loggerhead sea turtle recovery plan contains a number of goals and criteria that should be met to achieve recovery. These include reducing, to the best extent possible, take in international waters; identifying regional stocks to source beaches; ensuring all females estimated to nest annually (FENA) at "source beaches" are either stable or increasing for over 25 years; ensuring each "stock" has an average 5,000 FENA (or a biologically reasonable estimate based on the goal of maintaining a stable population in perpetuity) over six years; ensuring foraging areas are maintained as healthy environments; ensuring foraging populations are exhibiting statistically significant increases at several key foraging grounds within each stock region; ensuring all priority #1 tasks have been implemented; ensuring a management plan designed to maintain stable or increasing populations of turtles is in place; ensuring there is a formal cooperative relationship with a regional sea turtle management program; and ensuring international agreements are in place to protect shared stocks (e.g., Mexico and Japan). Priority #1 tasks include a number of actions, including but not limited to, monitoring of nesting activity, determining population trends, identifying stock boundaries, reducing incidental mortality in commercial fisheries, and ensuring protection of marine habitat.

We conclude that the incidental take and resulting mortality of North Pacific loggerhead turtles associated with the direct and indirect effects of NMFS' continued authorization of the HI SSLL fishery would not be expected to appreciably reduce, the endangered North Pacific loggerhead DPS sea turtle's likelihood of surviving and recovering in the wild. We expect the overall population to remain large enough to maintain genetic heterogeneity, broad demographic representation, and successful reproduction. The proposed action will have a small effect on each of the subpopulations and therefore on the overall size of the population, and we do not expect it to affect the loggerheads' ability to meet their lifecycle requirements and to retain the potential for recovery.

7.3 Green Sea Turtle

7.3.1 Central North Pacific DPS

As described in the *Status of Listed Species*, the green sea turtle Central North Pacific DPS is listed as threatened and since initial nesting surveys at the FFS index beach in 1973, there has been a marked increase in annual green turtle nesting (Balazs and Chaloupka 2004). This increase over the last 40 years corresponds to an annual increase of 5.4% (Balazs et al. 2015). Between 1973 and 2015, nesting activity has been variable, as is typical of green turtle nesting dynamics, ranging between a low of 67 in 1973 and an all-time high of 808 nesting females observed during the 2011 six-week sampling period at East Island, FFS (with a total estimate of 843 nesters for the season) (NMFS-PIFSC unpubl. data). In-water abundance of green turtles is consistent with the increase in nesting trends (Balazs 1996; Balazs and Chaloupka 2004; Chaloupka et al. 2007). In addition, there has been a dramatic increase in the number of basking turtles in the main Hawaiian Islands and throughout the NWHI (Balazs 1996; Balazs and Whittow 1982; Parker and Balazs 2010).

While the nesting population trajectory is positive and encouraging, more than 96% of nesting occurs at one site in the NWHI and it is highly vulnerable to threats. Results of mtDNA analysis indicate a low level of spatial structure and low genetic diversity within the DPS. Survival of this DPS is currently highly dependent on successful nesting at FFS (Niethammer et al. 1997). There has been a significant constriction in the spatial distribution of important reproduction sites, presenting a challenge to the population's future and making this DPS highly vulnerable. As discussed in the *Status of Listed Species*, in October of 2018, East Island was demolished by Hurricane Walaka. Impacts to the population are unknown at this time and warrant further research moving forward.

Central North Pacific green turtles are exposed to a variety of natural threats both at their nesting beaches and in the open ocean. The beaches where turtles nest and the nest themselves are threatened by hurricanes, tropical storms, storm surges, sand accretion, and rainfall associated with storms. Predators such as sea birds, and ghost crabs (*Ocypode* spp.) hunt the hatchlings. Sharks and other fish will also eat the hatchlings when they enter the marine environment. Large grouper (*Epinephelus tauvina*) are documented predators of post-hatchling green turtles in Hawaii; however, the extent of grouper depredation is unknown (Balazs 1995). Adult turtles are also killed by sharks and other marine predators (Seminoff et al. 2015). Fibropapillomatosis affects green turtles from this DPS (Francke et al. 2013). This disease results in internal and/or external tumors (fibropapillomas) that may grow large enough to hamper swimming, vision, feeding, and potential escape from predators. Anthropogenic threats include contamination and degradation of foraging areas due to nearshore development, land based sources of marine pollution and increased human activity, contamination due to past military practices, vessel groundings, and fishing practices.

In summary, the concentrated nature and relatively small size of the DPS make it vulnerable to random variation and stochasticities in the biological and physical environment, including natural catastrophes, as well as changes in climate and resulting effects such as sea level rise, and other anthropogenic threats. This increases its risk of extinction, even though it may have positive population growth (Seminoff et al. 2015). Both non-stochastic as well as stochastic events are significant current and future threats to this small, isolated, concentrated population.

As described in the *Environmental Baseline*, past and present fisheries interactions have been, and continue to be, a threat to Central North Pacific green turtles within the *Action Area*. Bycatch of green sea turtles 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 U.S. in recent years, large numbers of Central North Pacific green sea turtles are still routinely captured in international, federal and state commercial fisheries that target other species. Some Central North Pacific green turtles also interact with recreational hook-and-line fisheries. These activities are reasonably likely to continue, and may increase over time due to the effects of increased human population and increased human consumption of fish products.

In addition, warming sea surface temperatures may lead to potential fitness consequences in sea turtles resulting from altered seasonality and duration of nesting (Pike et al. 2006). Sea turtles may also expand their range as temperature-dependent distribution limits change (McMahon and Hays 2006; Poloczanska et al. 2009a). Further, sea turtles have temperature-dependent sex determination, and many populations produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Jensen et al. 2018; Newson et al. 2009; Patrício 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). Lastly, studies examining the spatio-temporal coincidence of marine turtle nesting with hurricanes, cyclones and storms suggest that cyclical loss of nesting beaches, decreased hatching success and hatchling emergence success could occur with greater frequency in the future due to global climate change (Hawkes et al. 2009). Environmental changes associated with climate change are occurring within the *Action Area* and are expected to continue into the future. Marine populations that are already at risk due to other threats are particularly vulnerable to the direct and indirect effects of climate change.

As described in the *Exposure Analysis*, hooking and entanglement are the most significant stressors that green sea turtles are exposed to directly from the action. Green sea turtles interact with the HI SSLL fishery in low numbers. In 14 years, 10 green sea turtles have been observed in this fishery. All ten of these were recorded with sufficient information to evaluate post interaction mortality risk using the criteria of Ryder et al. (2006). All ten green sea turtles caught in the HI SSLL fishery between 2004 and 2018 were released alive with no gear attached.

As previously discussed in the *Status of Listed Resources*; since 1996, NMFS has evaluated the mtDNA of 19 green sea turtles captured in the SSLL fishery. The majority of the turtles caught in the HI SSLL fishery are predominantly comprised of animals from the Eastern Pacific DPS and the Central North Pacific DPS (Dutton, pers. Comm. 17 December 2018). NMFS estimates (Table 100) that between 20-64% of the turtles caught in the shallow-set longline fishery could be from the Central North Pacific DPS (P. Dutton pers. comm. November 20, 2017).

Table 101. Genetic composition of green turtles sampled from the Hawaii shallow-set longline fishery (Dutton pers. comm. August 31, 2016). The number of genetic samples column is from the direct approach and the MSA% is the mixed stock analysis with the 95% confidence interval.

DPS (ID #)	Number of genetic samples	MSA mean %	95% CI
Central North Pacific (10)	8	41	20-64

Based on the 10 green sea turtles that were caught by the SSLL fishery between 2004-2018 as described in the *Effects Analysis* section, NMFS predicts that the HI SSLL fishery will interact with up to 5 green sea turtles per year, (or up to 10 over three years, with one mortality). If we assume that 10 individuals from this DPS could be hooked or entangled every three years and that one of those could result in a mortality, this represents 0.0004% of the population based on a size of approximately 265,600 individuals.

Based on the evidence available, we conclude that the incidental take and resulting mortality of green sea turtles from the Central North Pacific DPS associated with the direct and indirect effects of NMFS' continued authorization of the HI SSLL fishery would not be expected to appreciably reduce, the threatened Central North Pacific DPS green sea turtle's likelihood of surviving and recovering in the wild. We expect the overall population to remain large enough to maintain genetic heterogeneity, broad demographic representation, and successful reproduction. The proposed action will have a small effect on the overall size of the population, and we do not expect it to affect the Central North Pacifc green sea turtle DPS' ability to meet their lifecycle requirements and to retain the potential for recovery.

7.3.2 East Pacific DPS

As described in the *Status of Listed Resources*, the green sea turtle East Pacific DPS is listed as threatened andnesting has been steadily increasing at the primary nesting sites in Michoacán, Mexico, and in the Galapagos Islands since the 1990s (Delgado and Nichols 2005; Senko et al. 2011). Nesting trends at Colola have continued to increase since 2000 with the overall Eastern

Pacific green turtle population also increasing at other nesting beaches in the Galapagos and Costa Rica (Wallace et al. 2010, NMFS and FWS 2007b). Based on nesting beach data, the current adult female nester population for Colola, Michoacán is 11,588 females, which makes this the largest nesting aggregation in the East Pacific DPS, comprising nearly 58% of the total adult female population. The total for the entire Eastern Pacific DPS is estimated at 20,112 nesting females (Seminoff et al. 2015).

Eastern Pacific green turtles are exposed to a variety of natural threats both at their nesting beaches and in the open ocean. The beaches where turtles nest and the nest themselves are threatened by hurricanes, tropical storms, storm surges, sand accretion, and rainfall associated with storms. The largest threat on nesting beaches to the East Pacific green turtle DPS 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 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 turtles throughout the Eastern Pacific Ocean. Significant bycatch has been reported in artisanal gill net and longline shark and mahimahi 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 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 U.S. 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. Some Central North Pacific green turtles also interact with recreational hook-and-line fisheries. 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.

Effects of climate change include, among other things, sea surface temperature increases, the alteration of thermal sand characteristics of beaches (from warming temperatures), which could result in the reduction or cessation of male hatchling production (Hawkes et al. 2009; Poloczanska et al. 2009), and a significant rise in sea level, which could significantly restrict green turtle nesting habitat. Impacts from global climate change induced by human activities are likely to become more apparent in future years (IPCC 2007). However, at the primary nesting beach in Michoacán, Mexico (Colola), the beach slope aspect is extremely steep and the dune surface at which the vast majority of nests are laid is well-elevated. This site is likely buffered against short-term sea level rise 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. 2009a). Further, sea turtles have temperature-dependent sex determination, and many populations produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Jensen et al. 2018; Newson et al. 2009; Patrício et al. 2017).

As described in the *Exposure Analysis*, hooking and entanglement are the most significant stressors that green sea turtles are exposed to directly from the action. Green sea turtles interact with the HI SSLL fishery in low numbers. In 14 years, 10 green sea turtles have been observed in this fishery. All ten of these were recorded with sufficient information to evaluate post interaction mortality risk using the criteria of Ryder et al. (2006). All ten green sea turtles caught in the HI SSLL fishery between 2004 and 2018 were released alive with no gear attached.

As previously discussed in the *Status of Listed Resources*; since 1996, NMFS has evaluated the mtDNA of 19 green sea turtles captured in the SSLL fishery. The majority of the turtles caught in the HI SSLL are predominantly comprised of animals from the Eastern Pacific DPS and the Central North Pacific DPS (Dutton, pers. Comm. 17 December 2018). NMFS estimates (Table 101) that between 32-77% of the turtles caught in the shallow-set longline fishery could be from the Eastern Pacific DPS (P. Dutton pers. comm. November 20, 2017).

Table 102. Genetic composition of green turtles sampled from the Hawaii shallow-set longline fishery (Dutton pers. comm. August 31, 2016). The number of genetic samples column is from the direct approach and the MSA% is the mixed stock analysis with the 95% confidence interval.

DPS (ID #)	Number of genetic samples	MSA mean %	95% CI
East Pacific (11)	11	55	32-77

Based on the 10 green sea turtles that were caught by the SSLL fishery between 2004-2018 as described in the *Effects Analysis* section, NMFS predicts that the HI SSLL fishery will interact with up to 5 green sea turtles per year, (or up to 10 over three years, with one mortality). Therefore, the action is expected to reduce the abundance of individuals in the population, which may consequently affect the population's viability. Up to one mortality over a 3 year period, plus the loss of their potential offspring, from a DPS that has over 20,000 nesting females, is a small proportion of the overall population and not expected to reduce the fitness of the species. We estimate that up to10 individuals from the East Pacific DPS could be hooked or entangled which could result in one mortality over a 3 year period, which accounts for 0.005 % of the nesting females in this DPS or 0.002% when assuming a 50:50 sex ratio and estimating the adult population size is 40,224.

Based on the evidence available, we conclude that the incidental take and resulting mortality of green sea turtles from the East Pacific DPS associated with the direct and indirect effects of NMFS' continued authorization of the HI SSLL fishery would not be expected to appreciably reduce, the threatened East Pacific DPS green sea turtle's likelihood of surviving and recovering in the wild. We expect the overall population to remain large enough to maintain genetic heterogeneity, broad demographic representation, and successful reproduction. The proposed action will have a small effect on the overall size of the population, and we do not expect it to affect the East Pacific green sea turtle DPS' ability to meet their lifecycle requirements and to retain the potential for recovery.

7.3.3 Central South Pacific DPS

As described in the *Status of Listed Resources*, the green sea turtle Central South Pacific DPS is listed as endangered and population trends in this DPS are poorly understood. Based on available data, we estimate there are nearly 3,000 nesters in this DPS. 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., 1995) and has reportedly significantly declined in the past 30 years as a result of commercial exploitation (Conservation International Pacific Islands Program, 2013). No sites have long-term monitoring programs, and no single site has had standardized surveys for even 5 continuous years. Most nesting areas are in remote, low-lying atolls that are logistically difficult to access. Unsurprisingly, many nesting areas (21 of 59, or 36 percent) only have qualitative information that nesting is present, indicating that there is still much to learn about green turtle nesting in this region (Table 13.2). As these unquantified rookeries most likely each have a female abundance in the 1–100 range, their collective sum is probably fewer than 700 nesters. When added to our 2,902 total, this DPS likely has fewer than 3,600 nesters. Green turtle temporal population trends in the Central South Pacific DPS are poorly understood, with not even a single nesting site having five contiguous years of standardized monitoring that span entire nesting seasons. Therefore, we have no data to conduct a PVA, or even a simple bar 314 chart of annual nesting activity. As previously discussed, no nesting sites have standardized monitoring that span entire nesting seasons. However, some monitoring data from the largest nesting site (Scilly Atoll) in this DPS, suggests significant nesting declines occur from persistent and illegal commercial harvesting (Petit 2013). Conversely, nesting abundance is reported to be stable to increasing at Rose and Swains Atoll, Tetiaroa, Tikehau, and Maiao. However, these sites are of moderate to low abundance and in sum represent less than 16% of the population abundance at Scilly Atoll alone. Nesting abundance is reported to be stable to increasing at Tongareva Atoll (White and Galbraith 2013). The uncertainty surrounding the above trends, and the lack of long-term monitoring and data from this DPS, presents significant challenges to any trend analyses (Seminoff et al. 2015).

Central South Pacific DPS green sea turtles are exposed to a variety of natural threats both at their nesting beaches and in the open ocean. We do not know the extent and level of threat from disease for the South Central Pacific DPS. Depredation may have been a factor that contributed to the historical decline of this DPS. The best available data suggest that current nest and hatchling predation on several Central South Pacific DPS nesting beaches and in water habitats is a potential threat to this DPS. Identified predators that may constitute a terrestrial threat to turtles include hermit crabs, ghost crabs, Polynesian rats, frigate birds (*Fregata ariel, F. minor*), and reef herons (*Egretta sacra;* Balazs, 1983). In the marine environment, sharks and other carnivorous fish (e.g., groupers) prey on sea turtles.

Directed take 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 (Tuato'o-Bartley et al. 1993; NMFS and USFWS 1998; Weaver 1996; Obura and Stone 2002; Alefaio and Alefaio 2006; Balazs 1983). 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 DPS 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 U.S. in recent years, large numbers of Central South Pacific DPS green sea turtles may still be captured in international and U.S. commercial fisheries that target other species. These activities are reasonably likely to continue, and may increase over time due to the effects of increased human population and increased human consumption of fish products.

In addition, warming sea surface temperatures may lead to potential fitness consequences in sea turtles resulting from altered seasonality and duration of nesting (Pike et al. 2006). Sea turtles may also expand their range as temperature-dependent distribution limits change (McMahon and Hays 2006; Poloczanska et al. 2009a). Further, sea turtles have temperature-dependent sex determination, and many populations produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Jensen et al. 2018; Newson et al. 2009; Patrício 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). Lastly, studies examining the spatio-temporal coincidence of marine turtle nesting with hurricanes, cyclones and storms suggest that cyclical loss of nesting beaches, decreased hatching success and hatchling emergence success could occur with greater frequency in the future due to global climate change (Hawkes et al. 2009). Environmental changes associated with climate change are occurring within the *Action Area* and are expected to continue into the future. Marine populations that are already at risk due to other threats are particularly vulnerable to the direct and indirect effects of climate change.

As described in the *Exposure Analysis*, hooking and entanglement are the most significant stressors that green sea turtles are exposed to directly from the action. Green sea turtles interact with the HI SSLL fishery in low numbers. In 14 years, 10 green sea turtles have been observed in this fishery. All ten of these were recorded with sufficient information to evaluate post interaction mortality risk using the criteria of Ryder et al. (2006). All ten green sea turtles caught in the HI SSLL fishery between 2004 and 2018 were released alive with no gear attached.

As previously discussed in the *Status of Listed Resources*; since 1996, NMFS has evaluated the mtDNA of 19 green sea turtles captured in the SSLL fishery. The majority of the turtles caught in the HI SSLL are predominantly comprised of animals from the Eastern Pacific DPS and the Central North Pacific DPS (Dutton, pers. Comm. 17 December 2018). An estimate of the catch composition to date, using Bayesian mixed stock analysis suggests that a small number (<2%) of the green sea turtles may comprise animals from the East Indian - West Pacific DPS, Central North Pacific DPS, Southwest Pacific DPS, and the Central South Pacific DPS. To date, no animals from these additional four DPS's have been captured. However, the potential exists based on the movement of juvenile green sea turtles among these DPS within the Pacific (see Seminoff et al. 2015 for a discussion of genetics and tagging studies evaluated in determining DPS discreteness). NMFS estimates (Table 102) that between 0-5% of the turtles caught in the shallow-set longline fishery could be from the Central South Pacific DPS (P. Dutton pers. comm. November 20, 2017).

Table 103. Genetic composition of green turtles sampled from the Hawaii shallow-set longline fishery (Dutton pers. comm. August 31, 2016). The number of genetic samples column is from the direct approach and the MSA% is the mixed stock analysis with the 95% confidence interval.

DPS (ID #)	Number of genetic samples	MSA mean %	95% CI
Central South Pacific (9)	0	1	0-5

Therefore, since we cannot determine the percentages of turtles from each DPS with great certainty as evidenced from the large CI, we will use the upper 95% CI to analyze the potential impacts to each DPS. Based on the 10 green sea turtles that were caught by the SSLL fishery between 2004-2018 as described in the *Effects Analysis* section, NMFS predicts that the HI SSLL fishery will interact with up to 5 green sea turtles per year, (or up to 10 over three years with one mortality). Therefore, the action is expected to reduce the abundance of individuals in a population, which may consequently affect the population's viability. Up to one mortality over a 3 year period, plus the loss of their potential offspring, from a DPS that has over 3,000 nesting females, is a small proportion of the overall population and not expected to reduce the fitness of the species. We estimate that up to10 individuals from the Central South Pacific DPS could be hooked or entangled which could result in one mortality over a 3 year period, which accounts for 0.03 % of the nesting females in this DPS.

Based on the evidence available, we conclude that the incidental take and resulting mortality of green sea turtles from the Central South Pacific DPS associated with the direct and indirect effects of NMFS' continued authorization of the HI SSLL fishery would not be expected to appreciably reduce, the endangered Central South Pacific DPS green sea turtle's likelihood of surviving and recovering in the wild. We expect the overall population to remain large enough to maintain genetic heterogeneity, broad demographic representation, and successful reproduction. The proposed action will have a small effect on the overall size of the population, and we do not expect it to affect the Central South Pacific green sea turtle DPS' ability to meet their lifecycle requirements and to retain the potential for recovery.

7.3.4 Central West Pacific DPS

As described in the *Status of Listed Resources*, the green sea turtle Central West Pacific DPS is listed as endangered there is insufficient long-term and standardized monitoring information to adequately describe abundance and population trends for many areas of the Central West Pacific DPS. The limited available information suggests a nesting population decrease in some portions of the DPS 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 a number of unquantified nesting sites, possibly with small numbers, however specifics regarding these sites is unknown. The largest nesting site is in the FSM, and that particular site hosts approximately 22 percent of the total annual nesting females for this DPS (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 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. (2008) reports an estimated annual population growth rate of 6.8% per year for the Chichijima nesting site.

Central West Pacific DPS 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 throughout the DPS (Seminoff et al. 2015) Sharks and other fish will also eat the hatchlings when they enter the marine environment and adult turtles are also killed by sharks and other marine predators (Seminoff et al. 2015). Fibropapillomatosis is the most commonly identified disease in green turtles. This disease results in internal and/or external tumors (fibropapillomas) that may grow large enough to hamper swimming, vision, feeding, and potential escape from predators.

Directed take 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; Guilbeaux 2001; Hitipeuw and Maturbongs 2002; Philip 2002; Humber et al. 2014). In addition to the collection of eggs from nesting beaches, the killing of nesting females continues to threaten the stability of green turtle populations. Other anthropogenic threats include incidental capture in artisanal and commercial fisheries.

As described in the *Environmental Baseline*, past and present fisheries interactions may have been, and may continue to be, a threat to Central West Pacific DPS 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 U.S. in recent years, large numbers of Central West Pacific DPS green sea turtles may still be captured in international and U.S. commercial fisheries that target other species. These activities are reasonably likely to continue, and may increase over time due to the effects of increased human population and increased human consumption of fish products.

In addition, warming sea surface temperatures may lead to potential fitness consequences in sea turtles resulting from altered seasonality and duration of nesting (Pike et al. 2006). Sea turtles may also expand their range as temperature-dependent distribution limits change (McMahon and Hays 2006; Poloczanska et al. 2009a). Further, sea turtles have temperature-dependent sex determination, and many populations produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Jensen et al. 2018; Newson et al. 2009; Patrício 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). Lastly, studies examining the spatio-temporal coincidence of marine turtle nesting with hurricanes, cyclones and storms suggest that cyclical loss of nesting beaches, decreased hatching success and hatchling emergence success could occur with greater frequency in the future due to global climate change (Hawkes et al. 2009). Environmental changes associated with climate change are occurring within the *Action Area* and are expected to continue into the future. Marine populations that are already at risk due to other threats are particularly vulnerable to the direct and indirect effects of climate change.

As described in the *Exposure Analysis*, hooking and entanglement are the most significant stressors that green sea turtles are exposed to directly from the action. Green sea turtles interact with the HI SSLL fishery in low numbers. In 14 years, 10 green sea turtles have been observed in this fishery. All ten of these were recorded with sufficient information to evaluate post interaction mortality risk using the criteria of Ryder et al. (2006). All ten green sea turtles caught in the HI SSLL fishery between 2004 and 2018 were released alive with no gear attached.

As previously discussed in the *Status of Listed Resources*; since 1996, NMFS has evaluated the mtDNA of 19 green sea turtles captured in the SSLL fishery. The majority of the turtles caught in the HI SSLL are predominantly comprised of animals from the Eastern Pacific DPS and the Central North Pacific DPS (Dutton, pers. Comm. 17 December 2018). An estimate of the catch composition to date, using Bayesian mixed stock analysis suggests that a small number (<2%) of the green sea turtles may comprise animals from the East Indian - West Pacific DPS, Central West Pacific DPS, Southwest Pacific DPS, and the Central South Pacific DPS. To date, no animals from these additional four DPS's have been captured. However, the potential exists based on the movement of juvenile green sea turtles among these DPS within the Pacific (see Seminoff et al. 2015 for a discussion of genetics and tagging studies evaluated in determining DPS discreteness). NMFS estimates (Table 103) that between 0-8% of the turtles caught in the shallow-set longline fishery could be from the Central West Pacific DPS (P. Dutton pers. comm. November 20, 2017).

Table 104. Genetic composition of green turtles sampled from the Hawaii shallow-set longline fishery (Dutton pers. comm. August 31, 2016). The number of genetic samples column is from the direct approach and the MSA% is the mixed stock analysis with the 95% confidence interval.

DPS (ID #)	Number of genetic samples	MSA mean %	95% CI
Central West Pacific (7)	0	1	0-8

Therefore, since we cannot determine the percentages of turtles from each DPS with great certainty as evidenced from the large CI, we will use the upper 95% CI to analyze the potential impacts to each DPS. Based on the 10 green sea turtles that were caught by the SSLL fishery between 2004-2018 as described in the *Effects Analysis* section, NMFS predicts that the HI SSLL fishery will interact with up to 5 green sea turtles per year, (or up to 10 over three years with one mortality). Therefore, the action is expected to reduce the abundance of individuals in a population, which may consequently affect the population's viability. Up to one mortality over a

3 year period, plus the loss of their potential offspring, from a DPS that has over 6,518 nesting females, is a small proportion of the overall population and not expected to reduce the fitness of the species. We estimate that up to10 individuals from the Central West Pacific DPS could be hooked or entangled which could result in one mortality over a 3 year period, which accounts for 0.02 % of the nesting females in this DPS.

Based on the evidence available, we conclude that the incidental take and resulting mortality of green sea turtles from the Central West Pacific DPS associated with the direct and indirect effects of NMFS' continued authorization of the HI SSLL fishery would not be expected to appreciably reduce, the endangered Central West Pacific DPS green sea turtle's likelihood of surviving and recovering in the wild. We expect the overall population to remain large enough to maintain genetic heterogeneity, broad demographic representation, and successful reproduction. The proposed action will have a small effect on the overall size of the population, and we do not expect it to affect the Central West Pacific green sea turtle DPS' ability to meet their lifecycle requirements and to retain the potential for recovery.

7.3.5 Southwest Pacific DPS

As described in the *Status of Listed Resources*, the green sea turtle Southwest Pacific DPS is listed as threatened and nesting occurs in many islands throughout the range of the DPS, but there are only two nesting areas (Raine Island and Heron Island, described in more detail below) with long-term (>15 years) annual indices of nesting abundance. The Raine Island, Australia index count (1994–2004, intermittent) has high inter-annual variability and a slightly increasing linear trend. Heron Island, Australia, index count (1967– 2004, intermittent) also has high inter-annual variability and a slightly increasing linear trend. Although long robust time series are not available for New Caledonia, recent and historic accounts do not suggest a significant decline in abundance of green turtles nesting in New Caledonia (Maison et al. 2010). The trend at Vanuatu has not been documented (Maison et al. 2010). The Raine Island (nGBR) nesting index is the mean number of females ashore for nesting (during the first 2 weeks of December) that are counted during one survey of the nesting habitat per night (Limpus 2009). The number of nesters observed on nightly tally counts was relatively low from 1975 through the early 1980s, then had higher peaks starting in 1984 (Limpus 2009). From the mid-1990s to the mid-2000s, there has been a leveling off of the rate of increase (Chaloupka et al. 2008). The Heron Island, Australia, index count is derived from a tagging census of the total annual nesting population. There was a 3% per year increase in annual nesting abundance in the subset of data from 1974–1998 (Chaloupka and Limpus 2001) and a similar 3.8% per year increase from the 1974–2002 subset (Chaloupka et al. 2008). When including all years from 1967–2004 there is an increasing linear trend in the annual nesting population size, but the relationship was not significant (Limpus 2009). The increase in annual nesting females at Heron Island is concurrent with an estimated increase of 11% per year from 1985–1992 for the green turtle foraging population (immature and mature females and males) in Heron Reef/Wistari Reef complex (Chaloupka and Limpus 2001). The number of turtles nesting in the GBR area of Australia differs widely from year to year and is well correlated with an index of the Southern Oscillation (Limpus and Nicholls 2000). For example, the estimate of annual nesters at Raine Island during a medium density nesting season is about 25,000 (Limpus 2009), while in a high density season (1999–2000) the estimate of nesters at Raine Island increases to 78,672 ± 10,586. Heron Island is the index nesting beach for the sGBR, and nearly every nesting female on Heron Island has been tagged since 1974 (Limpus and Nicholls 2000). The mean annual nester abundance varied between 26 and 1,801 during 1999–2004 (Limpus 2009).

Southwest Pacific green turtles are exposed to a variety of natural threats both at their nesting beaches and in the open ocean. Primary hatchling and egg predators include crabs, birds, fish, and mammals. In Vanuatu, nest predation by feral dogs is a primary threat (Maison et al. 2010). Survivorship of hatchlings in southern Great Barrier Reef during the transition from nest to sea (accounting for crab and bird predation) may be quite high (0.98) (Limpus 1971), but survivorship of hatchlings as they (0.4) (Gyuris 1994 as cited in Limpus 2009) transition across the reef flat from the water's edge to deep water is likely considerably lower. Similar survivorship estimates are not available for the northern Great Barrier Reef, but survival during the nest to sea transition are expected to be low and variable, depending on the predator assemblage. Although many birds co-occur with sea turtle hatchlings in the northern Great Barrier Reef, only some birds like the rufous night heron are important predators (Limpus et al. 2003). Terrestrial crabs which occur throughout the northern Great Barrier Reef have been observed feeding on turtle hatchlings and eggs, but the crabs are generally of low density (Limpus et al. 2003). Shark predation on hatchlings as well as adults has been documented (Limpus et al. 2003).

The potential effects of diseases and endoparasites, as described for other DPSs, also exist for green turtles found in the Southwest Pacific DPS. Low levels of fibropapilloma-associated herpesvirus are common in green turtles in some but not all semi-enclosed waters like Moreton Bay and Repulse Bay in Australia, more infrequent in nearshore open waters and rare in offshore coral reef habitats (Limpus 2009). Other health conditions such as coccidiosis, parasites, and fungal infections also occur (Limpus, 2009).

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

As described in the *Environmental Baseline*, past and present fisheries interactions may have been, and may continue to be, a threat to Southwest Pacific DPS 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 U.S. in recent years, large numbers of Southwest Pacific DPS green sea turtles may still be captured in international and U.S. commercial fisheries that target other species. These activities are reasonably likely to continue, and may increase over time due to the effects of increased human population and increased human consumption of fish products.

In addition, warming sea surface temperatures may lead to potential fitness consequences in sea turtles resulting from altered seasonality and duration of nesting (Pike et al. 2006). Sea turtles may also expand their range as temperature-dependent distribution limits change (McMahon and Hays 2006; Poloczanska et al. 2009a). Further, sea turtles have temperature-dependent sex determination, and many populations produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Jensen et al. 2018; Newson et al. 2009; Patrício 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). Lastly, studies examining the spatio-temporal coincidence of marine turtle nesting with hurricanes, cyclones and storms suggest that cyclical loss of nesting beaches, decreased hatching success and hatchling emergence success could occur with greater frequency in the future due to global climate change (Hawkes et al. 2009). Environmental changes associated with climate change are occurring within the *Action Area* and are expected to continue into the future. Marine populations that are already at risk due to other threats are particularly vulnerable to the direct and indirect effects of climate change.

As described in the *Exposure Analysis*, hooking and entanglement are the most significant stressors that green sea turtles are exposed to directly from the action. Green sea turtles interact with the HI SSLL fishery in low numbers. In 14 years, 10 green sea turtles have been observed in this fishery. All ten of these were recorded with sufficient information to evaluate post interaction mortality risk using the criteria of Ryder et al. (2006). All ten green sea turtles caught in the HI SSLL fishery between 2004 and 2018 were released alive with no gear attached.

As previously discussed in the *Status of Listed Resources*; since 1996, NMFS has evaluated the mtDNA of 19 green sea turtles captured in the SSLL fishery. The majority of the turtles caught in the HI SSLL are predominantly comprised of animals from the Eastern Pacific DPS and the Central North Pacific DPS (Dutton, pers. Comm. 17 December 2018). An estimate of the catch composition to date, using Bayesian mixed stock analysis suggests that a small number (<2%) of the green sea turtles may comprise animals from the East Indian - West Pacific DPS, Central North Pacific DPS, Southwest Pacific DPS, and the Central South Pacific DPS. To date, no animals from these additional four DPS's have been captured. However, the potential exists based on the movement of juvenile green sea turtles among these DPS within the Pacific (see Seminoff et al. 2015 for a discussion of genetics and tagging studies evaluated in determining DPS discreteness). NMFS estimates (Table 104) that between 0-8% of the turtles caught in the shallow-set longline fishery could be from the Southwest Pacific DPS (P. Dutton pers. comm. November 20, 2017).

Table 105. Genetic composition of green turtles sampled from the Hawaii shallow-set longline fishery (Dutton pers. comm. August 31, 2016). The number of genetic samples column is from the direct approach and the MSA% is the mixed stock analysis with the 95% confidence interval.

DPS (ID #)	Number of genetic samples	MSA mean %	95% CI
Southwest Pacific (8)	0	1	0-8

Therefore, since we cannot determine the percentages of turtles from each DPS with great certainty as evidenced from the large CI, we will use the upper 95% CI to analyze the potential impacts to each DPS. Based on the 10 green sea turtles that were caught by the SSLL fishery between 2004-2018 as described in the *Effects Analysis* section, NMFS predicts that the HI SSLL fishery will interact with up to 5 green sea turtles per year, (or up to 10 over a three-year period). From these we expect that up to 1 green sea turtle could die over a three-year period. Therefore, the action is expected to reduce the abundance of individuals in a population, which may consequently affect the population's viability. Up to one mortality over a 3 year period, plus the loss of their potential offspring, from a DPS with a nester abundance of 83,058 females is a small proportion of the overall population and not expected to reduce the fitness of the species. We estimate that up to10 individuals from the Southwest Pacific DPS could be hooked or entangled which could result in one mortality over a 3 year period, which accounts for 0.001 % of the nesting females in this DPS.

Based on the evidence available, we conclude that the incidental take and resulting mortality of green sea turtles from the Southwest Pacific DPS associated with the direct and indirect effects of NMFS' continued authorization of the HI SSLL fishery would not be expected to appreciably reduce, the threatened Southwest Pacific DPS green sea turtle's likelihood of surviving and recovering in the wild. We expect the overall population to remain large enough to maintain genetic heterogeneity, broad demographic representation, and successful reproduction. The proposed action will have a small effect on the overall size of the population, and we do not expect it to affect the Southwest Pacific green sea turtle DPS' ability to meet their lifecycle requirements and to retain the potential for recovery.

7.3.6 East Indian/West Pacific DPS

As described in the *Status of Listed Species*, the green sea turtle East Indian/West Pacific DPS is listed as threatened and there are four sites for which 15 or more years of recent data are available for annual nester abundance: Sabah Turtle Islands in Malaysia; Royal Navy Center in Khram Island, Thailand; Redang in Terrengganu, Myanmar; and Thameela Islands, Myanmar. Only Sabah Turtle Islands represent a sizable nesting population, estimated at 7,011 in 2011 with a sex ratio of 1M: 4F (Pilcher 2010). The PVA indicates that the nesters from Sabah Turtle Islands in Malaysia, with an estimated 7,000 nesters, will likely continue to increase, while the nesters from the Royal Navy Center in Khram Island, Thailand (estimated 297 nesters), Redang in Terrengganu, Myanmar (estimated 278 nesters), and Thameela Islands, Myanmar (estimated 109 nesters) will likely continue to decline (Seminoff et al. 2015). The total abundance for this population is estimated at 77,009 nesters (Seminoff et al 2015) or 96,261 adults with a 1M:4F sex ratio.

For the Sabah Turtle Islands in Malaysia, the probability that this population will fall below the trend reference point (50% decline) at the end of 100 years approaches zero. The probability that this population falls below the absolute abundance reference (100 females per year) at the end of 100 also approaches zero. This trend may be a result of effective conservation measures made by the Sabah Government in the 1970s when the Sabah Turtle Islands were acquired from private ownership to provide complete protection to the nesting turtles and their eggs (Chan 2006).

Annual nesting in the Khram Island, Sea Turtle Conservation Center of the Royal Thai Navy, Gulf of Thailand has decreased from a mean of approximately 405 nests per year between 1975–1983 to a mean of approximately 250 nests per year from 1992–2001 (Charuchinda and

Monanunsap 1998; Charuchinda et al. 2002). For these beaches, there is a nearly 100% probability that this population will fall below the trend reference point (50% decline) within 100 years. There is also a nearly 100% probability that this population falls below the absolute abundance reference (100 females per year) at the end of 100 years. For Redang in Terengganu, Malaysia, there is a 72.9% probability that this population will fall below the trend reference point (50% decline) at the end of 100 years. There is an 89.8% probability that this population falls below the absolute abundance reference (100 females per year) at the end of 100 years. For Thameela in Myanmar, there is an 87.9% probability that this population will fall below the trend reference point (50% decline) at the end of 100 years. There is a 96.7% probability that this population falls below the absolute abundance reference (100 females per year) at the end of 100 years (Seminoff et al. 2015).

East Indian-West Pacific DPS green sea turtles are exposed to a variety of natural threats both at their nesting beaches and in the open ocean. The best available data suggest that current nest and hatchling predation on the East Indian-West Pacific DPS is prevalent. Depredation of nests by feral animals is also widespread in many South Asian areas (Sunderraj et al. 2001; Islam 2002). Nest predation by feral pigs and dogs is a major threat on the Andaman and Nicobar Islands of India (Fatima et al. 2011). Jackals, foxes, wild boars, and monitor lizards also predate green turtle nests and hatchlings along the beaches of Bangladesh (Andrews et al. 2006a). Lizards and ghost crabs are the natural predators of green turtle nests in several areas (Chantrapornsyl 1993). In Malaysia, ghost crabs predate green turtle eggs (Ali and Ibrahim, 2000), and gold-ringed cat snakes or mangrove snakes, (Asiatic) reticulated pythons, monitor lizards, and house mice predate hatchlings (Hendrickson, 1958). Monitor lizards, crabs, and ants predate eggs and hatchlings on the beaches of Vietnam (as cited in "Sea Turtle Migration-Tracking and Coastal Habitat Education Program– An Educator's Guide" http://www.ioseaturtles.org/Education/seaturtlebooklet.pdf). In Japan, raccoon dogs and weasels are a threat to nests (Kamezaki et al. 2003). In Taiwan, snakes predate the nests (Cheng et al. 2009). Hendrickson (1958) estimated that 4% of the adult females on Malaysian beaches showed signs of assumed shark attack-amputated flippers and missing shell. It has been speculated that sharks congregate in large numbers around the Sarawak Turtle Islands during the peak breeding season (Hirth 1997). On the North West Cape and the beaches of the Ningaloo coast of mainland Australia, a long established feral European red fox population historically preyed heavily on eggs and is thought to be responsible for the lower numbers of nesting turtles on the mainland beaches (Baldwin et al. 2003; Kelliher et al. 2011). During the 2010–2011 nesting season, foxes predated 23% of all the nests laid along the North West Cape and Cape Range Division of the Ningaloo Coast (Kelliher et al. 2011). Fibropapillomatosis has been found in green 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), 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.

Despite substantial declines in green turtle nesting numbers, egg harvest remains legal in several of the countries within this DPS (Humber et al. 2014). Directed take of turtles and their eggs is an ongoing problem in the Central West Pacific in Bangladesh, Thailand, Malaysia, Sabah, West Java, Indonesia, Philippines and Vietnam (Islam et al. 1999 as cited in Islam 2001, 2002; Charuchinda et al. 2002; Chan 2006; Siow and Moll 1982; Schulz 1987; Hirth 1997; Ministry of Fisheries 2003). Commercial harvest (a major threat), as well as subsistence, 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 East Indian-West Pacific DPS 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 U.S. in recent years, large numbers of East Indian-West Pacific DPS green sea turtles may still be captured in international and U.S. commercial fisheries that target other species. These activities are reasonably likely to continue, and may increase over time due to the effects of increased human population and increased human consumption of fish products.

In addition, warming sea surface temperatures may lead to potential fitness consequences in sea turtles resulting from altered seasonality and duration of nesting (Pike et al. 2006). Sea turtles may also expand their range as temperature-dependent distribution limits change (McMahon and Hays 2006; Poloczanska et al. 2009a). Further, sea turtles have temperature-dependent sex determination, and many populations produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Jensen et al. 2018; Newson et al. 2009; Patrício 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). Lastly, studies examining the spatio-temporal coincidence of marine turtle nesting with hurricanes, cyclones and storms suggest that cyclical loss of nesting beaches, decreased hatching success and hatchling emergence success could occur with greater frequency in the future due to global climate change (Hawkes et al. 2009). Environmental changes associated with climate change are occurring within the *Action Area* and are expected to continue into the future. Marine populations that are already at risk due to other threats are particularly vulnerable to the direct and indirect effects of climate change.

As described in the *Exposure Analysis*, hooking and entanglement are the most significant stressors that green sea turtles are exposed to directly from the action. Green sea turtles interact with the HI SSLL fishery in low numbers. In 14 years, 10 green sea turtles have been observed in this fishery. All ten of these were recorded with sufficient information to evaluate post interaction mortality risk using the criteria of Ryder et al. (2006). All ten green sea turtles caught in the HI SSLL fishery between 2004 and 2018 were released alive with no gear attached.

As previously discussed in the *Status of Listed Resources*; since 1996, NMFS has evaluated the mtDNA of 19 green sea turtles captured in the SSLL fishery. The majority of the turtles caught in the HI SSLL are predominantly comprised of animals from the Eastern Pacific DPS and the Central North Pacific DPS (Dutton, pers. Comm. 17 December 2018). An estimate of the catch composition to date, using Bayesian mixed stock analysis suggests that a small number (<2%) of the green sea turtles may comprise animals from the East Indian - West Pacific DPS, Central North Pacific DPS, Southwest Pacific DPS, and the Central South Pacific DPS. To date, no animals from these additional four DPS's have been captured. However, the potential exists based on the movement of juvenile green sea turtles among these DPS within the Pacific (see

Seminoff et al. 2015 for a discussion of genetics and tagging studies evaluated in determining DPS discreteness). NMFS estimates (Table 105) that between 0-10% of the turtles caught in the shallow-set longline fishery could be from the East Indian-West Pacific DPS (P. Dutton pers. comm. November 20, 2017).

Table 106. Genetic composition of green turtles sampled from the Hawaii shallow-set longline fishery (Dutton pers. comm. August 31, 2016). The number of genetic samples column is from the direct approach and the MSA% is the mixed stock analysis with the 95% confidence interval.

DPS (ID #)	Number of genetic samples	MSA mean %	95% CI
East Indian-West Pacific (6)	0	1	0-10

Therefore, since we cannot determine the percentages of turtles from each DPS with great certainty as evidenced from the large CI, we will use the upper 95% CI to analyze the potential impacts to each DPS. Based on the 10 green sea turtles that were caught by the SSLL fishery between 2004-2018 as described in the *Effects Analysis* section, NMFS predicts that the HI SSLL fishery will interact with up to 5 green sea turtles per year, (or up to 10 over a three-year period). From these we expect that up to 1 green sea turtle could die over a three-year period. If we assume that 10 individuals from this DPS could be hooked or entangled every three years and that one of those could result in a mortality every three years, this represents 0.001% of the adult population based on an abundance of 96,261 adults.

Based on the evidence available, we conclude that the incidental take and resulting mortality of green sea turtles from the East Indian-West Pacific DPS associated with the direct and indirect effects of NMFS' continued authorization of the HI SSLL fishery would not be expected to appreciably reduce, the threatened East Indian-West Pacific DPS green sea turtle's likelihood of surviving and recovering in the wild. We expect the overall population to remain large enough to maintain genetic heterogeneity, broad demographic representation, and successful reproduction. The proposed action will have a small effect on the overall size of the population, and we do not expect it to affect the East Indian-West Pacific green sea turtle DPS' ability to meet their lifecycle requirements and to retain the potential for recovery.

7.4 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. A weighted average of the yearly estimates of olive ridley abundance was 1.39 million (CI: 1.15 to 1.62 million), which is consistent with the increases seen on the eastern Pacific nesting beaches as a result of protection programs that began in the 1990s (Eguchi et al. 2007). Overall, olive ridley numbers are increasing since protections were implemented, but have not returned to historic levels. Large data gaps still exist in this species demography, including age and sex distribution; growth, birth, and death rates; immigration, and emigration (Zug et al. 2006; NMFS and FWS 2007, 2014)

Most olive ridley nesting beach surveys have taken place at arribada beaches where mass emergences in a spatially limited area present challenges to counting turtles directly or counting individual tracks left in the sand. Several methods have been used to estimate the number of turtles nesting during an arribada (Bézy and Valverde 2012; Cornelius and Robinson 1985; Gates et al. 1996; Márquez-M. and Van Dissel 1982; Valverde and Gates 1999) derived from multiple methods at the different arribada beaches and in some cases the method used at a specific arribada beach has changed over the years (e.g, La Escobilla). This renders comparisons among arribada beaches problematic and discerning population trends over time complicated. A further complication is that many nesting population estimates from arribada beaches have been calculated as the sum total of all the turtles nesting during arribadas within a given nesting season. An individual olive ridley may nest on the same beach multiple times during a nesting season and thus the sum total of all the turtles or tracks counted during surveys is not directly equivalent to the number of turtles present in any given nesting population. However, as we displayed in our casual loop diagram in the *Status of Listed Resources*, olive ridleys appear to have an overall positive population trend.

Reviewing threats from the *Status of Listed Resources*, fibropapillomatosis is the only significant disease that has been found in olive ridley sea turtles and does not appear to be a major threat. However, major anthropogenic threats to both threatened and endangered populations include impacts to nesting beaches resulting from development, direct harvest, and fishing bycatch. Fisheries operating near arribadas can take tens of thousands of adults as they congregate for breeding.

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

As described in the *Environmental Baseline*, effects from fisheries bycatch are the most significant threat for the species. There were 762 olive ridley sea turtles reported. Of these, 206 were a result of U.S. 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* section, since the shallow-set fishery re-opened in 2004 there have been a total of 10 observed interactions that have resulted in a hooking or entanglement. All 10 interactions resulted in all gear being removed. Projected interactions were 5, 8, and 11 olive ridleys over 1, 2 and 3 years respectively. In our *Response* section we concluded 0 mortalities across those three years. Based on the distribution of hooking injuries recorded for this species, and the mortality coefficients of Ryder et al. (2006), one mortality would be estimated to occur every 15 interactions. However, our sample size of olive ridley sea turtles is very small and small changes in the number and type of injury (i.e., hook location and depth) recorded by observers with subsequent interactions could significantly affect the predicted number of mortalities.

The low number of interactions expected from the Hawaii SSLL fishery when compared to the population numbers of the species leads us to conclude that these interactions will not have an appreciable effect of populations of olive ridley sea turtles in the *Action Area.* NMFS therefore does not expect the risks that the Hawaii SSLL fishery poses to individual olive ridley sea turtles to have a significant adverse consequences to the populations they represent. As a result, NMFS' continued authorization of the HI SSLL fishery would not be expected to appreciably reduce, directly or indirectly, the endangered or the threatened olive ridley sea turtle's likelihood of surviving and recovering in the wild.

7.5 Oceanic Whitetip Shark

Oceanic whitetip sharks listed as threatened throughout their range. They are exposed to fishing activities throughout the *Action Area*. As discussed in the *Status of Listed Species,* only one stock assessment has been completed to date, estimating the population at 200,000 and only pertains to the Western Pacific. Stock assessments have not been conducted for either the Eastern Pacific or for the global population. 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). Although Young et al. (2016) believes CPUE may have stabilized at a depressed state in the Pacific. The significant declining trends observed in all available abundance indices (e.g. standardized CPUE, biomass, and median size) of oceanic whitetips occurred as a result of increased fishing effort in the longline fisheries, with lesser impacts from targeted longline fishing and purse-seining.

The most significant threat to the species are cumulative impacts from fisheries bycatch and exploitation for the fin trade. Bycatch-related mortality in longline fisheries, are considered the primary drivers for these declines (Clarke et al. 2011; Rice and Harley 2012; Young et al. 2016), 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 driver of mortality of this species in commercial fisheries throughout its global range. As a result of this, oceanic whitetip biomass has declined by 86% since 1995 (Rice and Harley 2012; Young et al. 2016). Currently, the population is overfished and overfishing is still occurring throughout much of the species range. 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 Species*).

As described in the *Environmental Baseline*, effects from international and U.S. 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. (2016) believe they are likely to be minimal.

Clear spatio-temporal trends to the north of the MHI and NWHI between April and July were apparent when analyzing interactions with the shallow set fishery as depicted in the *Exposure* analysis. Comparing these data to aggregated fishing effort shows that sharks are being caught in a specific area during this time, despite that fishing effort is widely distributed throughout the *Action Area*. We also note that some areas with historical interactions within the expanded Monument boundaries are subject to a commercial fishing prohibition. Nevertheless, based on the available evidence, we conclude that fishing activities of the HI SSLL fishery are likely to adversely affect the population dynamics, behavioral ecology, or social dynamics of oceanic whitetip sharks through the loss of individuals. Furthermore, interactions where sharks survive the encounter, are likely to adversely affect an individual's fitness after they are released. Physiological responses and effects to sharks from stress associated with longline capture have been extensively studied. These studies reveal adverse reactions to an individual 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 data for the oceanic whitetip shark are lacking. Specifically, metrics pertaining to delayed mortality after sharks are released. Whether those studies are applicable to oceanic whitetips is not certain because some species may not be physiologically similar in resilience. However, the literature reveals that sharks generally respond adversely to capture.

At this time, post release mortality statistics are not available for the oceanic whitetip shark and have been identified by multiple sources as a significant data gap. The literature also suggests that hooking location is a predictor of fate, with internal hooking having increased deleterious effects. Some sharks that interact with the fishing gear may have had prior interactions with the fishery and may alter their foraging dynamics to avoid capture, while other individuals may continue to depredate bait or catch, which may result in additional hookings.

Given the number of interactions (875) with oceanic whitetip sharks in the HI SSLL fishery from 2004-2018, as described in the *Effects Analysis*, NMFS predicts future interaction levels, using the adjusted upper estimate to be 102, 166, and 299 for annual, 2 year, and 3 year interactions, respectively. Of these, we expect the at-vessel mortality rate of 12% rate to occur, with a post release mortality ranging from 6 to 19%. The total mortality therefore ranges between 18% and 31%. From these we expect that up to 32 oceanic whitetip sharks could die in any one year period (or up to 92 over a three-year period).

Therefore, the action is expected to reduce the abundance of individuals in the population, which may consequently affect the population's viability. However, while we expect a reduction in individual's fitness, when we take into account the number of expected interactions with the HI SSLL fishery, the number of sharks estimated to be present within the Western Pacific (200,000 individuals), and the uncertainty regarding population abundance estimates in the Eastern Pacific and globally; we conclude that the number of sharks this fishery interacts with would not be expected to appreciably reduce the oceanic whitetip shark's likelihood of survival and recovery. Thirty-two mortalities in the first year divided by a minimum population estimate of 200,000 individuals gives us an estimate of 0.016% of the population would be killed by the Hawaii SSLL fishery. This number represents the maximum number of individual oceanic whitetip sharks that we would expect to die in a single year in the HI SSLL fishery. Quantification of the species' abundance through other portions of its range would further reduce our estimate of the proportion of the species that this represents by creating a larger denominator in the equation (that is the population size would be larger than we are assuming it is here). We know the species is present in the other ocean basins due to continued harvest of the species by various countries fishing those waters as reviewed in the *Status of Listed Species*, however we do not have reliable data to quantify the total population abundance at this time. Therefore, the incidental take and resulting mortality of oceanic whitetip sharks associated with the direct and indirect effects of NMFS' continued authorization of the HI SSLL fishery is not likely to reduce the viability of the oceanic whitetip sharks. In conclusion, the continued authorization of the HI SSLL fishery would not be expected to appreciably reduce both the likelihood of the species surviving and recovering in the wild.

7.6 Giant Manta Ray

As described in the *Status of Listed Species*, the giant manta is listed as threatened throughout its range and there are no current and accurate abundance estimates available, as the species tends to be only sporadically observed. Most estimates of subpopulations are based on anecdotal diver or fisherman observations, which are subject to bias. These populations seem to potentially range from around 100-1,500 individuals. In regions where giant manta rays are (or were) actively targeted or caught as bycatch, such as the Philippines, Mexico, Sri Lanka, and Indonesia, populations appear to be decreasing. 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). Overall, in many regions, the status of the population of the giant manta ray appears be declining, up to as much as 80% over the last 75 years, and >30% globally (Marshall et al. 2011a). Conversely, numbers of giant manta rays identified through citizen science in Thailand's waters (primarily on the west coast, off Khao Lak and Koh Lanta) have been increasing over the past few years, from 108 in 2015 to 288 in 2016.

The most significant and certain threat to the giant manta ray is overutilization for commercial purposes and non-targeted bycatch and fishery interactions. Giant manta rays are both targeted fisheries driven by the international trade in gill plates, and caught as bycatch in a number of global fisheries throughout their range. Estimated take of giant manta rays, particularly in many portions of the Indo-Pacific, frequently exceeds numbers of observed individuals in those areas, and are correlated with observed declines in sightings and landings of the species. Giant manta rays' longevity is estimated to be greater than 20-40 years and their age of maturity ranges from 3 to >15 years. Their reproductive periodicity is anywhere from an annual cycle to a 5-year cycle, with a litter of only 1 pup, and their generation interval is estimated to be around 25 years. Given these life history traits, giant manta rays are especially vulnerable to threats that deplete its abundance, and have little potential to withstand high and sustained levels of increased mortality.

As described in the *Environmental Baseline*, effects from international and U.S. fisheries have resulted in interactions with the giant manta ray 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 mobulid gill plates.

In addition, 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 tolerance to both tropical and temperate waters, these animals likely have the ability to shift their range or distribution to remain in an environment conducive to their physiological and ecological needs, providing the species with resilience to these effects.

However, 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. 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 impact 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.

Given the estimated number of interactions (21) with giant manta rays in the HI SSLL fishery from 2004-2018, as described in the *Effects Analysis*, NMFS predicts future interaction levels, using the adjusted upper estimate, to be 13, 18 and 22 for annual, 2 year, and 3 year periods, respectively. As described in the *Response Analysis*, the most significant threat to giant manta rays by the HI SSLL fishery is hooking and entanglement by gear which can injure or kill them. If individual giant manta rays are hooked or entangled but do not immediately die from their wounds, they can suffer impaired swimming or foraging abilities, altered migratory behavior, altered breeding or reproductive patterns, and latent mortality from their interactions.

Giant manta rays tend to be more vulnerable to entanglement and foul hooking as opposed to being hooked in the mouth, and due to their size, they are seldom landed and brought onboard the vessel. Therefore, they tend to be released with at least the hook attached and often with trailing gear. However, data assessing the post-release survivorship for *Mobulidae* spp. and the effects of remaining gear is lacking. The literature suggests there is a similarity between leatherback sea turtles and mobulids in regards to observed hooking location. Since neither of these species are actively predating on the bait on longline gear, but rather attracted to marine life that collects on buoys and buoy lines at or near the surface, they tend to be foul hooked in the flipper and pectoral area as opposed to the mouth or esophagus. Due to these similarities, and the absence of species (or genus) specific mortality rates, we used the leatherback sea turtle as a proxy to assess post-release mortality and considered the Ryder et al. (2006) injury criteria for leatherback sea turtles as a possible proxy for determining post-hooking survival of giant manta rays. As a result of using the leatherback sea turtle mortality coefficients (Ryder et al. 2006), NMFS predicts anticipated mortalities of giant manta rays, based on the predicted exposure values for the adjusted upper estimate, to be 4, 7 and 9 for annual, 2 year, and 3 year periods, respectively. Therefore, the action is expected to reduce the abundance of individuals in the population, which may consequently affect the population's viability. However, while we expect a reduction in individual's fitness, when we take into account the number of expected interactions with the HI SSLL fishery, the minimum number of giant manta rays estimated to be present within the Western Pacific (2,200 individuals), and the uncertainty regarding population abundance estimates; we conclude that the number of giant manta rays this fishery interacts with would not be expected to appreciably reduce the giant manta ray's likelihood of survival and recovery. We estimate that up to 22 giant manta rays could be hooked or entangled which could result in nine mortalities over a 3 year period, which accounts for 0.4 % of the population.

Although, there is no information to indicate that the giant manta ray is composed of conspicuous source-sink populations or habitat patches, and it is unknown whether natural rates of dispersal among populations are too low to prevent sufficient gene flow among populations, given the available evidence, it is our conclusion that the rates of dispersal and gene flow have not been altered.

Based on the evidence available, we conclude that the incidental take and resulting mortality of giant manta rays associated with the direct and indirect effects of NMFS' continued authorization of the HI SSLL fishery would not be expected to appreciably reduce, the threatened giant manta ray's likelihood of surviving and recovering in the wild. We expect the overall population to remain large enough to maintain genetic heterogeneity, broad demographic representation, and successful reproduction. The proposed action will have a small effect on the overall size of the population, and we do not expect it to affect the giant manta ray's ability to meet its lifecycle requirements and to retain the potential for recovery.

7.7 Guadalupe Fur Seal

As described in the *Status of Listed Species*, it has been very difficult to determine the overall population abundance of the threatened Guadalupe fur seal. All the individuals of the population cannot be counted because all age and sex classes are never ashore at the same time, and some individuals that are on land are not visible during the census. The best scientific and commercial data available estimates a population size of approximately 20,000 animals, with ~17,500 at Isla Guadalupe and ~2,500 at Isla San Benito (Garcia-Capitanachi 2011, Aurioles-Gamboa 2015). The estimated population growth is estimated to be approximately 10% (Carretta et al. 2017).

Climate change, along with population growth, is likely causing and outward expansion of the population. Variations in ocean currents, ocean temperatures, with subsequent modifications in food chain dynamics, foraging strategies, interspecific competition, and consequential malnutrition have been documented thus far (Trillmich et al. 1991; Elorriage-Verplancken et al. 2016a, 2016b; Pablo-Rodriguez 2016; NMFS 2017; Juarez-Ruiz et al. 2018). Additionally, pup mortality appears to be high from negative effects caused by El Niño events and tropical cyclones (Fleischer 1987; Trillmich et al. 1991; Gallo-Reynoso 1994; Aurioles-Gamboa 2015). However, both pup and adult abundances drive the overall population trend and while this species faces both natural and anthropogenic threats at all life cycle stages, the number of recruits into the population appears to far out way the loss from the various stressors as we depicted in our casual loop diagram found in the *Status of Listed Resources*. As described in the *Environmental Baseline*, effects from derelict fishing gear of unknown origin and from U.S. fisheries have resulted in interactions with the Guadalupe fur seal. 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 outward expansion of the Guadalupe fur seal to its previous historic range.

We have very limited data on the incidental bycatch of Guadalupe fur seals in pelagic longlines, and even less data on entanglement in longline gear or from other fisheries. To date, the HI SSLL fishery is the only longline fishery known to interact with the species. As previously discussed in the *Status of Listed Resources*, Guadalupe fur seals are experiencing increased population abundance, along with climate change, are expanding their range to compete for resources. This expansion has led to individuals moving into the *Action Area* of the HI SSLL fishery and interacting with this fisehery.

However, Guadalupe fur seals are also known to become entangled in derelict fishing gear (e.g. gillnets). Animals may drag and swim with gear attached for long distances, ultimately resulting in fatigue, compromised feeding ability, or severe injury, which may lead to reduced reproductive success and death. As discussed in the *Exposure* section, most observed interactions with large amounts of gear attached the animal result in SI determinations.

Also, as described in the *Exposure* section, the adjusted upper exposure estimate results in 11, 14, and 17 individuals to interact with the HI SSLL fishery for the first, second, and third years respectively. We used a mortality rate of 80% in our analyses, recognizing this is likely the maximum number of individual Guadalupe fur seals that would die from interactions with the HI SSLL fishery based on the NSI/SI determinations to date. However we also cautioned that each interaction is a unique situation and multiple variables can affect a NSI/SI determination. We determined in the first year, of the 11 interactions that are expected to occur with the fishery, 9 of those animals will die. The second year is expected to result in 12 combined mortalities, and 14 mortalities are expected over three years.

Therefore, the action is expected to reduce the abundance of the population by removing individuals and consequently affecting the population's viability. However, while we expect a reduction in fitness of individual animals, given the size of the entire population of the species (20,000 individuals), the expected population growth rate (10% annually), and the low number of estimated interactions and mortalities over three years (17 and 14 respectively) from the action, we've concluded that while the risk to individuals may be significant, the resulting mortalities would not appreciably reduce the species' likelihood of survival or recovery. (Interactions: 17/20,000=0.00085*100= 0.055% of the species over three years; Mortalities: 14/20,000=0.0007*100= 0.07% of the species over three years). We also note that the estimated population growth is estimated to be approximately 10% (Carretta et al. 2017) and interactions have been occurring with the fishery since 2014. Therefore, the population abundance has been increasing and continues to increase during a time frame where fishing interactions are known to have occurred, therefore we expect the population will continue to increase despite the projected mortalities from the HI SSLL fishery and the other threats that the species faces.

8 CONCLUSION

After reviewing their current status, the environmental baseline for the action area, the effects of the proposed action and the cumulative effects, it is NMFS' biological opinion that NMFS continued operation of the HI SSLL fishery is not likely to jeopardize the continued existence of the following species under NMFS' jurisdiction:

> Endangered leatherback sea turtles, endangered loggerhead sea turtles (North Pacific DPS), threatened green sea turtles (Eastern Pacific, Central North Pacific, East Indian-West Pacific, Southwest Pacific DPS), endangered green sea turtles (Central West Pacific and Central South Pacific DPS), threatened olive ridley sea turtles and olive ridley sea turtles from the endangered Mexico breeding population, threatened oceanic whitetip sharks, threatened manta rays, and threatened Guadalupe fur seals.

9 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 nondiscretionary, and must be undertaken by NMFS for the exemption in section 7(o)(2) to apply. NMFS has a continuing duty to regulate the activity covered by this 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 leatherback sea turtles, endangered North Pacific DPS of loggerhead sea turtles, Eastern Pacific DPS of green sea turtles, Central North Pacific DPS of green sea turtles, East Indian-West Pacific DPS of green sea turtles, Central West Pacific DPS of green sea turtles, Southwest Pacific DPS of green sea turtles, Central South Pacific DPS of 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 endangered Guadalupe fur seals. Currently there are no take prohibition for oceanic white tip sharks 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 two species. However, consistent with the decision in *Center for Biological Diversity v. Salazar*, 695 F.3d 893 (9th Cir. 2012), we have included an 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.

9.1 MMPA Authorization

A marine mammal species or population stock that is listed as threatened or endangered under the ESA is, by definition, also considered depleted under the MMPA. The ESA allows takings of threatened and endangered marine mammals only if authorized by section 101(a)(5) of the MMPA. Section 101(a)(5)(E) of the Marine Mammal Protection Act (MMPA), 16 U.S.C. 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 U.S.C. 1531 *et seq.,* by U.S. vessels and those vessels which have valid fishing permits issued by the Secretary in accordance with section 204(b) of the Magnuson-Stevens Fishery Conservation and Management Act, 16 U.S.C. 1824(b), for a period of up to three years. NMFS may issue the authorization to take 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.

Further, 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). Accordingly, the terms of this ITS and the exemption from Section 9 of the ESA become effective only upon the issuance of MMPA authorization to take the marine mammals identified here. Absent such authorization, this ITS is inoperative for ESA-listed marine mammals.

Where an endangered or threatened marine mammal species is involved, section 7(b)(4) of the ESA requires that any incidental take be authorized pursuant to section 101(a)(5) of the MMPA in order to also provide take exemption under the ESA. Thus, to the extent this incidental take statement (ITS) addresses marine mammal species, it is prospective, and will only become operative once the taking is authorized pursuant to the MMPA. Specifically, the proposed action will require a three-year authorization under MMPA section 101(a)(5)(E) and therefore the marine mammal components of this ITS are not operative unless and until that annual MMPA authorization is in place.

9.2 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 indivdual 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 (USFWS and NMFS 1998). If, during the course of the action, this level of incidental take is exceeded for any of the species as listed, NMFS SFD must immediately reinitiate formal consultation with NMFS PRD pursuant to the section 7 regulations (50 CFR 402.16). NMFS PRD anticipates that the following species could be taken as a result of the proposed action by capturing, harming, wounding and killing:

The annual numbers of interactions and mortalities predictied to result from the implementation of the proposed action are shown in Table 107 below.

Table 107. The number of sea turtle, Guadalupe fur seal, oceanic whitetip shark, and giant manta ray interactions expected from the proposed action during one calendar year. The table also includes total mortalities (males and females, adults and juveniles) expected to result from this number of interactions.

Species	**Annual**	
	Number Captured	**Number Killed**
Leatherback sea turtle	21	3
Loggerhead sea turtle	36	6
*Olive ridley sea turtle (all species)	5	0
*Green sea turtle (all species)	5	0
**Oceanic whitetip shark	102	32
**Giant manta ray	13	4
Guadalupe fur seal	11	9

*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 DPS

**An ITS is not required to provide protective coverage for the Giant manta ray 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 serve as a check on the no-jeopardy conclusion by providing a reinitiation trigger if the level of take analyzed in the biological opinion is exceeded.

9.3 Reasonable and Prudent Measures

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 HI SSLL fishery, as described in the proposed action, on threatened and endangered species and to monitor the level and nature of any incidental takes. These measures are non-discretionary—they must be undertaken by NMFS SFD for the exemption in ESA section 7(o)(2) to apply.

1. To reduce the impact of a small number of vessels causing a large proportion of loggerhead and leatherback interactions and to provide early detection of higher fleet-wide interactions, NMFS SFD shall require individual vessel limits for loggerhead and leatherback sea turtles.
2. NMFS SFD shall develop a minimization measure, or a suite of minimization measures with a goal designed to reduce the incidental capture and associated mortality of leatherback and loggerhead sea turtles by at least 25%.

3. NMFS SFD shall collect data on the capture, injury, and mortality of ESA-listed marine species caught by the shallow-set longline fishery, and shall also collect basic life-history information, as available.
4. NMFS SFD shall require that ESA-listed species incidentally caught alive be released from fishing gear in a manner that minimizes injury and the likelihood of further gear entanglement or entrapment to increase post-release survivorship, to increase post release survivorship, and to the extent consistent with the safety of the vessel and crew, NMFS SFD shall require that ESA-listed species incidentally caught live be released from fishing gear in a manner that minimizes injury and the likelihood of further gear entanglement or entrapment.
5. NMFS SFD shall require that comatose or lethargic sea turtles shall be retained on board, handled, resuscitated, and released according to the established procedures, as practicable and in consideration of best practices for safe vessel and fishing operations.
6. NMFS SFD shall require retention of all sea turtle carcasses for research when turtles are brought on board a vessel and are dead or that do not resuscitate.
7. NMFS SFD shall use temporal and spatial data on oceanic whitetip shark and giant manta ray interactions to inform decision making, and shall modify fishing practices to minimize the respective incidental capture and mortality of oceanic whitetip sharks and giant manta rays to the maximum extent possible, based on this information.

9.3.1 Terms and Conditions

NMFS SFD shall undertake and comply with the following terms and conditions to implement the reasonable and prudent measures identified in Section 13.4 above. These terms and conditions are non-discretionary, and if NMFS SFD fails to adhere to these terms and conditions, or fails to implement measures requiring the Applicant to comply with these terms and conditions, the protective coverage of section 7(o)(2) may lapse.

1. The following terms and conditions implement Reasonable and Prudent Measure No. 1:
    a. NMFS SFD shall initiate rule-making, within 3 months of receiving a signed biological opinion, to set an annual per vessel limit for leatherback and loggerhead sea turtles. Individual vessels shall not be allowed to exceed 2 leatherback sea turtles or 6 loggerhead sea turtles annually.
    b. NMFS SFD shall conduct an analysis of interactions with leatherback and loggerhead sea turtles to evaluate patterns of interactions with these species and vessels and their registered owners. Within 18 months of receiving a signed biological opinion, NMFS SFD shall provide a report on their findings to NMFS PRD with an action plan for working with vessels that interact with a disproportionate number of leatherback and loggerhead sea turtles.
    c. NMFS SFD shall analyze interactions and behavior of any vessel that reaches a per vessel limit, and shall provide this report to PRD on an annual basis.
2. The following terms and conditions implement Reasonable and Prudent Measure No. 2:
    a. NMFS SFD shall develop and implement minimization measures, within two-years of receiving a signed biological opinion, to reduce the incidental capture

and associated mortality of leatherback and loggerhead sea turtles using active minimization measures with a goal to reduce incidental capture and mortality by at least 25%.

NMFS SFD's shall include in the evaluation, at a minimum, the following potential minimization measures: closing the area east 140°W in the first and fourth quarters of the year; prohibit fishing in the dynamic boundary with the sea surface temperature (SST) range of 17°–18.5°C; prohibiting fishing in the dynamic boundary with the SST range of 22.4°–23.4°C, consistent with Howell et al (2015).

Leatherback sea turtles populations have a clear declining trend at about 5% annually. Our PVA suggests that loggerhead sea turtles numbers may be increasing; however, there is sufficient uncertainty in this estimate because the data are from one subpopulation, the variance around our estimate suggest that the species could be declining, and we do not know the trends for the other two primary subpopulations. Consequently, there is a lot of uncertainty on the reproductive trends of the subpopulations. Being that the species is comprised of subpopulations that impart specific benefits to the species as a whole, the reduction of take to these subpopulations is important to the conservation of the species as a whole.

b. NMFS SFD shall, within 3 months of receiving a signed biological opinion, set an annual interaction limit for the fishery 16 leatherback sea turtles, which represents a 25% reduction in the number of turtles from our predicted interaction numbers in this biological opinion. If the fishery reaches this limit then the HI SSLL fishery shall close for the remainder of the calendar year. NMFS may modify this requirement as appropriate upon implementation of minimization measures identified in Term and Condition 2a.

c. NMFS SFD shall set an annual interaction limit of 36 for loggerhead sea turtles. If the fishery reaches this limit then the HI SSLL fishery shall close for the remainder of the calendar year. NMFS may modify this requirement as appropriate upon implementation of minimization measures identified in Term and Condition 2a.

d. NMFS SFD shall conduct an interactive workshop that includes fishermen, observers, and leatherback turtle experts and NMFS PRD to determine whether there are more effective methods for removing more fishing gear from leatherbacks to increase their chance of survival after interacting with longline gear. After the initial workshop, this workshop should be repeated as necessary and findings should be incorporated into the annual Protected Species Workshops given by NMFS SFD. As part of this workshop, NMFS SFD shall conduct a retrospective analysis of vessels data to evaluate vessel success rate and patterns of gear removal from leatherback sea turtles and report their findings at the workshop.

WPFMC estimated that mortalities for leatherback turtles could have been reduced by ~20% from 2004–2018 by reducing the amount of trailing gear (Ishizaki pers. comm. 2019). Identifying and incorporating such measures through the workshop may help to reduce the adverse effects of fishery interactions with leatherback turtles.

   e. NMFS shall conduct survivability studies to better understand post-interaction mortality of leatherback sea turtles, and research modifications to gear and handling techniques that reduce interactions and delayed mortality rates with leatherback sea turtles.

   f. On at least an annual basis, NMFS SFD shall evaluate the effectiveness of adopted measures and report findings to NMFS PRD.

3. The following terms and conditions implement Reasonable and Prudent Measure No. 3:

   a. NMFS SFD shall maintain observer coverage at rates that have been determined to be statistically reliable for estimating protected species interaction rates onboard Hawaii-based shallow-set longline vessels.

   b. In consideration of best practices for safe vessel and fishing operations, observers shall collect standardized information regarding the incidental capture, injury, and mortality of ESA-listed marine species for each interaction by species, gear, and set information, as well as the presence or absence of tags on these species. Observers shall place tags on any untagged turtles that are safely brought aboard a vessel. Observers shall also collect life-history information on ESA-listed marine species incidentally caught by the shallow-set fishery, including measurements, condition, skin biopsy samples, hooking location, and estimated length of gear left on the animal at release. To the extent practicable, these data are intended to allow NMFS to assign these interactions into the categories developed through NMFS' most current post-hooking mortality guidelines.

   c. NMFS SFD shall disseminate quarterly summaries of the data collected by observers on all ESA-listed marine species. These summaries shall be based on the date the interactions occurred.

4. The following terms and conditions implement Reasonable and Prudent Measure No. 4:

   a. NMFS SFD shall continue to require and conduct protected species workshops for owners, operators of registered vessels for use with Hawaii limited entry longline fishing permits vessels, and include crewmembers of those vessels, to educate them in handling and resuscitation techniques to minimize injury and promote survival of hooked or entangled ESA-listed marine species. The workshops shall include information on ESA-listed marine species biology and ways to avoid and minimize impacts to promote protection and conservation, including disseminating new scientific information and recommended handling measures. This information shall be communicated in a way that maximizes understanding of owners, operators and crewmembers.

   b. NMFS SFD shall report annually to NMFS PRD on protected species workshops success, and make all material available to NMFS PRD for review and feedback each year.

   c. NMFS SFD shall continue to train observers about ESA-listed marine species, and techniques for proper handling, dehooking, and resuscitation, as appropriate.

   d. NMFS SFD shall require that shallow-set longline fishermen remove hooks from ESA-listed marine species as quickly and carefully as possible to avoid injuring or killing the animal, as practicable, and in consideration of best practices for safe vessel and fishing operations. NMFS SFD shall require that each HI SSLL longline vessel carry a line clipper to cut the line as close to the hook as practicable and remove as much line as possible prior to releasing the animal in the event a hook cannot be removed (e.g., the hook is deeply ingested or the animal is too large to bring aboard).
   e. NMFS SFD shall require that each HI SSLL longline vessel carry a dip net to hoist a sea turtle onto the deck to facilitate hook removal. If the vessel is too small to carry a dip net, sea turtles must be eased onto the deck by grasping its carapace or flippers, to facilitate the removal of the hook. Any animal brought on board must not be dropped on to the deck. All requirements should consider practicality and best practices for safe vessel and fishing operations.
   f. NMFS SFD shall require each HI SSLL vessel to carry and use, as appropriate, a wire or bolt cutter that is capable of cutting through a hook that may be imbedded externally, including the head/beak area of the animal.
   g. NMFS SFD shall implement best handling practices for release methods to enhance survivorship for giant manta ray as adopted by Hutchinson et al. 2017, and as new information becomes available.
   h. NMFS SFD shall implement best handling practices for release methods to enhance survivorship for ESA-listed sharks as outlined in WCPFC CMM 2010-07; 2011-04, and as new information becomes available on best handling practices.
5. The following terms and conditions implement Reasonable and Prudent Measure No. 5:
   a. NMFS SFD shall require that HI SSLL vessel operators bring comatose sea turtles aboard and perform resuscitation techniques according to the procedures described at 50 CFR 665 and 50 CFR 223.206, as practicable and in consideration of best practices for safe vessel and fishing operations, except that the observer shall perform resuscitation techniques on comatose sea turtles if the observer is available.
6. The following terms and conditions implement Reasonable and Prudent Measure No. 6:
   a. NMFS SFD shall require that dead sea turtles may not be consumed, sold, landed, offloaded, transshipped, or kept below deck, but must be returned to the ocean after identification, unless NMFS requests the turtle be kept for further study.
7. The following terms and conditions implement Reasonable and Prudent Measure No. 7:
   a. NMFS SFD shall evaluate and adopt minimization measure(s) to reduce the incidental bycatch and increase survivability of oceanic whitetip shark and giant manta rays, such as dehooking devices and removing trailing gear.
   b. NMFS SFD shall coordinate with Western Pacific Fishery Management Council and explore as options for minimizing the incidental bycatch of oceanic whitetip and giant manta rays, such tools as individual vessel limits like move-on limits or

trip-limits, or other similar measures to redirect fishing efforts away from areas where spatial and temporal patterns indicate higher take rates for oceanic whitetip shark and giant manta rays occur.

9.4 Conservation Recommendations

1. NMFS SFD should continue to research modifications to fishing gear (e.g., hook size, hook shape, hook offset, hook appendage, bait type, line type, depth configuration, float configuration, deterrents, decoys, etc.) and ESA-listed species handling methods (dehookers, lifting methods, etc.) to reduce ESA-listed species bycatch and mortality in commercial longline fisheries.
2. NMFS SFD should continue to promote reduction of ESA-listed marine species bycatch in Pacific fisheries by supporting:
    a. The Inter-American Convention for the Protection and Conservation of Sea Turtles;
    b. The Western and Central Pacific Fisheries Commission (WCPFC) sea turtle and elasmobranch conservation and management measures for commercial longline fisheries operating in the western Pacific;
    c. The wide dissemination and implementation of NMFS ESA-listed marine species handling guidelines that increase post-hooking survivorship;
    d. Technical assistance workshops to assist other longlining nations to build capacity for observer programs and implement longline gear and handling measures on commercial vessels operating in the western Pacific;
    e. Continuation of ecological, habitat use, and genetics studies of loggerhead turtles, and other ESA-listed marine species, occurring in nearshore foraging habitats in the Pacific, continue monitoring impacts through stranding programs, and promote mitigation studies for fisheries operating in these waters;
    f. Continue bycatch reduction efforts in the Western Pacific to reduce commercial and artisanal fishery impacts on sea turtles (e.g., mitigation of Japan pound nets and other fisheries operating in the South China and Sulu Sulawesi Seas); and
    g. Begin bycatch reduction efforts in the Western Pacific to reduce commercial and artisanal fishery impacts on ESA-listed elasmobranchs.
3. NMFS should continue to encourage, support and work with regional partners to implement long-term sea turtle conservation and recovery programs at critical nesting, foraging and migratory habitats.
4. NMFS should continue to investigate long term climate variability and its impacts to ESA-listed marine species populations.
5. NMFS SFD should continue to prohibit wire traces or leaders in this fishery in the future, which also conforms to CMM 2014-05. Harley and Pilling (2016) modeled a reduction of 10% in projected mortality when wire traces and shark lines were not used. Therefore, allowing these gear modifications to occur may increase shark mortality within the fishery.

6. NMFS shall promote the use of TurtleWatch (Howell et al 2015) with regional partners

9.5 Reinitiation Notice

This concludes formal consultation on the continued operation of the Hawaii shallow-set longline swordfish fishery. As provided in 50 CFR 402.16, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained or is authorized by law, and if:

1. The amount or extent of incidental take for any species is exceeded;
2. New information reveals effects of the agency action that may affect listed species or critical habitat in a manner or to an extent not considered in this opinion;
3. The agency action is subsequently modified in a manner that may affect listed species or critical habitat to an extent in a way not considered in this opinion; or
4. A new species is listed or critical habitat designated that may be affected by the action.

Literature Cited