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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-2018-10335
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Approved By:	Michael D. Tosatto Regional Administrator, Pacific Islands Region
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1 INTRODUCTION

Section 7(a)(2) of the Endangered Species Act of 1973, as amended (ESA; 16 U.S.C. 1536(a)(2)) requires each federal agency to insure that any action they authorize, fund, or carry out is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of critical habitat of such species. When a federal agency's action "may affect" a listed species or its designated critical habitat, that agency is required to consult formally with the National Marine Fisheries Service (NMFS) or the 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 2018b), NMFS and FWS recovery plans and status reviews for sea turtles (NMFS and FWS 1998a, 1998b, 1998c, 1998d; 2007a, 2007b, 2007c, 2007d; 2011, 2013, 2014, 2019 in prep; Seminoff et al. 2015), and elasmobranchs (Young et al. 2017; 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) and Eastern Pacific scalloped hammerhead sharks (NMFS 2015b).

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 2015a).

On December 27, 2017, the U.S. 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 FWS issued a final rule to list 11 species (distinct population segments (DPS)) of green sea turtles under the ESA (81 FR 20058). This final rule removed the previous range-wide listing and, in its place, listed eight as threatened and three as endangered. Six green sea turtle species occur in the Pacific Ocean and within range of the HI SSLL fishery: the East Indian-West Pacific, Central West Pacific,

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[1] The Humpback Whale global listing was divided into 14 distinct population segments (DPS) September 8, 2016; four DPSs were listed as endangered, one as threatened, and the remaining nine were not warranted (81 FR 62259). The Hawaii humpback whale was one of the nine species that was not warranted.

Southwest Pacific, Central South Pacific, Central North Pacific, and the East Pacific green sea turtles. 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 "species" 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 (83 FR 35062). The designated area encompasses waters from the 45-meter (m) 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 incidental take statement (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 2008a). Longline fishing allows a vessel to distribute effort over a large area to harvest fish that are not concentrated in great numbers. Overall catch rates in relation to the number of hooks are generally low (less than 2%). Longline fishing involves setting (deploying) a mainline horizontally at a preferred depth in the water column using floats spaced at regular intervals. Crewmembers usually attach three to five radio buoys at regular intervals along the mainline so the line may be easily located for retrieving (hauling) the gear and retrieving line segments if the mainline breaks during fishing operations.

Crewmembers clip branch lines to the mainline at regular intervals, and each branch line has a single baited hook. Mainline lengths can be 30 to 100 kilometers (km) (18 to 60 nautical miles (nm)) long. After deploying the mainline, the gear fishes (soaks) for several hours before being hauled. In longlining, a "set" is the deployment and retrieval of a discrete unbroken section of mainline, floats, and branch lines. 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 2005; WPRFMC 2006; Beverly and Chapman 2007; WPRFMC 2009a). Fig. 2 illustrates the difference between shallow-set and deep-set gear configuration.

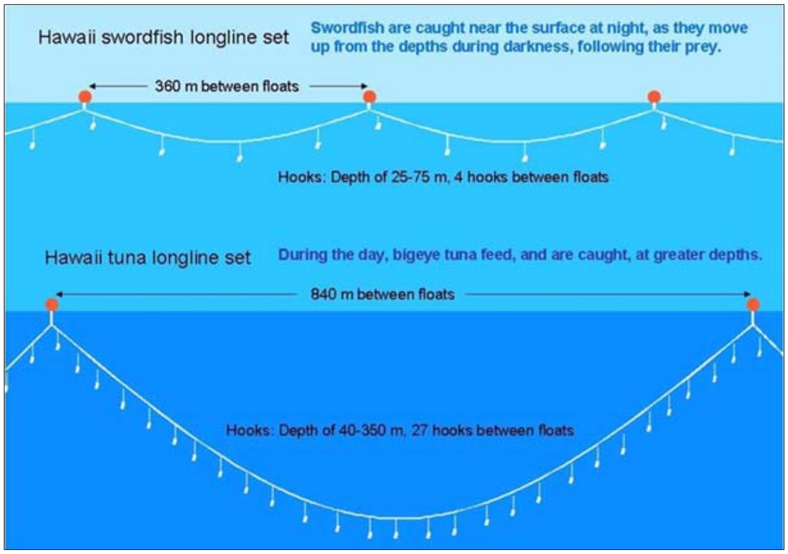


Figure 1. Generalized depiction of shallow-set and deep-set gear configuration

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 knots (kt). Vessel sizes range up to nearly the maximum 100-foot (ft.) 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.

- PapahānaumokuākeaMarine 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 2014d).

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.

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).

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

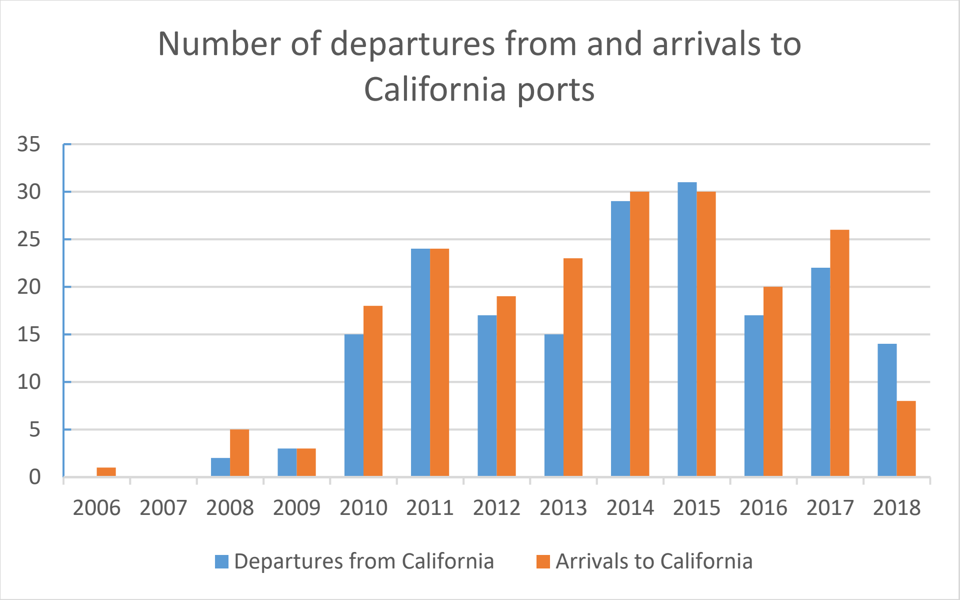


Figure 2. Graph depicting the number of vessel arrivals and departures to and from California ports in each calendar year. Some trips start in Honolulu and end in California or in the reverse order (NMFS unpublished observer data).

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

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. Specifically, we focus on three variables in the jeopardy definition that determine a species likelihood of survival and recovery in the wild: reductions in the species' reproduction, numbers or distribution.

The first of the variables—*reproduction*— encompasses the reproductive ecology of wild species, specifically, the size of the adult population, recruitment into the adult population, their fecundity and maternity (number of eggs versus number of live births), gender ratios, rearing behavior, and influence of habitat on reproductive success. Reducing one or more of these components of a species' reproductive ecology can reduce its increase in abundance or alter its population dynamics. Although scientific studies demonstrate the importance of reproductive ecology to the persistence and recovery of imperiled species, the reproductive ecology of many endangered and threatened species remains largely unknown and is, therefore, difficult to assess.

The second of these variables—*numbers*—receives the most attention in most risk assessments. It usually represents the total number of individuals that comprise the species or, in the case of imperiled species like sea turtles, the total number of adult females that are counted on nesting beaches. However, for species faced with extinction or endangerment other numbers matter: the number of populations that comprise the species, the size of those populations, the proportion of reproductively active adults in those populations, the proportion of sub-adults that can be expected to recruit into the adult population in any time interval, the proportion of younger individuals that can be expected to become sub-adults, the proportion of individuals in the different genders (where applicable) in the different populations, and the number of individuals that move between populations over time (immigration and emigration). Reducing these numbers and proportions can alter the dynamics of wild populations in ways that can reinforce their tendency to decline, their rate of decline, or both.

The third of these variables—*distribution*— refers to the number and geographic arrangement of the populations that comprise species. This geographic arrangement helps determine whether all of those populations are affected by the same natural and anthropogenic stressors and whether some populations occur in protected areas or are at least protected from stressors that afflict other populations. Changes in the *distribution of populations* provides insight into how a species is responding to long-term changes in their environment (for example, to climate change). Because species become extinct with the death of the last individual of the last population, any assessment of the extinction risks facing species must consider the extinction risks facing the populations that comprise the species. Changes in the *number of populations* or relationships between them provides the strongest evidence of their extinction risks or their probability of recovery.

Ecological theory and learned experience tell us that all populations of a threatened or endangered species, so long as appropriately defined by ecological and evolutionary discontinuity are important to the viability of the species. Not only are populations the fundamental unit of conservation, the distribution and abundance of species expands or contracts as the distribution and abundance of the populations that comprise them expands or contracts. The growth or decline of a population is captured by the fundamental equation:

$$N_t = N_0 + (\text{Births} + \text{Immigration}) - (\text{Deaths} + \text{Emigration})$$

Every population model derives from this equation (the “BIDE” equation). If N_0 is considered an initial bank balance, “births” as interest, “immigration” as deposits, “deaths” as spending, and “emigration” as transfers, the same mathematical relationships apply to a bank account. For species like sea turtles, in which the N_0 component of the BIDE equation includes the abundance of eggs, hatchlings, juveniles, sub-adults, as well as adults, a better analogy would be an account filled with long-term bonds that have different maturation dates.

We treat an interbreeding group of individuals as a “true” population when the group’s fate is determined by internal demographic processes (birth-death processes) rather than external processes (immigration-emigration) even when some flow of genes or individuals occurs. Disjunctions can result from spatial separation, temporal separation (different reproductive timing), genetic separation, behavioral separation, dietary separation (such as we find in orcas), or combinations of these.

Importantly, the ESA does not recognize a species, even the most imperiled species with a particularly dire pre-action status, as having a “baseline” jeopardy condition. The term “jeopardize the continued existence of,” in the plain language of section 7(a)(2) is a determination made about the effects of a federal action. Reductions in reproduction, numbers, or distribution of a species resulting from the action that have only an inconsequential impact at the species level would not be considered to rise to the level of "reduce appreciably” (83 Fed. Reg. 35178; July 25, 2018). In fact, the BIDE equation reveals the error in arguments that assert that the added loss of a few individuals from a population that exhibits a declining trend necessarily “jeopardizes” the continued existence of a population or species. A declining trend only means that the ratio between N_t and N_0 is less than 1.0 (or substantially less than 1.0, if we consider year-to-year variation). However, a population experiencing such a decline still has births and, in some cases, immigration. A small number of deaths would not alter the trajectory of even a declining population if the number of births exceeds the number of deaths in the same time interval (or if recruitment into a life history stage exceeds the number of deaths in that stage).

Our risk analyses reflect the 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 Mills and Beatty 1979; Stearns 1992; Anderson 2000). 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. 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. The significance of an effect, or clinical association, is informed by the effect size and the confidence intervals, and can provide a range of plausible values and be viewed as a measure of the strength of the evidence available.

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.

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 Regional 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* (WPRFMC 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: "ontogenetic 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 were limited to a small number of data sets 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—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.

We used causal loop diagrams as a point of discussion within our analytical team to achieve methodical conceptualization, elicit expert input, engage in scientific exchange, and establish a setting for collective learning, dialogue and inquiry. Where we could, we included a graphic in our causal loop diagram to represent the behavior of influential data and how it affected a particular loop. However, the purpose of the causal loop diagrams is to aid in deciphering how the species' life history, and the multiple threats posed to it, influence the species chances of survival and recovery, including how the proposed action effects the species chances of persistence. The causal loop diagrams are provided as a visual aid to understanding the interplay between a species life cycle and the threats it faces. Creating the causal loop diagrams became a method to address this multidimensional complex problem, and helped ensure that we examined the parts and the whole, and the interdependencies between the parts and the whole, so that we could be sure we adequately evaluated each species and their associated risk, and see what, if anything, we were inadvertently omitting as we conducted our analyses. In this respect, the causal loop diagram helped us focus our analyses and our work as a team to complete this consultation.

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.

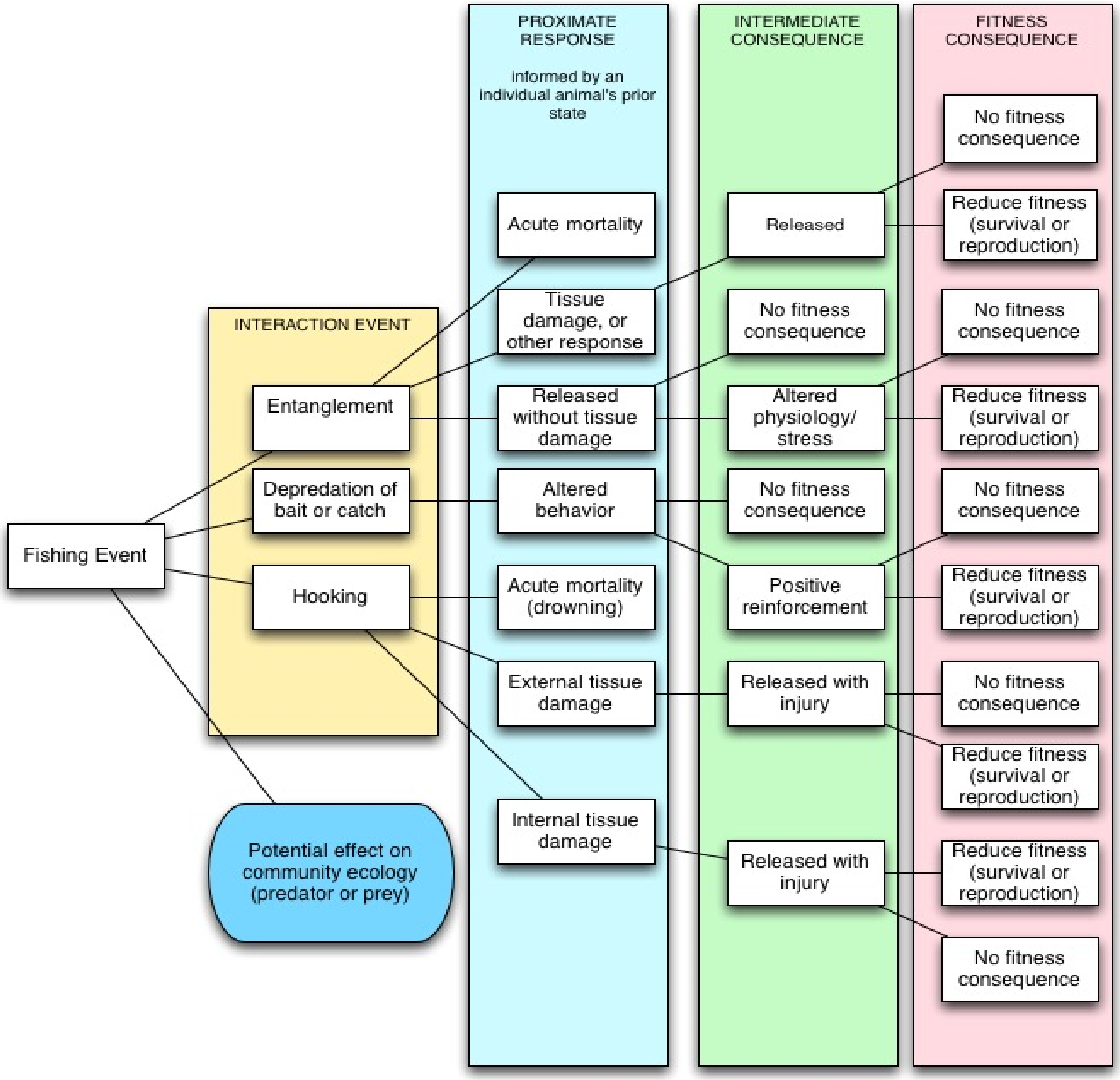


Figure 3. Conceptual model of the potential responses of sea turtles to exposure in the HI SSLL fishery and the pathways between exposure (the interaction event the individual animal has with the fishing gear), responses, and potential fitness consequences to the individual animals

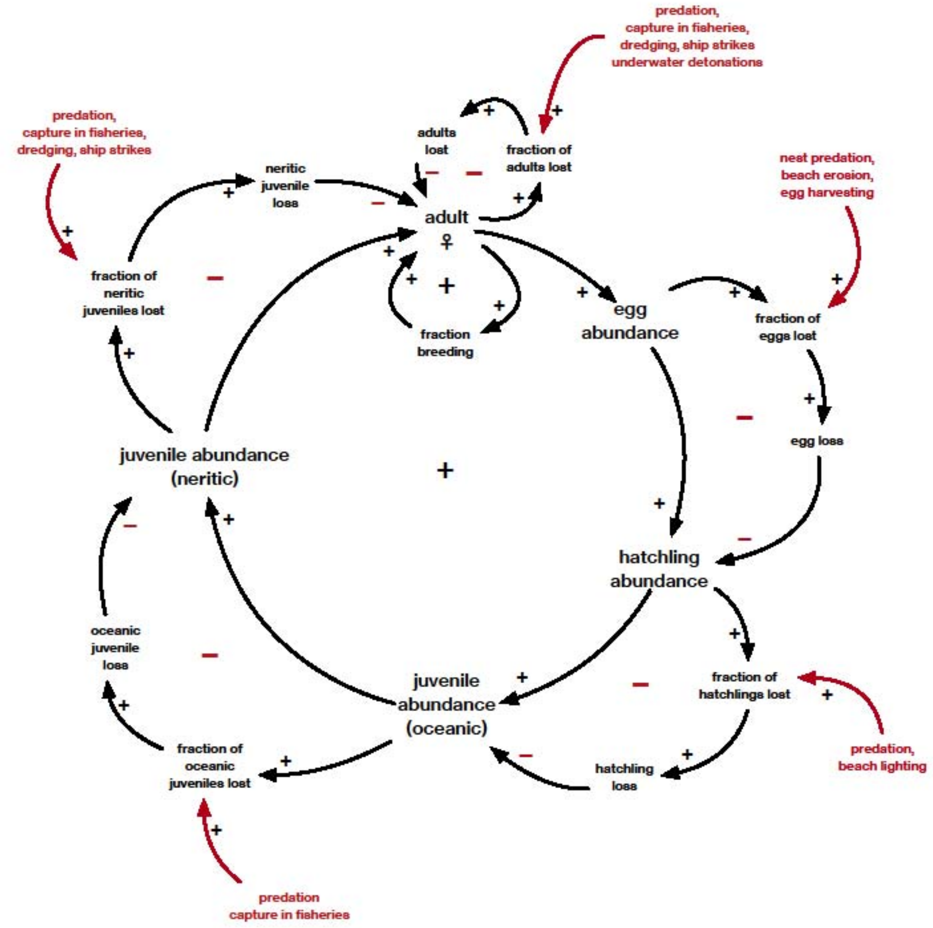
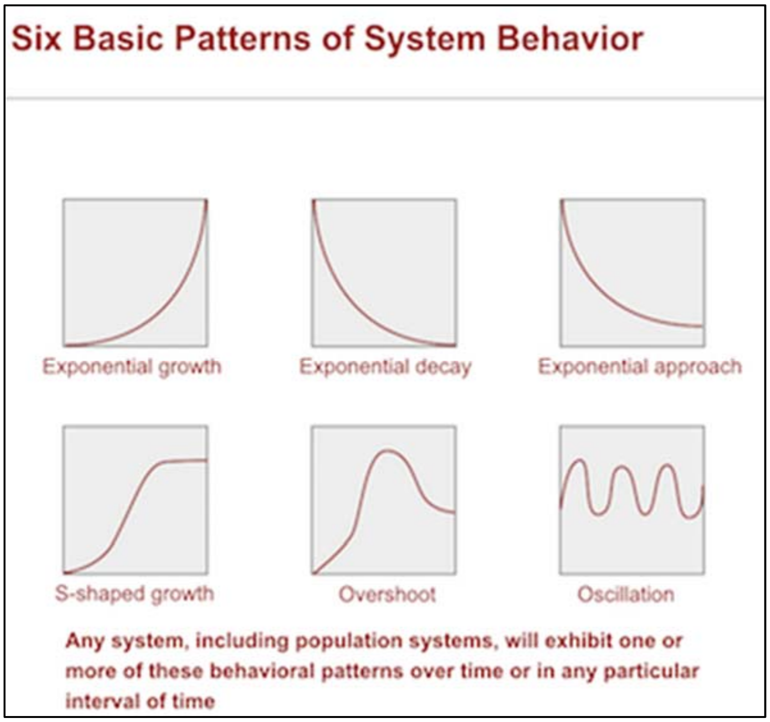


Figure 4. Example causal loop model for loggerhead sea turtles

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 identify where time lags can develop or accumulate in the system because they are often a source of imbalance in the feedback structure. In this case, we would be particularly concerned about variables that create a source of imbalance in a population's ecology or the processes that drive a species survival and recovery.

According to system dynamics, each loop in the diagram will likely correspond to one of six common dynamic patterns of behavior (Figure 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)

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

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 were 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 these data do 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 2016b) is to use climate indicator values projected under the Intergovernmental Panel on Climate Change (IPCC)'s Representative Concentration Pathway (RCP) 8.5 when data are available, or best available science that is as consistent as possible with RCP 8.5. RCP 8.5, like the other RCPs, were produced from integrated assessment models and the published literature; RCP 8.5 is a high pathway for which radiative forcing reaches >8.5 W/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. Presently, the IPCC predicts that climate-related risks for natural and human systems are higher for global warming of 1.5 ºC but lower than the 2 ºC presented in Table 2 (IPCC 2018). Changes in parameters will not be uniform, and IPCC projects that areas like the equatorial Pacific will likely experience an increase in annual mean precipitation under scenario 8.5, whereas other mid-latitude and subtropical dry regions will likely experience decreases in mean precipitation. Sea level rise is expected to continue to rise well beyond 2100 and while the magnitude and rate depends upon emissions pathways, low-lying coastal areas, deltas, and small islands will be at greater risk (IPCC 2018)

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

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

Given the limited data available on sea turtle populations, and other listed species like 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 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 U.S. 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 v. Winter, No. CV07-0335 (CD Cal. Mar. 22, 2007 and Ocean Mammal Institute v. Gates), 546 F. Supp. 2d 960 (D. Hawaii 2008) and Randall et al. (2010)). 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 step of our effects analyses—the risk analyses—adheres to the approach described earlier. We began these analyses by considering what probable responses can be expected when individuals are exposed to the HI SSLL fishery and what those mean for the fitness of the individuals exposed (changes in an individual's growth, survival, annual reproductive success, or lifetime reproductive success). Then we considered how any reductions in the fitness of these individuals affects the viability of the population(s) those individuals represent. We conclude our analyses by considering how any reductions in the viability of those populations can be expected to affect the survival and recovery of endangered or threatened species, as they are listed. If we conclude that one or more population is likely to experience an increase in its' extinction probability (measured using probability of demographic, ecological, or genetic extinction in 10, 25, 50, or 100 years) we assess the effect of that increase on the probable reproduction, numbers, or distribution of the species. For these analyses, we combine our knowledge of the patterns that accompanied the decline, collapse, or extinction of populations and species that have experienced these phenomena in the past supplemented by indicators of probable demographic performance produced by population viability analyses.

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 Holmes 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 only the mean number of incidental interactions in this biological opinion as the basis of our jeopardy analysis, which we have done in some past biological opinions, we report the mean incidental interaction and mean mortality predictions, but we also used an upper prediction interval (the 95th percentile) for estimating interactions and mortality in our jeopardy evaluation. We took this approach to avoid underestimating the risk of capturing and killing threatened and endangered species. Importantly, this metric is a better predictor of what *could* happen into the future, with a certain amount of assurance if fishing continues, whereas the mean would merely give us a summary statistic of what *has* happened in the past. Using the upper prediction does not assert that the number of interactions will reach or exceed this number, it does suggest that it would be unlikely that the fishery would exceed this number in any year.

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

For sea turtles available nesting count data provide one indicator of the population's performance; however, its limitations are many and as such, in contrast to the 2012 biological opinion on this same fishery, we are not using these 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 data 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 which could mask the true impact to populations and therefore the species). 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" (which includes DPSs) 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 et al. 2007; NMFS and FWS 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 (PNG), Vanuatu, Papua Barat in Indonesia, and the Solomon Islands (Dutton et al. 2007). Although West Pacific Ocean leatherback sea turtle mitochondrial Deoxyribonucleic Acid (mtDNA) studies indicate the population is genetically homogenous, our analyses recognize that species and their populations may be comprised of smaller, demographically distinct units based on other available information like is the case for West Pacific Ocean leatherback sea turtle (Dutton 2007; Dutton and Shanker in Spotila and Tomillo 2015).

The data available on nesting leatherback sea turtle females come 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 these data 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 demographic units that comprise the species 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.

In this consultation, we have constructed an analytical approach that relies on quantitative and qualitative methods for evaluating the action's effects on listed species and their designated critical habitat. We use both methods throughout this biological opinion to achieve what is ultimately, a qualitative meta-analysis that integrates the species status, environmental baseline, and cumulative effects with the effects of the action now and into the future. Our approach better incorporates available information into our assessment that are otherwise poorly studied or integrated by quantitative methods like demographic processes and threats or the interplay between demographic processes and threats that affect a species risk of extinction. Our integrated approach to evaluating risks allows us to combine quantitative information with qualitative information to enhance our understanding of the risks facing threatened and endangered species. We recognize that qualitative evaluations can be as robust as quantitative methods and in fact, qualitative evaluation can allow for deeper evaluation into quantitative numbers, what they mean, and understanding the implications of exposing a listed species to various threat regimes including threats posed by the proposed action. For these reasons, coupled with the inherent lack of demographic data for most species considered herein and a gross lack of numeric data to appropriately categorize threats, we necessarily relied upon a combination of analytical approaches throughout this consultation and biological opinion. Importantly, we tried to be explicit where we made certain assumptions and how they affected the outcome of our evaluation.

In this consultation, we examined several scenarios to estimate the risk the proposed action poses listed species. We considered that threatened and endangered species may be exposed to the proposed action for at least another 25 years. We choose this evaluation period recognizing that there is a long history of longlining in Hawaii, and yet significant changes in the management regime of this fishery occurred in a shorter period (the last 14 years). With this history, it seems reasonable that limitations in our predictive capabilities and changes in future management regimes would render predictions of a longer period increasingly speculative. Even so, we also considered multiple scenarios and different time-periods to assess the long-term prognosis of the species' status using prediction intervals typical of population viability analyses, and we used many of these same general metrics in assessing the risk the proposed action poses threatened and endangered species. For instance, our final risk evaluation considered the effect of the action against the available or best current numbers estimated for a particular species, as well as a future population size of 50%, 25% and 12.5% the current estimate. This means we examined the effect of the action based on a future condition that varied years into the future depending upon the species that is well beyond 25 years, based on our best estimates.

Importantly, this analytical approach 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 three 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 three 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 uncorrelated with one another even for successive 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.

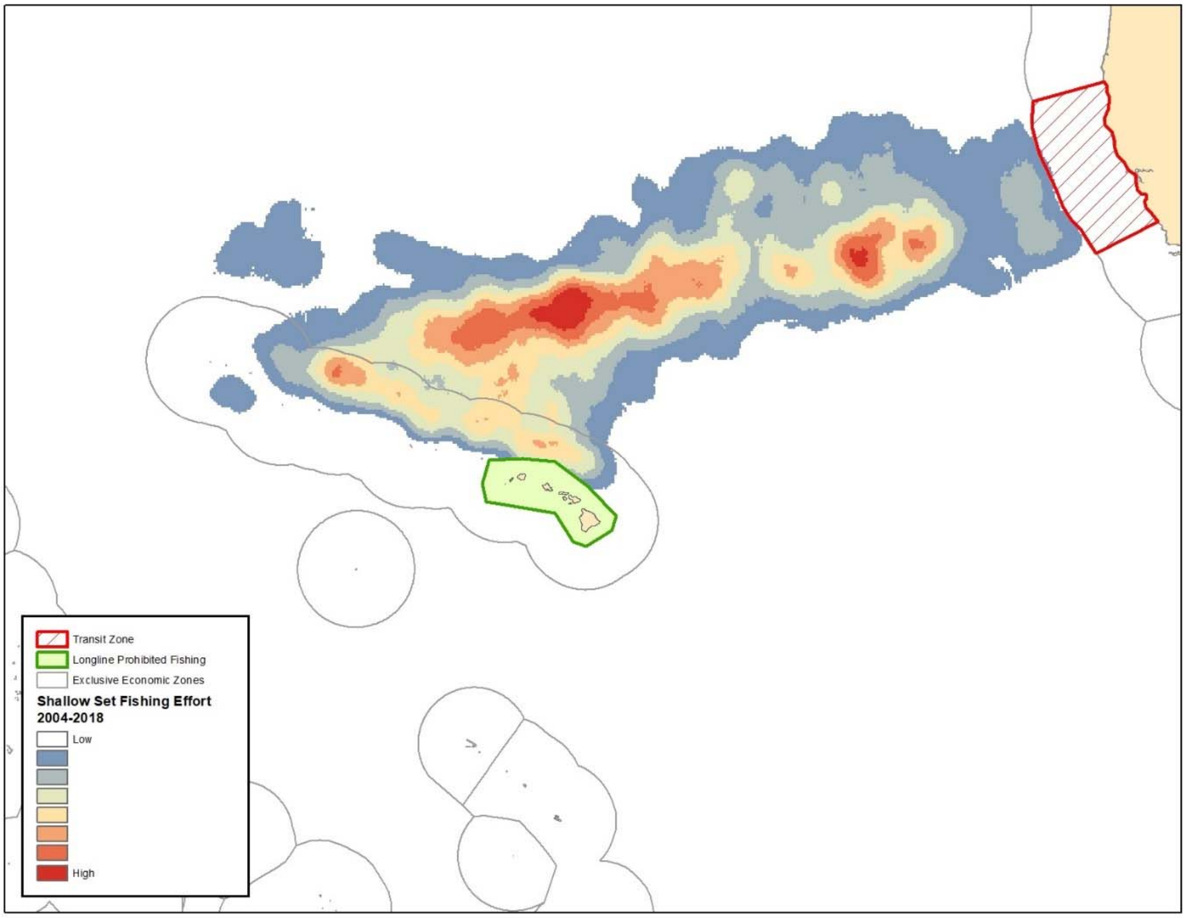


Figure 6. Action area with location of shallow sets made by the Hawaii longline fishery from 2004– 2018.

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	*Caretta caretta*	Endangered
Green sea turtle, Eastern Pacific	*Chelonia mydas*	Threatened
Green sea turtle, Central North Pacific		Threatened
Green sea turtle, East Indian - West Pacific		Threatened
Green sea turtle, Central West Pacific		Endangered
Green sea turtle, Southwest Pacific		Threatened
Green sea turtle, Central South Pacific		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*	Threatened
Hawaiian monk seal	*Neomonachus schauinslandi*	Endangered
MHI insular false killer whale	*Pseudorca crassidens*	Endangered
Humpback whale, Central America	*Megaptera novaeangliae*	Endangered
Humpback whale, Mexico		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	*Orcinus orca*	Endangered
FISH		
Eastern Pacific scalloped hammerhead shark	*Sphyrna lewini*	Endangered
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
Southern North American green sturgeon	*Acipenser medirostris*	Threatened

SPECIES COMMON NAME	SCIENTIFIC NAME	STATUS
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. These data shaped the observed pelagic exposure profile. Although species within this exposure profile may collide with a vessel, the primary stressor from the proposed action is hooking and entanglement. Lastly, the potential pelagic exposure is comprised of species that occur in the action area that could potentially interact with fishing gear or collide with a vessel. However, there are no documented observations of any fishery interactions between these species and the 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; Mexico humpback whale; Central America humpback whale; blue whale; sei whale; North Pacific right whale, Southern Resident killer whale; Hawaiian monk seal; Eastern Pacific scalloped hammerhead shark; Central California coast coho salmon; Central Valley spring-run Chinook salmon; Sacramento River winter-run Chinook salmon; Central California coast steelhead; California coast steelhead; Southern North American green sturgeon; 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, southern North American 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	Mexico Humpback whale
Sperm whale	Blue whale
Loggerhead sea turtle	North Pacific right whale
Leatherback sea turtle*	Sei whale
Green sea turtle*	Eastern Pacific scalloped hammerhead
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
	MHI Insular false killer whale critical habitat
	Southern resident killer whale
	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
	Southern North American green sturgeon
	Southern North American green sturgeon critical habitat
	Black abalone
	Black abalone critical habitat
	White abalone

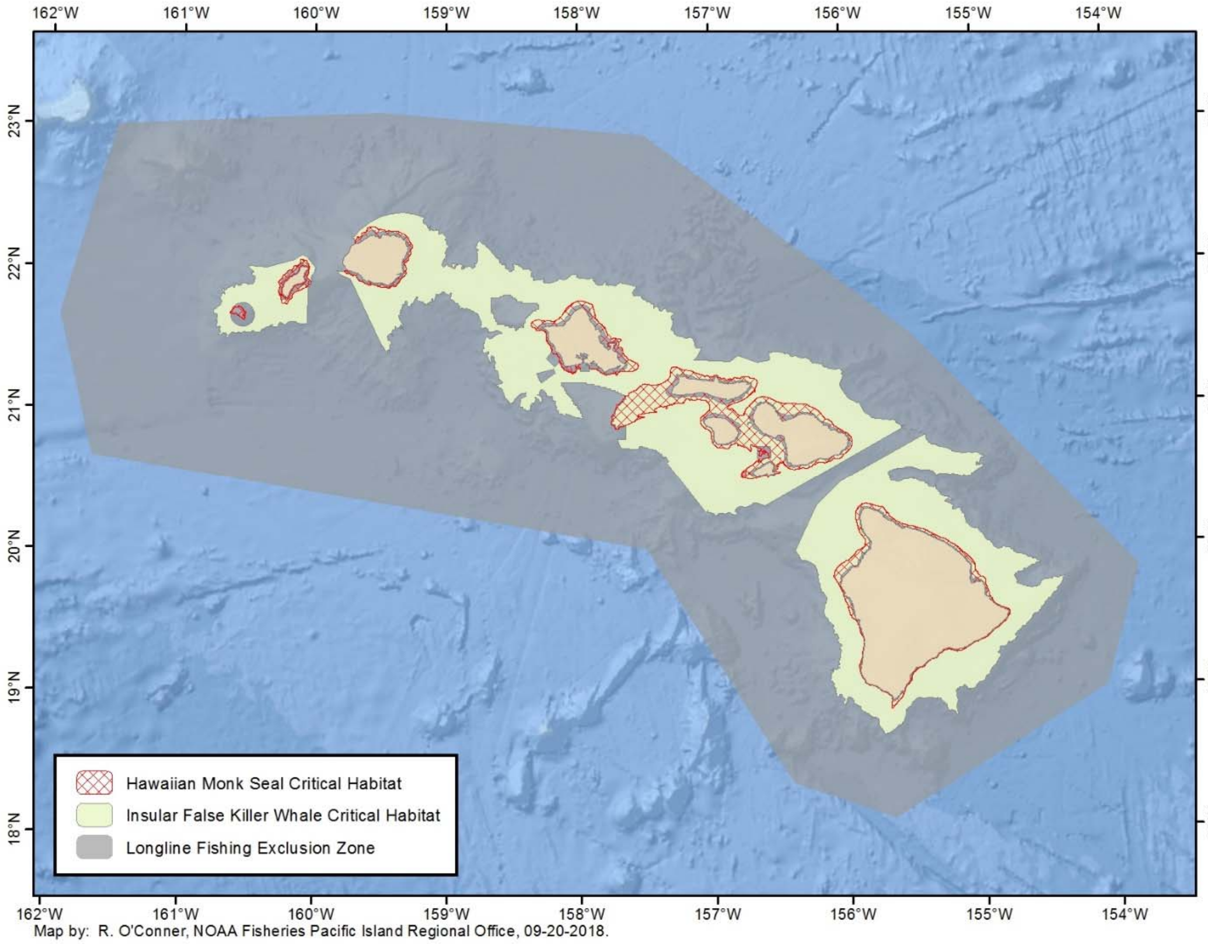


Figure 7. The longline fishing exclusion zone around the Main Hawaiian Islands

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 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 scalloped hammerhead sharks, Mexico 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 are 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 known interactions with the HI SSLL fishery.

The HI SSLL fishery has had four interactions with the humpback whale, although, these interactions were attributed to the Hawaii humpback whale, which is not listed. Due to the northern migrations of Central America and Mexico 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 species 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 that the population of the Hawaii humpback is approximately 3.5 times greater than that of the Mexico humpback whale, the probability that the humpback whale is a member of the Hawaii humpback whale is significantly higher than the alternative.

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 hammerhead, and eight that were unidentified hammerheads, were captured in the vicinity of the non-listed central Pacific hammerhead shark. There has never been an observed interaction with an endangered Eastern Pacific scalloped hammerhead shark. 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 fishery 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 fisheries. 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. However, given the small number of vessels participating in the fishery, the small number of anticipated vessel trips, the slow vessel speeds during fishing operations and vessel transiting, the expectation that ESA-listed marine species would be widely scattered throughout the proposed action area, the potential for an incidental vessel strike is extremely unlikely to occur. Thus, NMFS expects this stressor would have insignificant effects on the ESA-listed resources in Table 3 and Table 4.

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 kt (NMFS 2018b). Since 2004, the number of vessels participating in the HI SSLL fishery has fluctuated, albeit numbers have always been low with a high of only 35 vessels in 2006. Similarly, there is variability in the amount of fishing trips each year with a high of 112 trips in 2009. Although some trips would depart California ports, the risk of collision with a vessel fishing in this fishery is even smaller in California coastal areas because the number of vessels transiting these areas is a fraction of the total fleet. As a result, the risk of collision with a HI SSLL fishery vessel is considered unlikely.

Even in high density areas like areas around Hawaii, collisions between protected species 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 2008b). This calculates to a 0.006% probability of a vessel strike with sea turtles for all vessels and trips, and many of these vessels are not likely reducing speeds or employing lookouts for listed species.

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). Masking and behavioral avoidance are the most likely responses of animals in the vicinity of HI SSLL fishing vessels. However, NMFS expects that vessel noises would have an insignificant effect on listed species because they would not be expected to result in measurable responses (should never reach the scale where harm or harassment occurs).

Given the size of the HI SSLL fishery (the small number of vessels in the fishery), the fact that the sound field produced by the vessels in the fishery is relatively small and would move with the vessel, the animals would be moving as well, vessel speeds would be slow, vessel transit vectors would be predictable, sudden or loud noises would be unlikely or infrequent, and generally the sound field would be in motion, we would expect that any exposure to noises generated by this fishery would be short-term and transient and would generally be ignored by animals that are temporarily exposed to sounds emanating from the vessels in this fishery. Numerous studies demonstrate that marine animals are unlikely to change their behavior when confronted with stimuli with these attributes, and we would also expect masking would be highly unlikely to occur, if not improbable. Although hydraulics may have the potential to create loud noises; due to the expected above water operations, frequency and duration of time these species spend at the surface, dissipation of sound from the source, and the poor transference of airborne generated sounds from the vessel to ocean water through the hull, it is highly unlikely noises generated from vessel operations would elicit behavioral reactions from ESA-listed species considered in this consultation. Additionally, 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 IFKW. Thus, NMFS expects this stressor would have insignificant effects on the ESA-listed resources in Table 3 and Table 4.

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

The diffuse stressors associated with the longline fisheries: vessel waste discharge, gear loss, and carbon emissions and greenhouse gasses, can affect both pelagic and coastal areas. ESA listed resources could be exposed to discharges, and run-off from vessels that contain chemicals such as fuel oils, gasoline, lubricants, hydraulic fluids and other toxicants. Although, local and Federal regulations prohibit the intentional discharge of toxic wastes and plastics into the marine environment at least four enforcement actions were taken against Hawaii longliners in 2018 (The Maritime Executive 2018). Even so, the amount of vessel waste discharge from the HI SSLL fishing fleet is difficult to quantify with any accuracy and presumably quite small relatively to other sources of similar wastes. NMFS' observer program database contains 37 entries of documented releases of wastes in HI SSLL fishery 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). Accidental loss and breakage of gear is also common, and lost gear can continue to fish and incidentally hook and entangle marine species or get stuck in areas of designated critical habitat.

HI SSLL fishery vessels also 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 fishery, we expect the contribution to global greenhouse gases to be relatively inconsequential based on the low number of participants in the fishery.

Although leakage, wastes, gear loss and vessel emissions would occur as a result of the HI SSLL fishery, given the small number of vessels participating in the fishery, the small number of anticipated vessel trips, the small chance that ESA-listed resources would be exposed to measurable or detectable amounts of wastes, gear, or emissions from this fishery, NMFS expects that this stressor would have discountable and insignificant effects on the ESA-listed resources in Table 3 and Table 4.

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: (1) Northwest Atlantic Ocean; (2) Southeast Atlantic Ocean; (3) Southwest Atlantic Ocean; (4) Northeast, and (5) Southwest Indian Ocean; (6) West Pacific; and (7) East Pacific Ocean populations (35 FR 8491; Wallace et al. 2013b, 2013c). 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 (Eckhert et al. 2012; NMFS and FWS 2013). 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 are primarily 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 HI DSLL 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 centimeters (cm) straight carapace length (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 this variation may influence demographic performance of populations that are geographically separated (use different Large Marine Ecosystems (LME) 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 occur in Indonesia, Papua New Guinea, the Solomon Islands, and Vanuatu (Eckert et al. 2012; Tiwari et al. 2013). The majority of documented nesting occurs along the Bird's Head region of Papua Barat, Indonesia (70% or greater depending upon conclusions of the author and method of estimate; Dutton et al. 2007) although numbers have steadily declined (Tiwari et al. 2013). The IUCN review evaluated six populations (nesting (index) aggregations), and within the Papua Barat nesting aggregation, Jamursba-Medi beach comprised the largest proportion of nesting activity 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 trends in Indonesia would suggest the overall nesting distribution of the population has likely experienced concomitant declines although recent data from other nesting areas are not available (NMFS and FWS in prep).

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 of nesting beaches in Indonesia (Jamursba-Medi and Wermon beaches, Papua Barat), PNG (Huon Coast beaches), and the Solomon Islands (Benson et al. 2011). In PNG, however, some females may have visited additional nesting areas during a single nesting season 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, and nesting is documented year round. Based on Benson et al. (2011), summer nesting aggregations migrate into the temperate North Pacific Ocean through the action area, south of the action area, and into tropical waters of the South China Sea, and winter nesting aggregations migrate into the southern hemisphere (see Figure 1 in Benson et al. 2011). What appears to be a latitudinal cline in the occurrence of life history strategies may, however, be a result of incomplete monitoring. There may be year-round nesting in other places in the West Pacific, including the Solomon Islands and PNG.

Turtles nesting in during the boreal summer months migrate into the South China Sea and 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, 2007b, 2011). A portion of summer nesting turtles from the Papua Barat, Indonesia Bird's Head region exhibit 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 (Eckert et al. 1988). Stable isotope analysis can complement satellite data of leatherback sea turtle movements and identify important foraging areas that reflect regional food webs (Seminoff et al. 2012).

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, 2012a). 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. 2012b). Prey abundance and distribution may explain why the Pacific populations are in worse shape than the Atlantic populations (Bailey et al. 2012a).

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., Hamann et al. 2011; Gaspar et al. 2012; 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, Huon Coast, 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). Gaspar and Lalire (2017) hypothesized that juveniles migrating across the Pacific may reach sexual maturity after 15 years, the mean age at which active turtles reach the California ecoregion. Approximately 30 to 60 percent of Jamursba-Medi summer nesters (n=78 in 2007 and 2010) foraged in waters off California (Seminoff et al. 2012).

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. 2013b [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). 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 data set available was developed by Inter-American Tropical Tuna Commission (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 (Donoso and Dutton 2010; NMFS and FWS 2013).

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 (Shillinger et al. 2009, 2011; Bailey et al. 2012).

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 HI DSLL 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. 2010a, 2011, 2013b). 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. (2013b) 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. 2013b) 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). 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 (subpopulations) make up the majority of the West Pacific Ocean leatherback sea turtle population: Papua Barat, Indonesia, PNG, and the Solomon Islands (Dutton et al. 2006; Dutton et al. 2007; 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). Animals that travel long distances for foraging might be expected to have longer remigration intervals and larger energy reserves as evidenced by body size, than those rearing in areas close to nesting beaches where they could return to nest more quickly and could do so with less energy reserves. However, Benson et al. (2011) revealed that the CCL and curved carapace width (CCW) of winter nesters was significantly larger than the summer nesters although temperate foragers generally had greater curved carapace width over tropical foragers for all life history types combined (Benson et al. 2011). Benson et al. (2011) suggested that would indicate that temperate foragers may have greater weight gains before returning to nest, and that in turn they would likely have higher reproductive output.

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 smallest recorded breeding leatherback sea turtle is 106 cm SCL, which suggests that leatherbacks may be reproductive at smaller sizes (Stewart et al. 2007), the West Pacific Ocean population of leatherback sea turtles does tend to be larger bodied (~145 cm CCL; Stewart et al. 2007). Based on the size distribution of leatherback sea turtles captured in the HI SSLL fishery, we expect that the majority are adult and reproductively mature (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 5 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 5. 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					

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 (%)**
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 (48)			9	39.4 +/- 22.1 (48)	26.5 (33)
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 Kamiali beach, PNG over a 10-year boreal winter nesting period (2000-2009) to estimate female survival. The annual survival probability (0.85) was constant over the 10-year period (Pilcher and Chaloupka 2013), but was lower than those estimated for two Atlantic rookeries which were 0.893 and 0.91, respectively (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 2013). Some subpopulations are stable or increasing, but other populations for which information is available are either decreasing or have collapsed (PLAWG 2012; NMFS and FWS 2013; Wallace et al. 2013b; NMFS and FWS in prep). 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. 2013b). 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. (2013b) 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. Based on available data collected through 2010 Tiwari et al. 2013 estimated that 1,438 individuals make up the West Pacific Ocean mature adult leatherback sea turtle population (includes males, females and non-breeding females). Wallace et al. (2013b, 2013c) 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 to 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. 2013b, 2013c]). Jones et al. (2012) estimated there were 294,088 (range 114,663-628,875) leatherback sea turtles in the Pacific and roughly 85% of their estimate represented the West Pacific Ocean population. At the time, based on Jones et al. (2012) the West Pacific Ocean population would have totaled approximately 249,974 (range 97,463-534,543) leatherback sea turtles. The Jones et al. (2012) study estimated the Pacific contained 6,199 (range 4,292-8,103) adult leatherback sea turtles. The adjusted estimate for the West Pacific Ocean population would have been 5,269 (3,648-6,888) adults. Jones et al. (2012) estimates were based on available nesting data current to 2004 as listed in Hitipeuw et al. (2007) and Dutton et al. (2007). Jones et al. (2012) assumed that the population had a 1:1 sex ratio, which would have meant the West Pacific Ocean population was comprised of about 2,635 (1,824-3,444) adult female leatherback sea turtles in 2004. The current estimate of adult female nesters for approximately 75% of the population is in Table 7. Using this number and assuming a 3:1 ratio of females to males, we estimate the current adult portion of the population is 1,851 (1,488-2,320). We used the proportion or change in the estimates derived from the information contained in Jones et al. (2012) and more recent Jones et al. (2018) to estimate the current population size of the West Pacific Ocean leatherback sea turtle. The total West Pacific Ocean population is comprised of about 175,000 leatherback sea turtles but may range between 68,000 and 360,000 individuals.

Indonesia

The Bird's Head subpopulation harbors the last remaining nesting aggregation of significant size in the Pacific (Dutton et al. 2007; Tapilatu et al. 2013) with approximately 75% of regional nesting occurring along the northwest coast of Papua Barat, Indonesia (also known as the Bird's Head Peninsula) (Dutton 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 beaches 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 18-year period 1993-2011, 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). Monitoring at Wermon beach began in 2002 and ran through 2011 with nesting activity ranging from a high of 1,788 nests laid during the 2002/03 winter nesting season to a low of 1,096 nests laid during the 2010/11 season (Hitipeuw et al. 2007; Tapilatu et al. 2013). Since 2005, nesting has declined by 11.6% per year at Wermon, representative of 189-249 females, or a 62% decline since monitoring began in 2002 (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.

Since 2012, monitoring effort at Jamursba-Medi and Wermon beaches has been somewhat variable and the overall nesting trend has continued to decline by 5%. While there appears to be a slight upside to an oscillating trend (as depicted in Figure 5) in recent nesting activity, at the moment it is not affecting the long term trend and we will need more years of data to understand what the upside in the oscillation means for the population (Tiwari *et al.* in prep; Jones et al. 2018, see figures 9 and 10) 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 as cited in NMFS and FWS in prep).

In the Manokwari region, Papua Barat, 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 so is not used in current population estimates.

A recent NOAA funded, WWF-Indonesian assessment team identified a new leatherback nesting area in 2017 on three north coast beaches of Buru Island in Central Maluku (i.e., Waenibe, Waspait, and Wamlana villages, Fena Leisela District, Buru Regency; WWF 2018). 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 as cited in NMFS and FWS in prep). Monitoring activities are ongoing, including genetic sample collection.

Papua New Guinea

In Papua New Guinea, the majority of known nesting activity occurs during the winter months (November to February) along the Huon Coast on the northeastern coast of the Morobe Province, where 576 females have been tagged between 1999 and 2013 (Pilcher 2006, 2009, 2010a, 2011, 2012, 2013; Pilcher and Chaloupka 2013). Aerial surveys along the Huon Coast in January and December between 2004 and 2006 documented 276 nests, with an estimate of 500 nests per season (Benson et al. 2007b; Dutton et al. 2007). During the Huon Coast Leatherback Turtle Project, between 2005 and 2012, an average of 258 nests were laid per season (range: 193 to 527) at seven beaches which comprised approximately 35 km of nesting habitat along the Huon Coast (Pilcher 2013; WPRFMC 2015). One challenge in estimating nesting activity in Papua New Guinea is that leatherback site fidelity appears to be variable, with satellite tagged animals seen visiting a number of areas during one nesting season (Benson et al. 2007b). For example, a number of Huon Coast nesters visited other nearby beaches and at east-facing beaches of the Huon Peninsula, including Bougainville and Woodlark Islands during a single nesting season (Benson et al. 2007b). Therefore, for this assessment NMFS considers the aggregation of Huon Coast nesting beaches as a subpopulation.

Additional nesting activity occurs in other areas of PNG, such as along the north coast of the Madang Province and on several islands including Manus, Long, New Britain, Bougainville, New Ireland, and Normanby (Spring 1982; Prichard 1982; Benson et al. 2007c; Dutton et al. 2007). In these areas, nesting activity has not been quantified via standardized or consistent methods but information has been obtained via community surveys, aerial surveys, or rapid assessments. Nesting occurs primarily in the winter months, although low-level, year-round nesting may also occur (Spring 1982; Dutton et al. 2007; Pilcher and Chaloupka 2013). Approximately 50 nests may be laid annually along the north coast of the Madang Province (Benson et al. 2007b; TIRN 2017). The Islands of New Britain and Bougainville may host approximately 140 to 160 nests per year, respectively (Benson et al. 2007b; Dutton et al. 2007; Kinch et al. 2009). On Bougainville Island, aerial surveys conducted during the 2005 and 2007 nesting seasons documented a mean of 68 nests (range: 41 to 107 nests) or an extrapolated estimate of 160 to 415 nests per year (Benson et al. 2007b; Dutton et al. 2007). In 2009, a one week full-island ground survey (conducted by boat and foot), recorded 46 leatherback nests (Kinch et al. 2009).

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 deposited approximately 100 nests per season (Ramohia et al. 2001; Pita and Broderick 2005). The Tetepare Descendants' Association (TDA) 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 (MacKay 2005; Goby et al. 2010).

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 2010b), 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, TDA 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 (Pilcher 2010b; TDA 2013).

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). 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.

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; Wallace et al. 2013b). 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. 2006c).

Population Viability

To provide insight into the potential extinction risks facing West Pacific leatherback sea turtles, Jones et al. (2018) developed a mathematical model of the West Pacific leatherback sea turtle population and used that model to simulate the population’s pattern of growth or decline over time. 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 some 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 beaches 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 were 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 are lacking to index other age classes, or the male portion of the population, there is inherent uncertainty. So we consider *r* an index of the population's growth rate rather than an estimate of its growth rate.

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 and 2002b as cited in NMFS 2004a), 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 6 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 6. 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, 2002b	▪ 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 2004a	▪ 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 as cited in Jones et al. 2018), and projections (see below) were completed in R (R Development Core Team 2012) using the 'coda' and 'jagsUI' packages (Kellner 2015 and Plummer et al. 2006 as cited in Jones et al. 2018). 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 3-year running sum was based on assuming a 3-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.

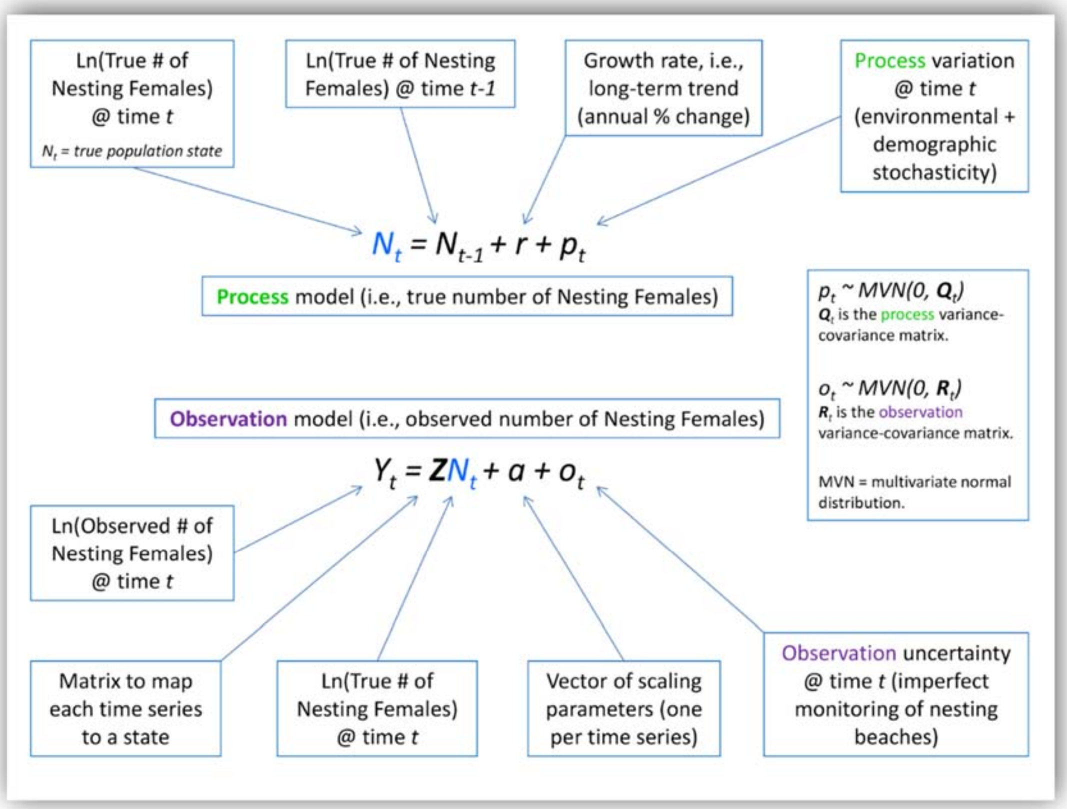


Figure 8. Schematic illustration of Bayesian state-space model based on stochastic density-independent exponential population growth.

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 (Tapilatu et al. 2013). 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.

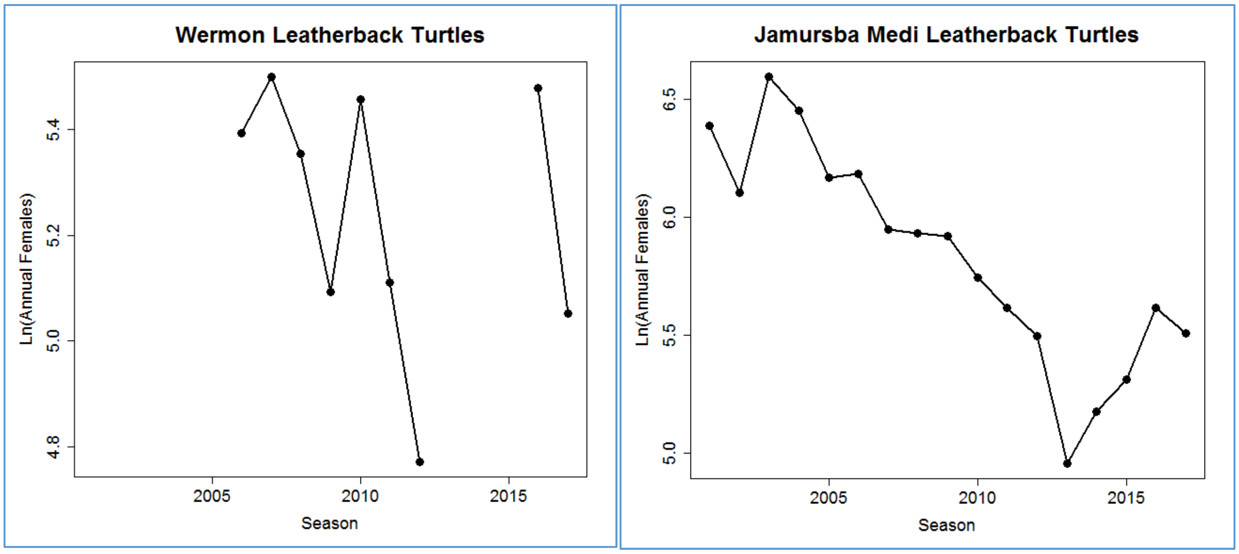


Figure 9. Natural log of annual nesting female leatherback sea turtles in Jamursba Medi and Wermon aggregations.

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.

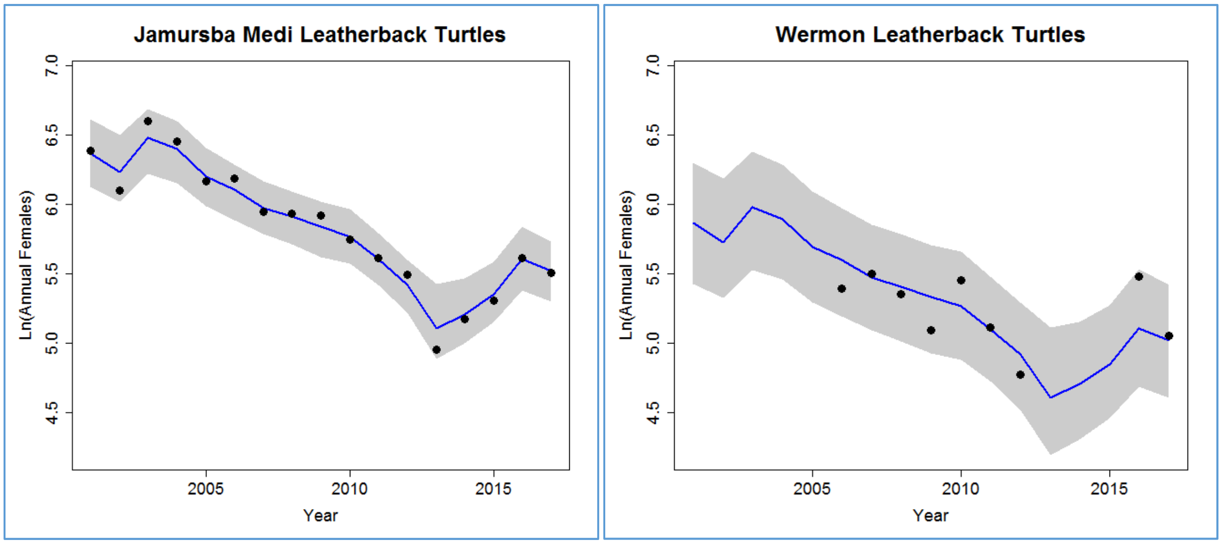


Figure 10. Predicted trends in nesting female leatherback sea turtles from Jamursba Medi and Wermon (median (blue line) of the natural log of annual nesting females with 95% credible intervals (gray shading)).

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 7.

Table 7. 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 8. Whereas Table 9 lists the probability of the population reaching abundance thresholds at 5, 10, 25, 50, and 100 years from the data year (2017).

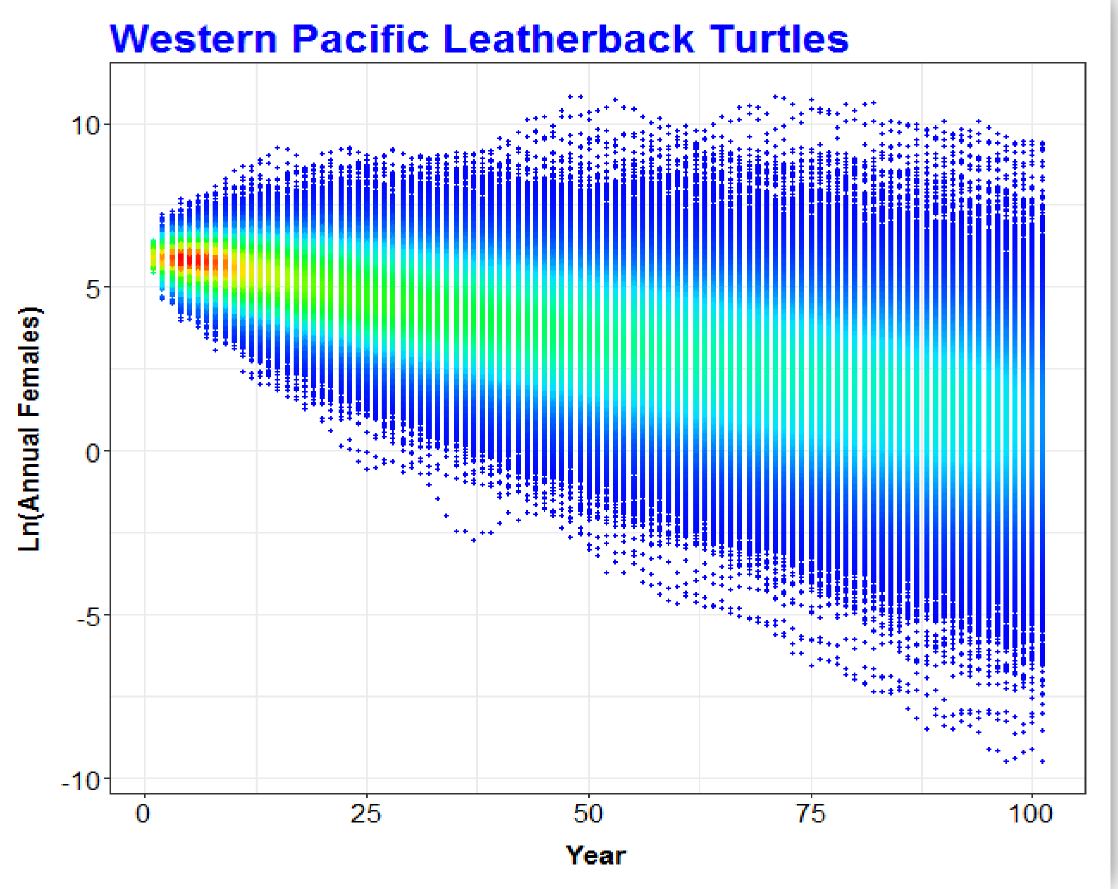


Figure 11. Model projections (10,000) of annual West Pacific Ocean leatherback sea turtle females in natural log space for 100 years into the future from 2017 (data year).

Table 8. 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 9. 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 (Pritchard 1982; Spotila et al. 2000; 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 (MTSG 2012; NMFS and FWS 2013). Wallace et al. (2013b) 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, U.S. 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 km (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. 2006b) 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 2013). 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 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 2018 the American Samoa longline fishery is estimated to have had 55 interactions (38 mortalities) (NMFS 2019a). Six samples have been analyzed and all are from the West Pacific Ocean population (P. 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 to 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 East and West Pacific Ocean leatherback sea turtle populations are not entirely clear, it is likely the result of historic intensive egg and turtle harvest on nesting beaches and in foraging habitats, incidental capture of adults and juveniles in fisheries, and natural fluctuations due to changing environmental conditions that influence prey abundance and distribution (Wetherall et al. 1993; Sarti Martinez et al. 2007; Santidrian Tomillo et al. 2007, 2008; Wallace et al. 2010b; Saba et al. 2012; WWF 2018 as cited in NMFS and FWS in prep). High levels of bycatch in the pelagic drift gillnet fishery which operated in the Central North Pacific and resulted in potentially thousands of leatherback turtles interactions cannot be discounted (Wetherall et al. 1993). 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; NMFS and FWS 2013), and egg consumption by humans and domestic animals (e.g., dogs) persist on nesting beaches where protection is incomplete or projects do not exist (Tapilatu and Tiwari 2007; Urteaga et al. 2012; NMFS and FWS 2013; Pilcher 2013). Direct take and fisheries bycatch are still considered major obstacles to population recovery (Wallace and Saba 2009; MTSG 2012; NMFS and FWS 2013, in prep).

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. 2013a). 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; WPRFMC 2009b). Since the HI SSLL fishery re-opened in 2004, a 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 2018i) 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 2018i).

In addition to the harvest of females and their nests, foraging leatherback turtles are harvested in the waters of the Kei Kecil Islands, Maluku Province, Indonesia. In this location, an indigenous and subsistence harvest of juvenile, subadult and adult turtles (52-204 cm CCL) occurs and has likely been a key feature of the local traditional culture for centuries (Compost 1980; Hitipeuw and Lawalata 2006, 2008; IOSEA 2013). Customary law ("*hak adat*") authorizes the ritual turtle hunt in the nine villages of the traditional kingdom of the Nafit people living within the Kei Kecil Islands. Suarez and Starbird (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 and Lawalata 2008). At one point, it was assumed that harvest pressure may have been on the decline and possibly no longer an issue (NMFS and FWS 2013); 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 10) (WWF 2018 as cited in NMFS and FWS in prep). 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 (Hitipeuw and Lawalata 2006).

Table 10. Number of direct takes in the Kei Kecil Islands documented since 1994.

Time frame	Number harvested	Source
3 month period in 1994	200	Starbird 1994
annually	100	Suarez and Starbird 1996
2003-2006	100	Hitipeuw and Lawalata 2006
Over 8 years	431	Hittipeuw and Lawalata 2008
2017	103	WWF 2018 as cited in NMFS and FWS in prep

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 (Bellagio Steering Committee 2008; PLAWG 2012; NMFS and FWS 2013).

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 (Bellagio Steering Committee 2008; NMFS and FWS 2013). 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 2013). 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 and FWS 2013).

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 FWS 2013). 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 Nino Southern Oscillation (ENSO). Saba et al. (2007) showed that nesting females in Costa Rica exhibited a strong sensitivity to ENSO where as cool La Nina events correspond with a higher remigration probability and warm El Nino 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 Nino/La Nina 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 in combination with fin-fish bait) or to allow turtles to escape without harm (e.g., turtle exclusion devices), 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 (NOAA 2013b). For example, switching to large circle hooks and mackerel bait in 2004 with complimentary fishery-based outreach and education resulted in an 84% reduction in the leatherback sea turtle interaction rate in the HI SSLL fishery (Swimmer at al. 2017). 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. In 1989 the United Nations

General Assembly placed a moratorium on the practice of pelagic drift net fishing, and in 1992 the UN banned the use of drift nets in international waters including the Taiwan and Japanese drift gillnet fishery which is believed to have captured thousands of leatherback turtles during their North Pacific migrations (Wetherall et al. 1993; Benson et al. 2015).

NMFS and partners have been involved in leatherback sea turtle research and conservation activities in the Western Pacific since 1999 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 worked with various organizations over time 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 between 2007 and 2011 (NMFS 2011). 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 WPRFMC 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), recent survey information indicates that harvest pressure continues (WWF 2018).

In PNG, the community-based leatherback project which operated between 2004 and 2013 along the Huon Coast monitored nesting activity, implemented conservation measures to protect nests from dog predation (e.g., bamboo grids), and worked to reduce localized harvest through community development incentives (CDI) since its inception in 2003 through 2013. Through community development incentives, communities at large experienced the benefits of the leatherback sea turtle project over time even if they themselves did not personally gain (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). Community development incentives projects 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). As a result of the Huon Coast leatherback turtle project, over 100,000 hatchlings may have been produced (WPRFMC 2015). Unfortunately, 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 at Sasakolo and Litogarhira has supported villagers to monitor nesting activity and 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 NMFS' Southwest Fisheries Science Center and The Nature Conservancy, with additional funding support from the International Sustainable Seafood Foundation, the Ocean Foundation, and NMFS). Additionally, the 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 supported a local NGO, Wan Smolbag, intermittently between 2007 and 2015 to train local villagers to monitor nesting activity, conserve leatherback sea nests, 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. During the 2011-2012 nesting season, 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% (Lopez 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. 2013b; NMFS and FWS 2013).

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 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 for vessels that fish in a shallow-set manor and target swordfish. 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. While these methods have been proven successful in reducing sea turtle interactions, we note that a recent analysis of the number of vessels that actually use these measures is only a small percentage of the overall longline effort in the Pacific. During a workshop convened to assess the effectiveness of WCPFC's Sea Turtle CMM, they found that the measure applies to approximately one percent of longline fisheries in the Convention Area (Common Oceans (ABNJ) Tuna Project 2017). The remaining effort is from vessels that deep-set or shallow-set fisheries that target species other than swordfish (Common Oceans (ABNJ) Tuna Project 2017). A new measure was adopted in 2018 (CMM 2018-04) that extends the measure to all vessels that fish in a shallow-set manner (WCPFC 2018b).

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

4.2.1.6 Summary of the Status of the Leatherback Sea Turtle

In this section of this biological opinion, we explained that the endangered leatherback sea turtle, which is endangered throughout its range, is a global metapopulation represented by 7 extant populations (a metapopulation in this respect is a set of discrete populations or subpopulations of the same species in which migration between those populations is significantly restricted). The Pacific Ocean portion of the metapopulation was once comprised of three spatially and genetically distinct populations. One population, Terengganu, Malaysia, is considered functionally extinct. The remaining two populations the East Pacific and West Pacific Ocean populations, are critically endangered. Presently the Eastern Pacific leatherback nesting is likely comprised of no more than 1,000 adult females, with only 150 to 200 females nesting annually per year at primary and secondary nesting beaches (NMFS and FWS 2013; MTSG 2012). Population trends have declined by 97.4% during the past three generations and best estimates suggest that the population will decline by 99.9% over the next generation (2040) to fewer than 30 adult females (Wallace et al. 2013b).

We focused most of our analysis on the West Pacific Ocean population because it is the only population recorded as captured in the HI SSLL fishery, and while the globally listed species is comprised of seven demographically discrete populations, the West Pacific Ocean population is demographically unique, it plays an important role in the larger metapopulation represented by the global species and the loss of the West Pacific Ocean population would represent a substantial impediment to the survival and recovery of the species as a whole. Furthermore, with the loss of one population (Terengganu) and extensive declines in the other Pacific population (East Pacific Ocean) the West Pacific Ocean leatherback sea turtle population represents the largest remaining leatherback population within the Pacific Ocean and may represent the best chance of recovering the species within the Pacific Ocean.

NMFS used several mathematical models to assess the probable trend of the West Pacific Ocean leatherback turtle population. Those models suggest that the population is declining at a rate of about 5% per year and is at risk of falling to less than half its current abundance in as few as 3 years (the best estimate from NMFS mathematical model is a mean of 26 years and the range is 3-86 years with a 98% probability it will fall below this threshold value; see Table 8). Importantly, we consider this an index estimate because it only represents a portion of the population—we lack data on remigrations, additional nesting subpopulations, other age classes, males, and there is other inherent uncertainty.

We constructed a diagram of the primary factors that affect the leatherback sea turtle's population dynamics (as illustrated in Figure 12), and used this as a point of discussion within our analytical team. Generally speaking, leatherback sea turtles face various threats throughout each stage of their respective life cycles. Our diagram of the species' population dynamics includes natural and anthropogenic threats that affect each life cycle stage. 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 asymptotically 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.

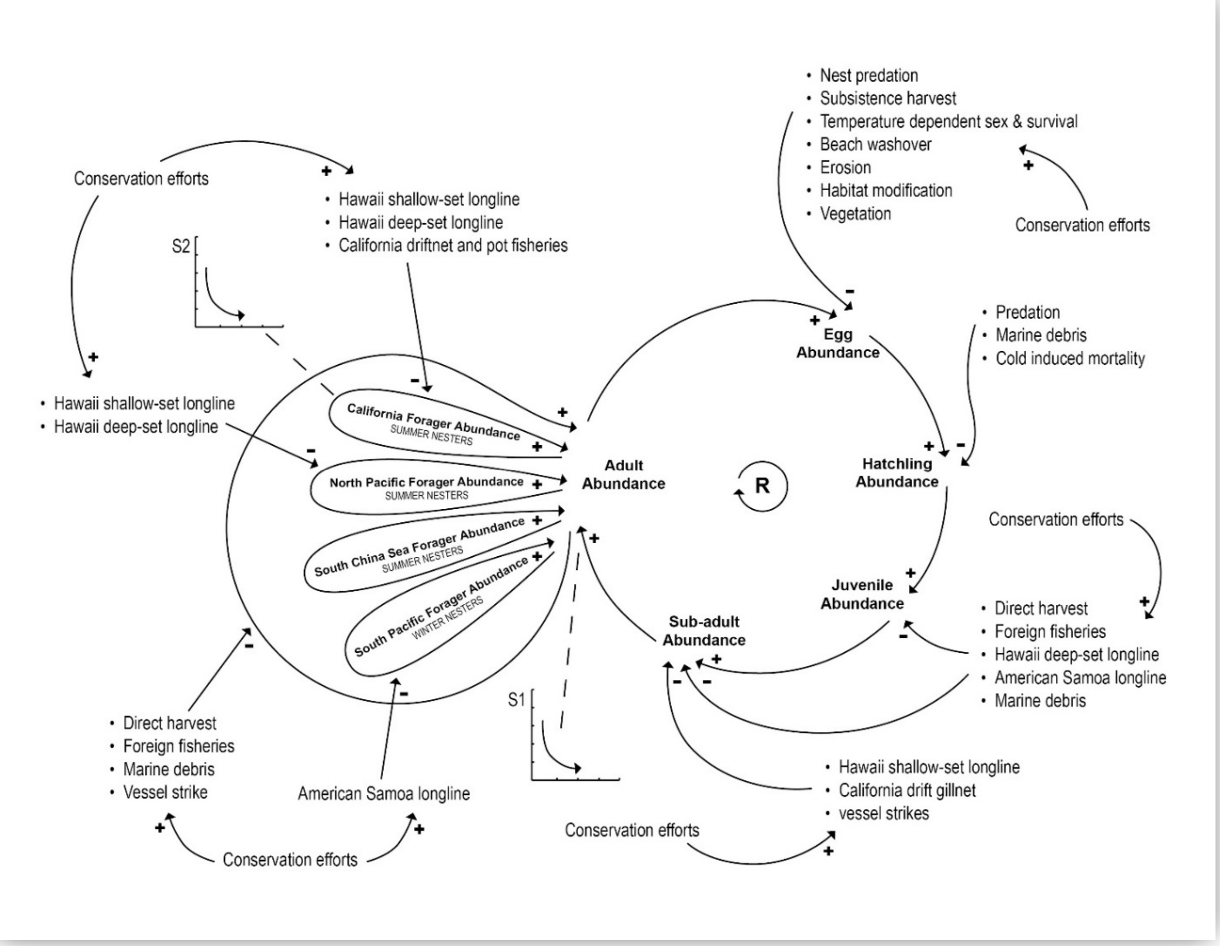


Figure 12. Leatherback sea turtle casual loop diagram depicting various stressor-response relationships for each life cycle stage.

4.2.2 North Pacific Loggerhead Sea Turtle

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). They are the only species of loggerhead sea turtle in the action area (76 FR 58868).

North Pacific loggerhead sea turtles appear to occupy an ecological setting that is distinct from other loggerheads, including those of the South Pacific loggerhead sea turtles. In general, the North Pacific loggerhead sea turtle is 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. 2004b; Peckham et al. 2007) and in the Western Pacific in the South and East China Seas and as far south as the Philippines and Malaysia (Limpus 2009; Kobayashi et al. 2008, 2011), 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 as cited in Conant et al. 2009); (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

North Pacific loggerhead sea turtles nest 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). North Pacific loggerhead sea turtles have also been documented in the Gulf of California, Mexico (Seminoff et al. 2004b; Zavala et al. 2017) and along the coast of southern California, during warm water periods (Allen et al. 2013; Eguchi et al. 2018).

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 juvenile loggerhead sea turtles may occur in the Gulf of Ulloa. After spending years foraging, potentially two decades (Tomaszewicz et al. 2015), 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. 2002b; Ishihara et al. 2011). The East China Sea is a major habitat for post-nesting adult females based on satellite telemetry and 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 (Parker et al. 2005b). 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. 2005b; 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 loggerheads) nesting in Japan and a southern hemisphere “population” that nests primarily in Australia (South Pacific loggerheads).

All loggerhead sea turtles in the action area are North Pacific loggerhead turtles. This species 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 (P. 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 (P. 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 11 and Table 12 and should be interpreted cautiously due to a small sample size.

Table 11. Flat priors mixed stock analysis for Loggerhead sea turtles (P. 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 12. Weighted Priors based on population size for loggerhead sea turtles (P. 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. While the underlying mechanism regarding the connectivity between oceanic and coastal foraging habitats remain unknown, Brisco et al. (in review) theorizes that a 'thermal corridor' exists that allows for 'pulsed' recruitment of loggerheads across an Eastern Pacific oceanic barrier during favorable environmental conditions. Evidence suggests this transport pathway opens intermittently with anomalously warm ocean conditions (e.g., El Nino events). Given that Eguchi et al. (2018) estimated 70,000 juvenile loggerhead turtles during El Nino conditions in 2015, but 2011 surveys resulted in no loggerhead sightings during a non-El Nino period, demonstrates high variability of loggerhead density in the region related to oceanographic conditions.

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 13) 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 species.

Table 13. 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 $\delta^{13}C$ and $\delta^{15}N$ 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 as cited in Seminoff et al. 2015; 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 2000; 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, 2003) 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; TEWG 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[2] 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. 2002c). 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 43,226 (CV = 0.51, 95% CI range = 1 s. 017 to 100,444) juvenile loggerhead sea turtles may occur annually in the Gulf of Ulloa, Baja California Peninsula. For purposes of this biological opinion, Jones estimated that there are approximately 328,744 juveniles (year 1-25) in the population and adding in adults results in a total population estimate of about 340,000 North Pacific loggerhead sea turtles (T. Jones pers. comm. 2019). Using the modeled estimate of 3,632 females nesting in Yakushima which represents 52% of nesting females, the total number of nesting females is 6,984. This assumes a remigration interval of 3

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[2] 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).

years between nesting, that each female lays three nests, and each nest has an emergence rate of 0.61, hatchling survival is 0.25 and aquatic survival is 0.4, juvenile survival is 0.858, and an age at first reproduction of 25 (Conant et al. 2009; Jones et al. 2012; Jones et al. 2018). 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; 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) 2009, 2010, 2012; Y. Matsuzawapers. 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 loggerheads 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 mathematical model to assist us in understanding the current status of North Pacific loggerhead sea turtles (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 are 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.

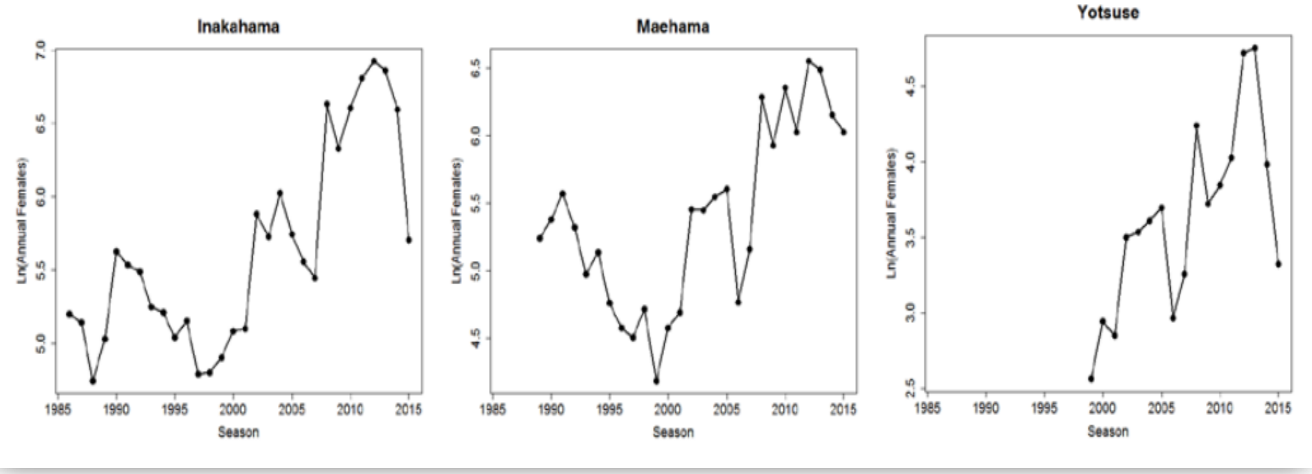

Figure 13. Natural log of annual nesting female loggerhead sea turtles from three nesting beaches in Yakushima, Japan (Source: Jones et al. 2018).

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).

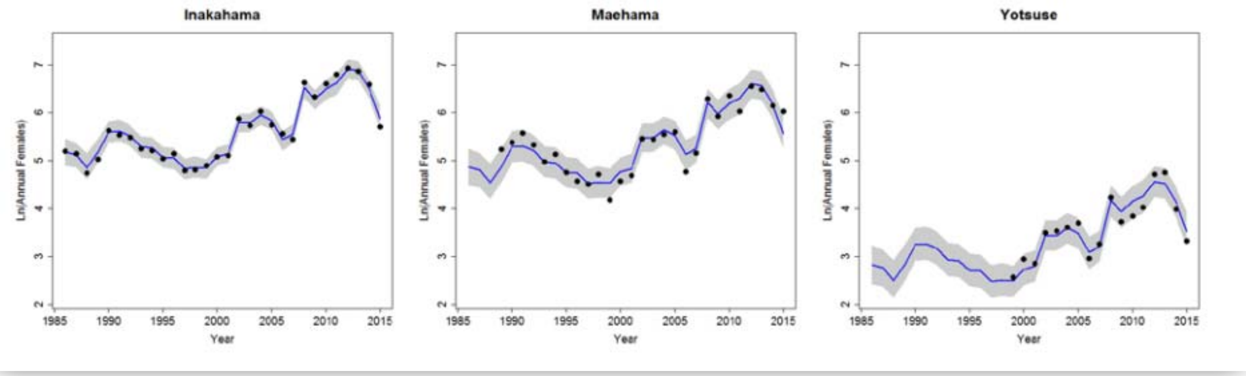

Figure 14. Predicted trends in nesting female loggerhead sea turtles from three nesting beaches in Yakushima, Japan (median (blue line) of the natural log of annual nesting females with 95% credible intervals (gray shading)) (Source: Jones et al. 2018).

Jones et al. (2018) 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 14.

Table 14. 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.

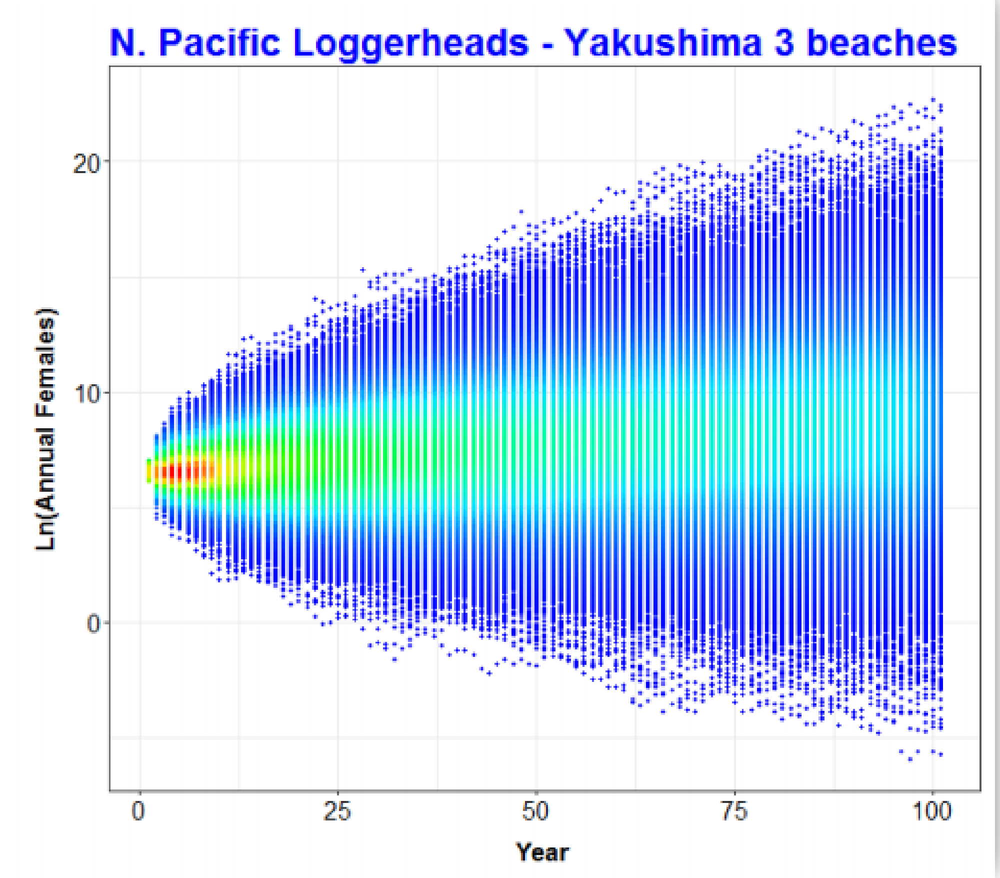

Figure 15. Model projections (10,000) of annual females in natural log space for 100 years into the future from 2015 (data year) (from Jones et al. 2018).

Next, 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 represented in Table 15. Whereas Table 16 lists the probability of the population reaching abundance thresholds at 5, 10, 25, 50 and 100 years from the data year (2015).

Table 15. 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 16. 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 (λ):

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 can be expected to 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 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 and Randall 1962; Fowler 1979; Lutcavage et al. 1997; Moulis 1997; Yerli et al. 1997; Allen et al. 2001; Parris et al. 2002; Limpus and Limpus 2003b; Blamires 2004; Donlan et al. 2004; Wetterer et al. 2014; Garmestani et al. 2005; Caut et al. 2006b (and references therein); Engeman et al. 2006, 2010; Ficetola 2008; Leighton et al. 2008; Tomillo et al. 2010; Whiting and Whiting 2011; Wyeneken et al. 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; Chaloupka and Limpus 2001; Dutton et al. 2005; Caut et al. 2006a, 2006b; Whiting et al. 2007b; Barton and Roth 2008; 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 and Barata 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 species 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 et al. 2002; Stewart and Wyneken 2004; Whelan and Wyneken 2007; Sim 2014; Brost et al. 2015; Silva et al. 2017); however, information specific to North Pacific loggerhead sea turtles 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 (Compagno 1984; Witzell 1987; Stevens and Lyle 1989; Marquez 1990; Cliff and Dudley 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; Simpfendorfer et al. 2001; Compagno 2001; Heithaus 2001; Heithaus et al. 2002a, 2002b, 2005, 2008; Sutherland and Sutherland 2003; Pitman and Dutton 2004; Fertl and Fulling 2007; Blumenthal et al. 2009; 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 North Pacific loggerhead sea turtles 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, and would not be an applicable threat. Whereas, interactions between species like crocodiles and North Pacific loggerhead sea turtles 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 North Pacific loggerhead sea turtles 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 species because of the proportion of beaches with shoreline armoring that prevents or interferes with the ability of nesting females to access suitable nesting habitat.

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). Fortunately, egg harvest in Japan ended in 1973 when laws were first enacted, with laws extending to all of the Yakushima prefecture by 1988 (Omuta 2006), and has not been a management challenge in Miyazaki since 1985 (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 turtles (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; NOAA 2013a; Senko et al. 2017). 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. (NOAA 2016, 2017).

Overutilization for commercial purposes in both Japan and Mexico was likely a factor that contributed to the historical declines of this species. High levels of bycatch in the pelagic drift gillnet fishery which operated in the Central North Pacific and resulted in thousands of loggerhead turtles interactions during their North Pacific migrations cannot be discounted (Wetherall et al. 1993). The greatest threat to the North Pacific loggerhead is likely bycatch of adult turtles in western Pacific fisheries operating in the Sea of Japana, and South and East China Seas (I. Kelly in consultation with Y. Matsuzawa 2018).

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 in combination with fin-fish bait), 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 2013). For example, switching to large circle hooks and mackerel bait in 2004 reduced the interaction rate by approximately 90% in the HI SSLL fishery (Swimmer et al. 2017). In 2003, NMFS implemented a time/area closure in southern California during forecasted or existing El Nino-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 Nino 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 t*hat 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). In 1989 the United Nations General Assembly placed a moratorium on the practice of pelagic drift net fishing, and in 1992 the UN banned the use of drift nets in international waters including the Taiwan and Japanese drift gillnet fishery which is believed to have captured thousands of loggerhead turtles during their North Pacific migrations (Wetherall et al. 1993).

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; Peckham et al. 2016). 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).

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 (Conant et al. 2009; Peckham 2014). 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).

Despite research and conservation efforts in Baja California between 2003 and 2012, the U.S. identified Mexico under the Magnuson Stevens Reauthorization Act for fishing interactions involving the bycatch of loggerhead sea turtles (NMFS 2013; Senko et al. 2017). As a result, Mexico had to demonstrate and implement a regulatory program to reduce sea turtle bycatch. In September 2016, NMFS issued a positive determination based on Mexico new regulatory program (NMFS 2016, 2017). These regulations establish a temporary fisheries reserve (authorized for two years) that includes monitoring of bycatch, a loggerhead bycatch mortality cap, temporal and spatial restrictions on fishing gear and practices, and a closure of all finfish fisheries during the summer of 2016. As a result, turtle bycatch was likely substantially reduced in 2016, but there were significant economic cost to artisanal fishers and it is unclear if bycatch reductions persist given the economic burdens of these regulatory measures (Senko et al. 2017).

In Japan, due to concerns of high sub-adult and adult loggerhead sea turtle mortality in mid-water coastal pound nets, researchers with the STAJ, ProCaguama, local fisherman, and NMFS have collaborated 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 four promising gear mitigation options for pound net fisheries were discovered (Ishihara et al. 2012a). Pound-net mitigation activities stalled in 2012, but continued collaborative efforts are needed 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. 2012b).

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); although these structures have since been replaced (I. Kelly in consultation with Y. Matsuzawa 2018). The Ministry of Environment has supported the local NGO, Yakushima Umigame Kan, to conduct turtle surveys and protect nests on Yakushima Island by 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). The cessation of commercialized and subsistence loggerhead turtle egg harvest in 1973 was a significant conservation measure (Omuta 2006) that has likely had significant effect on the increasing trend of the population (Y. Matsuzaw pers. comm.)

In 2003, and extending through 2012, the WPRFMC 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 included conducting nightly patrols during the summer nesting season to relocating 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 (WPRFMC 2015). STAJ has developed techniques for nest relocation that 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 WPRFMC 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 WCPFC adopted a CMM (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 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 for vessels that fish in a shallow-set manor and target swordfish. 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. While these methods have been proven successful in reducing sea turtle interactions, we note that a recent analysis of the number of vessels that actually use these measures is only a small percentage of the overall longline effort in the Pacific. During a workshop convened to assess the effectiveness of WCPFC's Sea Turtle CMMs, they found that the measure applies to approximately one percent of longline fisheries in the Convention Area (ABNJ 2017). The remaining effort is from vessels that deep-set or shallow-set and target species other than swordfish (ABNJ 2017). A new measure was adopted in 2018 (CMM 2018-04) that extends the measure to all vessels that fish in a shallow-set manner (WCPFC 2018b).

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 2007c; Ishihara et al. 2012a, 2012b; Swimmer et al. 2017; Senko et al. 2017).

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 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 show 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 (however, the best estimate or mean is 49 years, and the range is 2-99 years. In contrast to leatherback sea turtles, NMFS estimates suggest there is only a 25% probability of falling below this threshold abundance value in 49 years, whereas there is a 98% probability that leatherback sea turtles will fall below the threshold value of half its current abundance in 26 years (see Table 15 and Table 8; 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, 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 loggerhead sea turtles. 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.

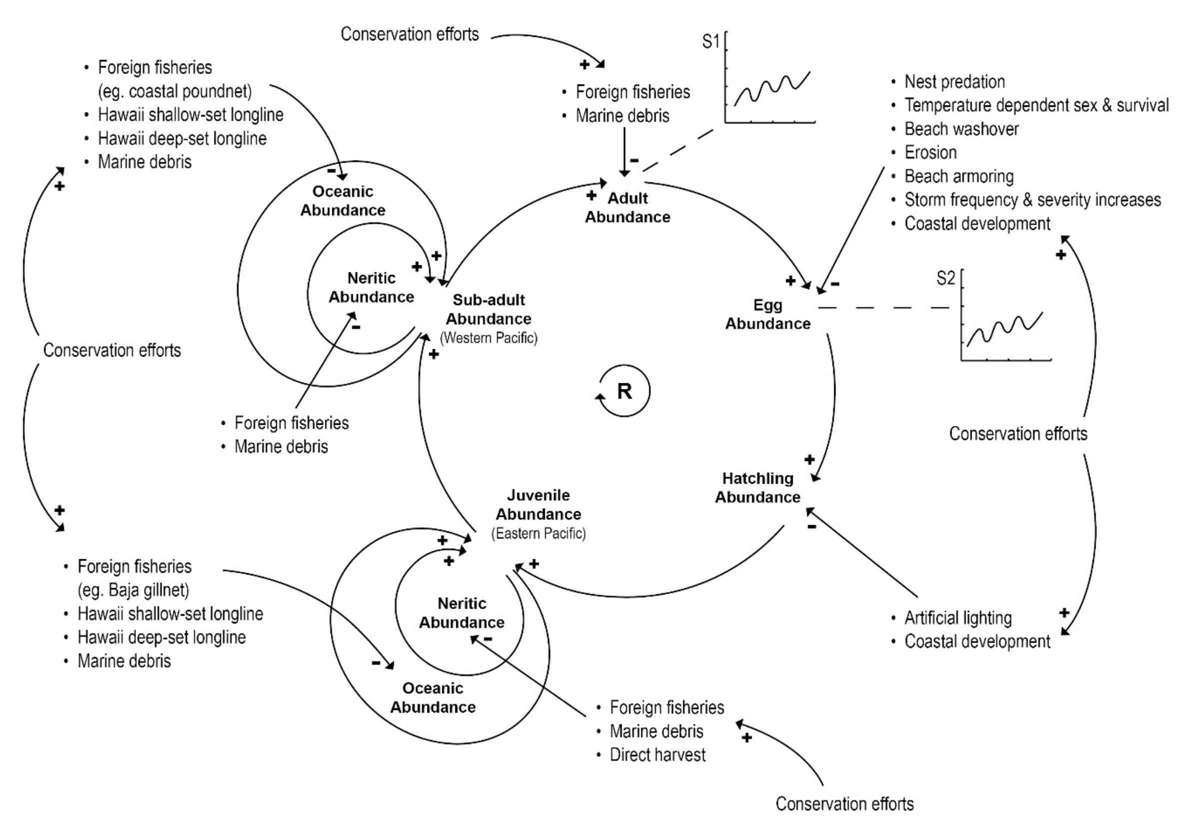

Figure 16. North Pacific loggerhead sea turtle casual loop diagram depicting various stressor-response relationships for each life cycle stage

4.2.3 Green Sea Turtle

Green sea turtles were listed as threatened on July 28, 1978 (43 FR 32800), except for breeding populations that occur in Florida and the Pacific coast of Mexico, which were listed as endangered. On April 6, 2016, NMFS and the FWS published a final rule finding that the green sea turtle is composed of 11 DPSs (Figure 17) that each qualify as a “species” for purposes of the ESA. The Services removed the current range-wide listing and, in its place, listed eight species of green sea turtle as threatened and three as endangered. The species of green sea turtles most likely to occur in the range of the HI SSLL fishery are limited to those that occur in the Pacific Ocean.

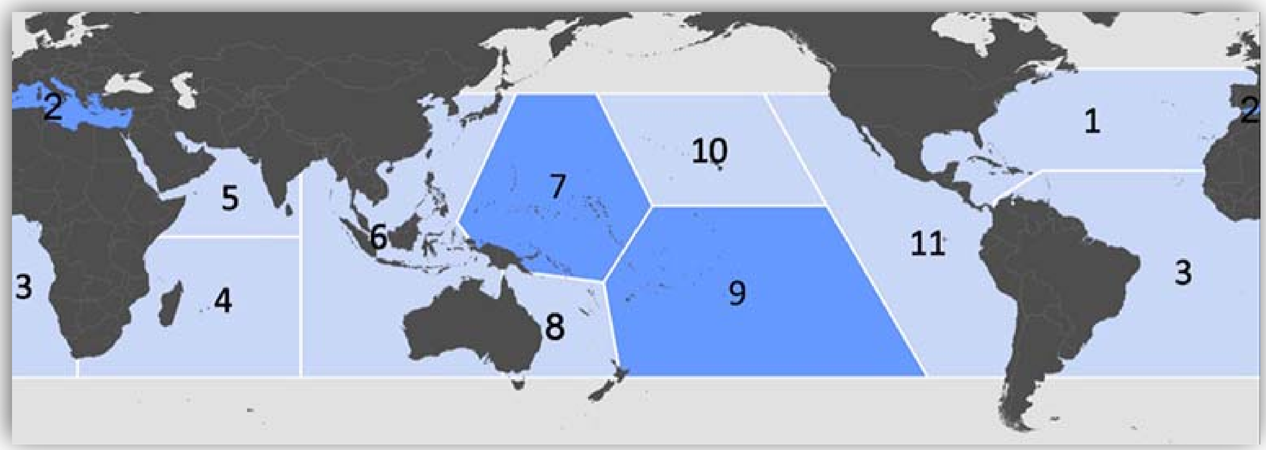

Figure 17. Overview of listed green sea turtle species; (1) North Atlantic; (2) Mediterranean; (3) South Atlantic; (4) Southwest Indian; (5) North Indian; (6) East Indian-West Pacific; (7) Central West Pacific; (8) Southwest Pacific; (9) Central South Pacific; (10) Central North Pacific; and (11) East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

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 (P. Dutton pers. comm. August 31, 2016; see Table 17 for summary information from these assessments). Due to the very small sample sizes we cannot determine the percentages of turtles from each of these green turtle species 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 species in consideration, we used the results from the genetic analysis cautiously. Because the sample size is small and we cannot determine the percentages of green sea turtles from each species with great certainty (as evidenced by the large CI), our analyses considered the 95% CI to minimize the chances of underestimating the risks the species face.

Table 17. Genetic composition of green turtles sampled from the Hawaii shallow-set longline fishery (P. 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.

Species (ID #)	Number of genetic samples	MSA mean %	95% CI
East Pacific (11)	11	55	32-77
Central North Pacific (10)	8	41	20-64

Species (ID #)	Number of genetic samples	MSA mean %	95% CI
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

In the following sections, we describe the general pelagic distribution and behaviors common to the adult and juvenile life stages of the six species of green sea turtles considered in this biological opinion. Subsequent sections describe the status of each individual green sea turtle species in detail.

General Developmental Distribution of Pacific Green Sea Turtles

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 25), which we describe in more detail later in this section. The six species of green turtles discussed in this biological opinion comprise a mosaic of species, each with unique nesting sites and, in many cases, possessing disparate demographic features (e.g., mean body size, age at first reproduction; NMFS and FWS 2007a). However, despite these differences, green turtles from different species often mix in common foraging grounds, thus creating unique challenges when attempting to distinguish the various species (Jensen et al. 2013), and this is particularly true of the various green sea turtle species that may be exposed to fisheries in the Pacific Ocean.

The six species of green sea turtle considered in this biological opinion likely spend most 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 Nino Southern Oscillation events (Carballo et al. 2002).

Distinctive features of morphology, behavior, or life history that might indicate important local adaptations are documented for turtles from various areas including the 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). Different life history and distributional patterns, status and threats for the following green turtle species that occur in the action area are later in this biological opinion.

General Adult Foraging and Diving Behaviors of Pacific Green Sea Turtles

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 (Quinones 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 (Quinones et al. 2010).

Seminoff et al. (2002a) studied the diet of green turtles at Bahía de los Angeles in the Gulf of California, Mexico. 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 (Lavin 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 percent 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. (1975) 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 Central North Pacific green turtles has been comprehensively reviewed by Mortimer (1982) and Bjorndal (1985) and although the studies were conducted on the Central North Pacific green sea turtles, we discuss them here because they represent some of the most robust information that we have on green sea turtle diet in the Pacific. We expect that it is generally true for all green sea turtle species considered in this biological opinion. 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 French Frigate Shoals (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 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 species is present, we provide that additional information later.

General Information on Pacific Juvenile cohorts - a comprehensive evaluation

The following is general information that is likely applicable to every species of green sea turtle in this biological opinion, even though some of the information come from studies done on green turtles in the Atlantic. 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 as cited in Seminoff et al. 2015). 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; 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 FWS 2007a; 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; Lopez-Mendilaharsu et al. 2005; Amorocho and Reina 2007; Carrion-Cortez et al. 2010).

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 Green Sea Turtles

Differential Distribution

Central North Pacific green sea turtles occur in the Hawaiian Archipelago and Johnston Atoll (Figure 18). The Hawaiian Archipelago is the most geographically isolated island group and, therefore, these green sea turtles 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 sea turtles recruit to coastal habitats. Average size at recruitment to these neritic habitats for Pacific green sea turtles ranges from 35-50 cm CCL (Balazs 1980; Limpus et al. 2003).

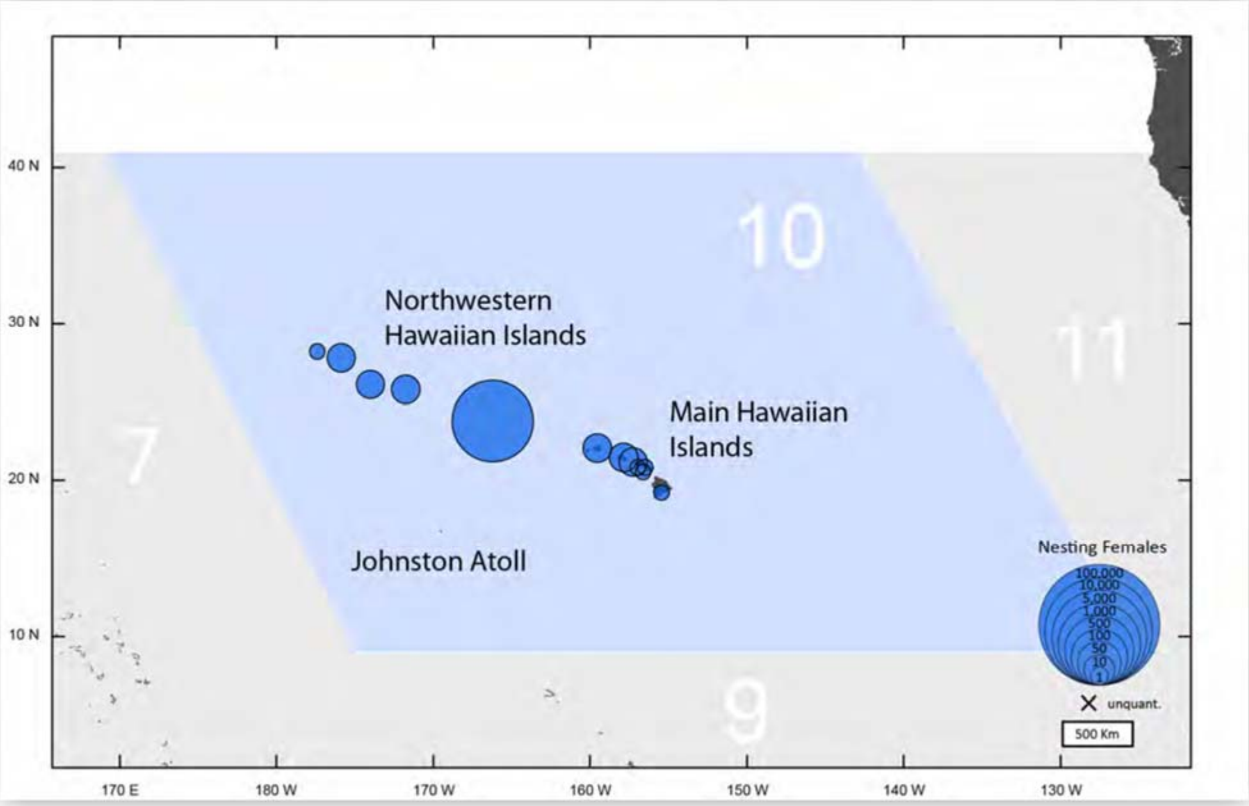

Figure 18. Geographic area of the Central North Pacific green sea turtle. Size of circles indicates estimated nester abundance (see Section 14.2.1). The geographic range of this species of green sea turtle encompasses the entire Archipelago of Hawaii and Johnston Atoll.

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 Central North Pacific green sea turtles (Figure 18).

Adult Foraging and Diving

See *General Adult Foraging and Diving Behaviors of Pacific Green Sea Turtles* above.

Population Dynamics

Structure

Central North Pacific green sea turtles are spatially and genetically separated from other green sea turtle species in the Pacific Ocean (Seminoff et al. 2015). These observations, with other data led NMFS and FWS to conclude that the Central North Pacific green sea turtle is a discrete species (81 FR 91097).

Based on spatially concentrated (limited) distributions of nesting and lack of evidence of genetic substructuring, Seminoff et al. (2015) concluded that the Central North Pacific green sea turtle consists of a single population. The species nests primarily on one site (FFS, which representing 96% of the species), and although nesting has been observed at 12 other sites it occurs in very small numbers (Seminoff et al. 2015). Because the species nests in one concentrated location Seminoff et al. (2015) expressed concern for how it could be affected by "environmental uncertainty" like natural catastrophes. Then in October 2018, Hurricane Walaka decimated East Island where approximately 50% of the species nests. At the time of this writing, NMFS is unclear what impacts will be seen in the species from this event.

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. NMFS and FWS (1998a) 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. 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) (NMFS and FWS 1998a).

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 species of green sea turtle. Demographic studies of green sea turtles do not reveal any structuring of traits within the species, although variable ecosystem productivity has produced differences in body conditions of nearshore foraging turtles (Balazs and Chaloupka 2004a; 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 of Central North Pacific green sea turtles occurs from May through August (1980), and nesters return to breed at an interval of 4 years (G. Balazs pers. comm. 2013). Nest-level inventories are incomplete for this species, 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 sea turtles in this species begin breeding on average at 23 years (K. Van Houtan 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 species.

Genetic sampling in the Central North Pacific green sea turtle has been extensive and representative given that there are few nesting populations in this region. Results of 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 species. Within the NWHI, studies show no significant differentiation (based on mtDNA haplotype frequency) between FFS and Laysan Island (P. Dutton 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 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 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 species. With nesting in Central North Pacific green sea turtles 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 species 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 species, lies within the Papahānaumokuākea 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 diversity among nesting sites in the Central North Pacific green sea turtle based on a total of four closely related mtDNA haplotypes identified (P. Dutton pers. comm. 2013).

From 1965 to 2013, 17,536 green turtles have been tagged including 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 19). Information from tagging at FFS, other areas in the NWHI, areas in the 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.

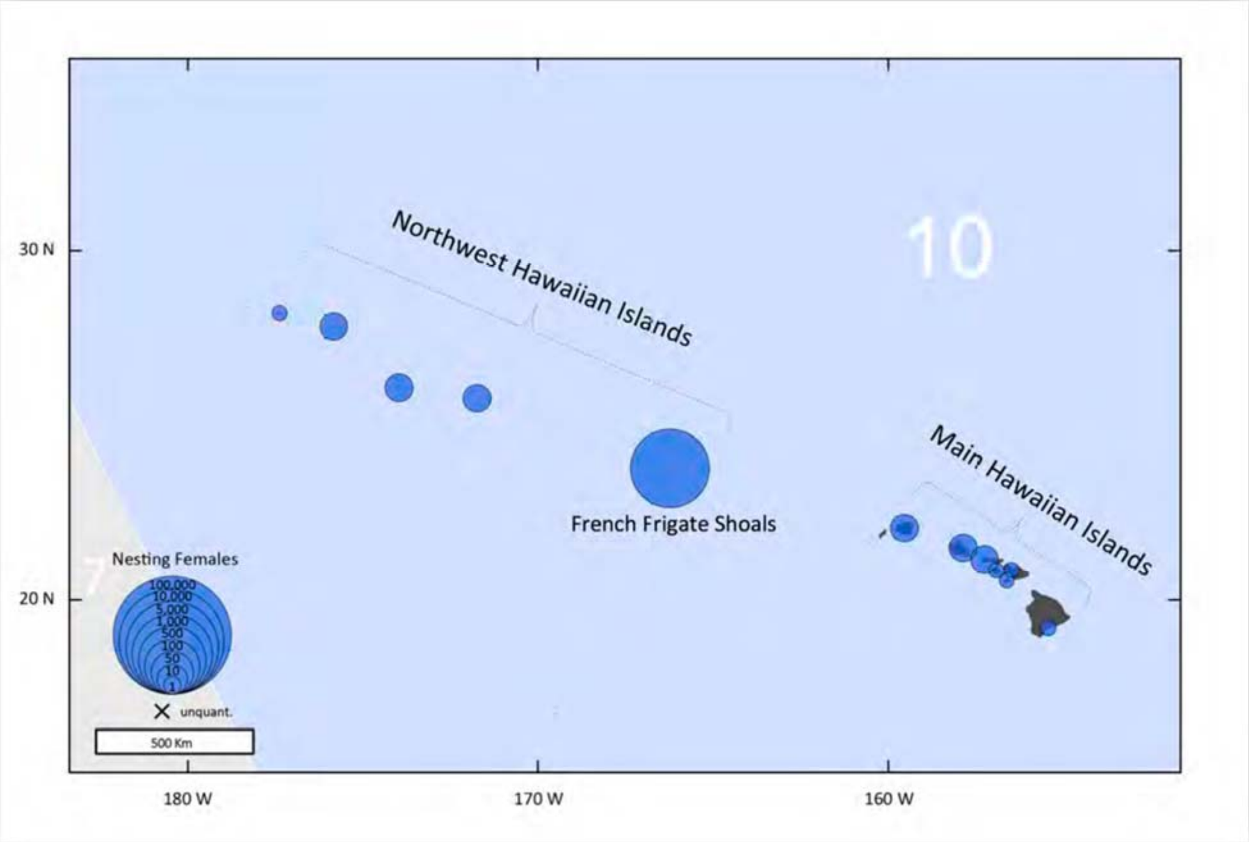

Figure 19. Close up of nesting green turtles in the Central North Pacific green sea turtle. Size of circles indicates estimated nester abundance.

Status

The Central North Pacific green sea turtle is listed as threatened. The species exhibits low nesting abundance. At the time of the most recent status review nester abundance was 3,846 nesting females at 13 nesting sites. Since initial nesting surveys at the FFS index beach in 1973, Central North Pacific green sea turtle nesting has been increasing (Balazs and Chaloupka 2004b); however nesting site diversity is extremely limited within 96% of nesting on one low-lying atoll (FFS).

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 (Chaloupka and Balazs 2007; IUCN 2012). The historic population size is estimated to be approximately 320,000 individuals (IUCN 2012). Increases in the past 40 years correspond 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 as cited in Seminoff et al. 2015). 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 and Chaloupka 2004b; Chaloupka et al. 2007). In addition, there has been a dramatic increase in the number of basking turtles in the MHI and throughout the NWHI (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, 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 2004b).

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 species. Survival of this Central North Pacific green sea turtle 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 this species' future and making it highly vulnerable to stochastic events. As much as 80% of historically major nesting populations were 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 Central North Pacific green sea turtle makes 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 Central North Pacific green sea turtles. Central North Pacific green sea turtles 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, support efforts to recover and protect this species, and 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 as cited in Seminoff et al. 2015). 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 Papahānaumokuākea 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 Papahānaumokuākea 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 Monument. 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 apply to Central North Pacific green turtles. 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 species.

In 2008, the Western and Central Pacific Fisheries Commission issued a CMM (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 Central North Pacific green sea turtles (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 species 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 turtles 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 species (Baker et al. 2006). The primary nesting area for the Central North Pacific species is threatened by sea level rise. For example, Whale-Skate Island at FFS was formerly a primary green turtle nesting site but the island has subsided and is no longer available for nesting (Kittinger et al. 2013).

Threats to Central North Pacific green sea turtles habitat in neritic and/or oceanic zones 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. Development and other activities that affect the quality of coastal habitats in the main Hawaiian Islands used by Central North Pacific green sea turtles are expected to continue and possibly increase with an increasing human population and annual influx of millions of tourists. Such activities include marina construction, beach development, siltation of forage areas, contamination of forage areas from anthropogenic activities, resort development or activities, and increased vessel traffic (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 Central North Pacific green sea turtles. 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 green turtles caught in the HI SSLL fishery are likely Central North Pacific green sea turtles (P. Dutton pers. comm. November 20, 2017). This means that 2-6 turtles from this species are estimated to have been captured by the HI SSLL fishery since 2004. NMFS estimates that between 4-43% of the green sea turtles caught in the HI DSLL fishery are likely Central North Pacific green sea turtles (P. Dutton pers. comm. June 29, 2018). This means an estimated 3 to 29 Central North Pacific green sea turtles to have been captured in the HI DSLL longline fishery since 2004. The American Samoa longline fishery does not interact with Central North Pacific green sea turtles based on the genetic analysis and what we know about the migrations and foraging behavior of this species.

4.2.3.2 East Pacific Green Sea Turtle

Differential Distribution

The East Pacific green sea turtle extends from the California/Oregon border, USA (42°N) southward along the Pacific coast of the Americas to central Chile (40°S). The species ranges from the aforementioned locations in the USA and Chile to 143°W and 96°W, respectively. This species encompasses the Revillagigedos Archipelago (Mexico) and Galapagos Archipelago (Ecuador).

Adult Foraging and Diving

See *General Adult Foraging and Diving Behaviors of Pacific Green Sea Turtles* 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 sea turtles (Hays et al. 2000) and the average depth where green sea 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 species and other turtle species.

Population Dynamics

Structure

East Pacific green sea turtles are spatially and genetically separated from other green sea turtle species in the Pacific Ocean (Seminoff et al. 2015). These observations, with other data led NMFS and FWS to conclude that the Central North Pacific green sea turtle is a discrete species (81 FR 91097).

At least 39 nestings locations are documented for this species, with primary concentrations in Mexico, Ecuador, and Costa Rica (Seminoff et al. 2015). At least four populations comprise the species as identified by genetic data: two in Mexico, one in Costa Rica, one in the Eastern Pacific, and one in the Galapagos (Dutton unpubl. data as cited in Seminoff et al. 2015). Male-mediated gene flow between the Galapagos and Mexico populations appears to be limited (Roden et al. 2013 as cited in Seminoff et al. 2015).

Relative to other green sea turtles, East Pacific green turtles are known to mature at smaller sizes (60–77 cm SCL) (Seminoff et al. 2002b) 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. (2002b) 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 in the East Pacific green sea turtle. Among all nesting assemblages in the East Pacific green sea turtle, 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 as cited in Seminoff et al. 2015). The internesting interval ranges from 9 to 18 days (Witherington and Ehrhart 1989; 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 as cited in Seminoff et al. 2015). Remigration intervals have been reported between 2 and 3-years (Witherington and Ehrhart 1989; Zurita et al. 1994; Troeng and Chaloupka 2007).

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, 2002, 2003, 2005, 2006, 2008 as cited in Seminoff et al. 2015; Guzman-Hernández and G. Alvarado 2009, 2010 and 2011 as cited in Seminoff et al. 2015) and in Tortuguero, Costa Rica was 108 (Tiwari et al. 2006). Hatching success rate was 92% in Mexico (Xavier et al. 2006 as cited in Seminoff et al. 2015), 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 this species . The East Pacific green sea turtle has a very broad nesting range, with nesting occurring from the tip of the Baja California Peninsula to Northern Peru (Figure 20). Such a broad latitudinal range may be advantageous to this species in the face of global climate change, sea level rise, increased storm events and loss of nesting habitat in vulnerable areas. The East Pacific green sea turtle exhibits year round nesting at several sites and non-overlapping nesting seasons at others. Thus, it appears that this species may benefit from nesting season temporal diversity in relation to population resilience. The East Pacific green sea turtle 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 species' range. With respect to genetic diversity, there is significant genetic substructuring within this species, perhaps suggesting a level of resilience to population genetic bottlenecks (Seminoff et al. 2015).

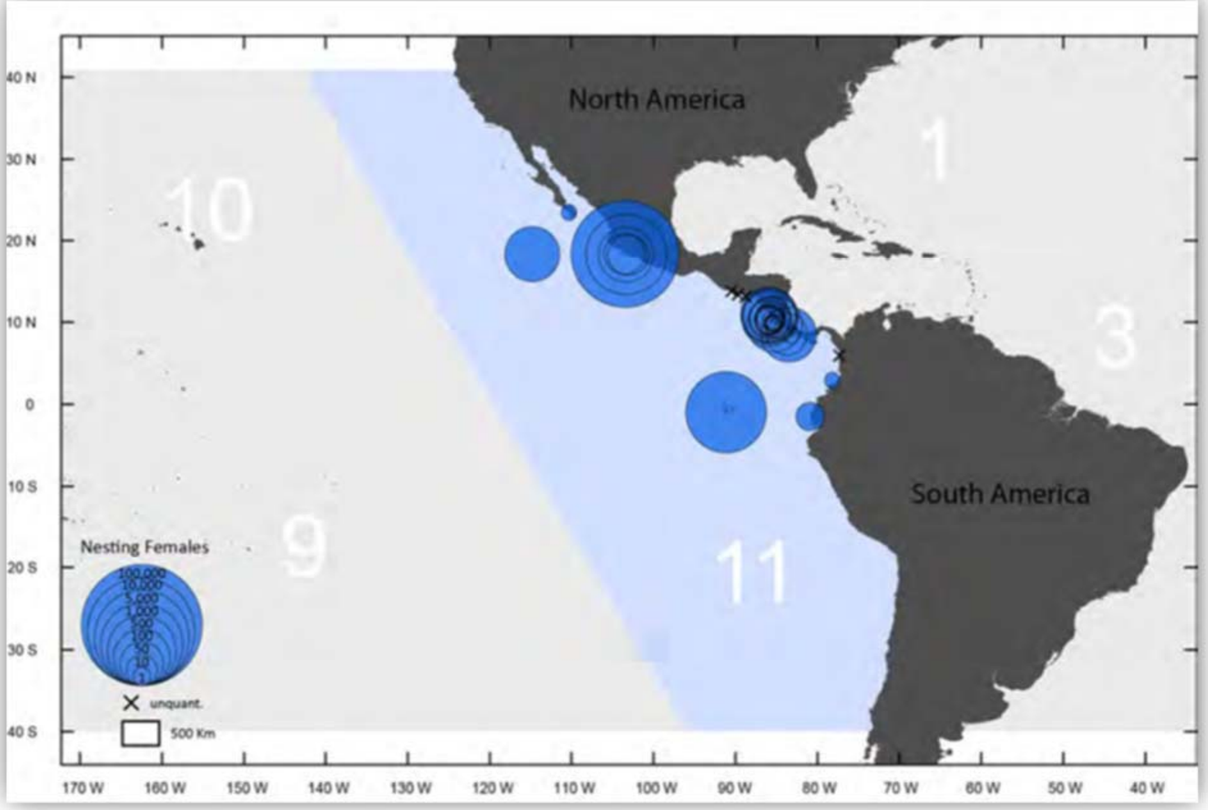

Figure 20. Nesting distribution of East Pacific green sea turtles (blue-shaded area marked with '11'). Size of circles indicates estimated nester abundance.

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 green sea turtles occur within the action area. A total of 19 green turtle samples were collected by observers from 1996-2017 in the HI SSLL fishery and were sequenced for assignment to species (Dutton et al. 2017). Eleven turtles were assigned to the Eastern Pacific green sea turtle.

Seminoff et al. (2015) identified 39 total nesting sites for which abundance information is available. There are sporadic nesting events in many other areas within the range of the East Pacific green sea turtle, 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 of this species 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 sea 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 (Rostral 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 East Pacific green sea turtle is listed as threatened. Seminoff et al. (2015) ranked the species as having a low risk of extinction based on nesting abundance. 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 estimated to comprise about 58% of the total adult females for the species) East Pacific green turtle nesting appears to have increased since the population's low point in the mid-1980s. This observed increase may have resulted from the onset of nesting beach protection in 1979, as is suggested by the similarity in timing between the onset of beach conservation and the age-to-maturity for green turtles in Pacific Mexico. Similarly, data from the Galapagos Archipelago suggest that this population may be increasing.

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 range of the East Pacific green sea turtle.

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 as cited in Seminoff et al. 2015).

In 2008, the Western and Central Pacific Fisheries Commission issued a CMM (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. 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 as cited in Seminoff et al. 2015) at beaches in Costa Rica, but this is not a major problem for the species.

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 species.

The best available data suggest that fibropapillomatosis does not pose a current threat to the East Pacific green sea turtle (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 as cited in Seminoff et al. 2015). 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). Lastly, Resendiz et al. (2019) reported a sub adult female with a lesion from Baja California Sur, Mexico.

Anthropogenic Threats

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

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 species, 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 as cited in Seminoff et al. 2015). Since then, bycatch has not been thoroughly evaluated but it is widely expected that most fishers either improperly implement TEDs or remove them entirely from their trawls.

The HI SSLL fishery interacted with a total of ten green turtles (zero mortalities) from 2004-2018 (NMFS unpublished data). Based on genetic samples form 19 green sea turtles captured since the 1990s, NMFS estimates that between 32 and 77% of the turtles caught in the HI SSLL fishery are likely East Pacific green sea turtles (P. Dutton pers. comm. November 20, 2017). This means that between 3 and 8 turtles from this species have been captured by the HI SSLL fishery since 2004. The HI DSLL fishery interacted with approximately 71 green turtles (65 estimated mortalities) between 2004 and 2018 (NMFS 2019b). Based on genetic samples from 31 green sea turtles, NMFS estimates that between 31 and 80% of the turtles caught in the HI DSLL fishery may have been East Pacific green sea turtles (P. Dutton pers. comm. June 29, 2018). This means that an estimated 22 to 57 East Pacific green sea turtles were captured in the HI DSLL fishery since 2004. The American Samoa longline fishery interacted with approximately 286 green turtles (with 259 estimated mortalities) between 2006 and 2018 (NMFS 2019a). Based on genetic samples from 31 green sea turtles, NMFS estimates that between 2 and 23% may have been East Pacific green sea turtles (P. Dutton pers. comm. July 5, 2018). This means that an estimated 5 to 66 East Pacific green sea turtles were captured by the American Samoa longline fishery since 2006.

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

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 Green Sea Turtle

Differential Distribution

The Central South Pacific green sea turtle is distributed north from northern New Zealand to Fiji, Tuvalu, and Kiribati and east to include French Polynesia. 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 species range 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 *General Adult Foraging and Diving Behaviors of Pacific Green Sea Turtles* above for general information on green turtle feeding and diving behavior.

Central South Pacific green sea turtles departing nesting grounds travel throughout the South Pacific Ocean. Post-nesting 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

Central South Pacific green sea turtles are spatially and genetically separated from other green sea turtle species in the Pacific Ocean (Seminoff et al. 2015). These observations, with other data led NMFS and FWS to conclude that the Central South Pacific green sea turtle is a discrete species (81 FR 91097). The Central South Pacific green sea turtle is comprised of at least two distinct populations based on genetics, although genetic sampling has been limited and given that nesting is broadly spatially separated and documented at on 57 different sites (Seminoff et al. 2015) additional substructuring to the species may be present that is not yet known.

Central South Pacific green sea turtle nesting is geographically widespread at low levels. The most abundant nesting area is Scilly Atoll, French Polynesia, which in the early 1990s hosted 300–400 nesters annually (Balazs et al. 1995b). 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 FWS 1998a; 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 species, 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 species. However, the largest nesting site, Scilly Atoll, which comprises roughly one third of the entire nesting abundance, was last monitored in the early 1990s (Balazs et al. 1995b) and has reportedly significantly declined in the past 30 years as a result of commercial exploitation (Conservation International Pacific Islands Program 2013). 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 species likely has fewer than 3,600 nesters. However, data are insufficient to develop trends for the Central South Pacific green sea turtle. Partial and inconsistent monitoring on the largest nesting site, 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).

Genetic sampling in the Central South Pacific green sea turtle 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 (Seminoff et al. 2015). Based on limited sampling, there is evidence of significant spatial structuring. There appears to be significant population substructuring in the Central South Pacific green sea turtle (F_{st}[3] 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 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 species, 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 species. Limited demographic information is available for Central South Pacific green sea turtles. 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. 1995b). 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).

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

Diversity

The Central South Pacific green sea turtle has a broad geographical range, although 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 Central South Pacific green sea turtle nesting is in remote low-lying atolls, at low abundance levels and low spatial densities.

Nesting occurs sporadically throughout the geographic distribution of the population, with isolated locations having relatively low to moderate nesting activity (Figure 21). Central South Pacific green sea turtles departing nesting grounds 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.

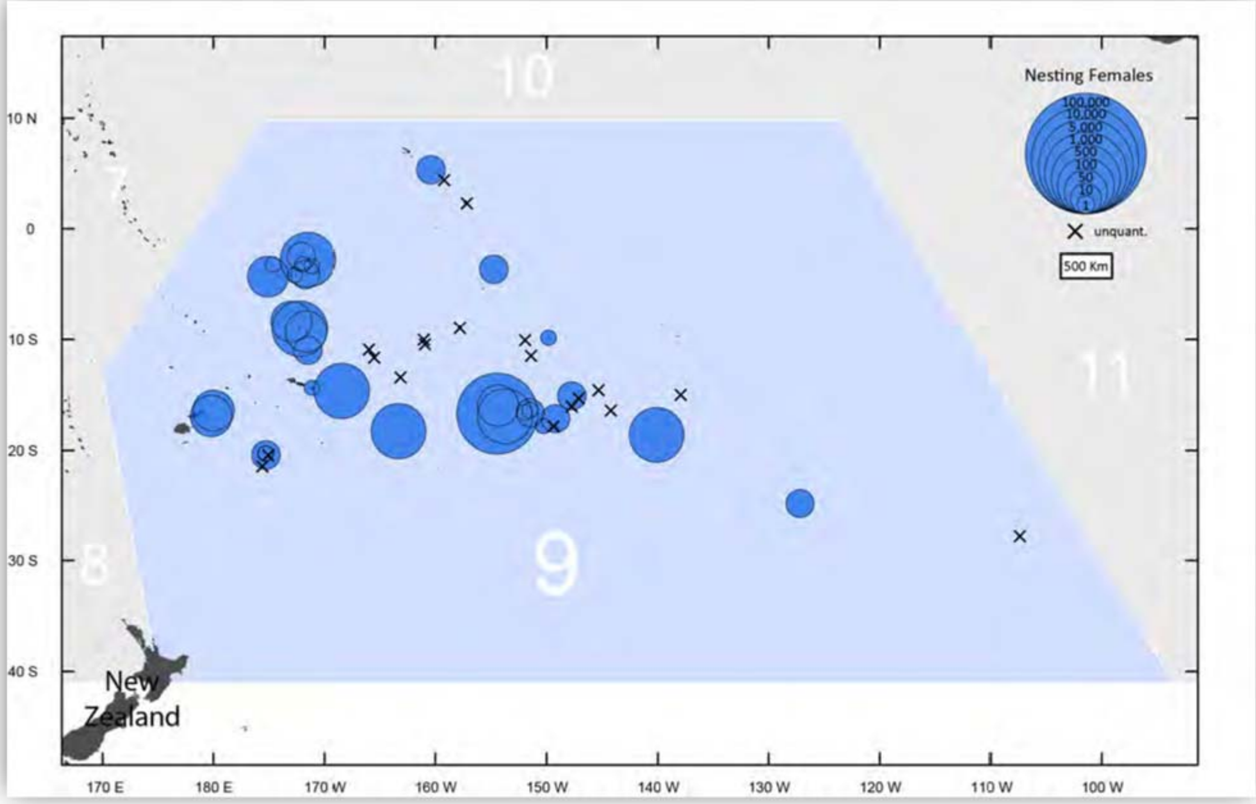

Figure 21. Nesting distribution of Central South Pacific green sea turtles. Size of circles indicates estimated nester abundance (see Section 13.2.1). Locations marked with 'X·' indicate nesting sites lacking abundance information.

Status

The Central South Pacific green sea turtle is listed as endangered. Although population trends are not known because no nesting sites have standardized monitoring that span entire nesting seasons, declines at the largest nesting site (Scilly Atoll) from persistent and illegal commercial harvesting (Petit 2013), and low nesting diversity provides this species with little resilience against current threats. 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 SRT estimated that this species likely has fewer than 3,600 nesters and during their assessment of risk, the SRT concluded the species had a 62% probability of having a greater than 1% extinction risk of extinction over the next 100 years (Seminoff et al. 2015).

Threats to the Species

Natural Threats

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

Anthropogenic Threats

Directed take in the marine environment has been a significant source of mortality in American Samoa (Tuato'o-Bartley et al. 1993; NMFS and FWS 1998a), 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. While 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), actual levels are unknown. South Central Pacific green sea 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 green sea 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 South Central Pacific green sea turtles.

Turtles have been traditionally harvested for food and shells in Samoa and have cultural significance (Craig 1993). 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 U.S. $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 as cited in Seminoff et al. 2015). 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 were 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 et al. 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 South Central Pacific green sea turtles. 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 31 and 73% of the turtles caught in the American Samoa longline fishery could be Central South Pacific green sea turtles (P. Dutton pers. comm. July 5, 2018). This means that the American Samoa longline fishery captured between 83 and 197 South Central Pacific green sea turtles since 2006. It is unclear exactly how many Central South Pacific green sea turtles are taken in other longline fisheries, however it is estimated that over 200 South Central Pacific green sea turtles may be killed annually by longline fishing around American Samoa bounded by 180° and 155° W longitude, and 3° S–32° S latitude (Maison et al. 2010).

In Fiji, South Central Pacific green sea 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 as cited in Seminoff et al. 2015) reports that green sea 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 South Central Pacific green sea turtles is unclear. The most recent data, provided by Fiji to the WCPFC (2016a, 2018c), reports 79 green sea turtles have been captured between 2011 and 2017. Of those 79 total turtles, 44 were deceased (56%; WCPFC 2016a, 2018c).

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 as cited in Seminoff et al. 2015), although it is unclear how effective these regulations are. McCoy (2008 as cited in Seminoff et al. 2015) 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 and 5% of the green sea turtles caught in the HI SSLL fishery could be Central South Pacific green sea turtles (P. Dutton pers. comm. November 20, 2017). This means that the HI SSLL fishery may have captured one South Central Pacific green sea turtle since 2004. NMFS estimates that between 0 and 24% of the green sea turtles caught in the HI DSLL fishery may be Central South Pacific green sea turtles (P. Dutton pers. comm. June 29, 2018). This means the HI DSLL fishery captured between 0 and 16 Central South Pacific green sea turtles since 2004.

4.2.3.4 Central West Pacific Green Sea Turtle

Differential Distribution

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

Adult Foraging and Diving

See *General Adult Foraging and Diving Behaviors of Pacific Green Sea Turtles* above for general information on green turtle feeding and diving behavior.

In many areas, Central West Pacific green sea turtle foraging areas coincide with fishing zones and urban development. Central West Pacific green sea turtles forage on seagrass beds around Saipan (Kolinski et al. 2001), Tinian (Kolinski et al. 2004), and Rota (Kolinski et al. 2006) Islands. Guam's coral reefs likely also provide food sources for turtles (NMFS and FWS 1998a). 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

Central West Pacific green sea turtles are spatially and genetically separated from other green sea turtle species in the Pacific Ocean (Seminoff et al. 2015). These observations, with other data led NMFS and FWS to conclude that the Central West Pacific green sea turtle is a discrete species (81 FR 91097). This species is characterized by a small nesting population spread across a large geographic area, and is dominated by insular nesting. It is unclear how many populations comprise this species, but genetic analyses indicate that rookeries separated by more than 1,000 km were significantly different from each other, while rookeries within 500 km show no genetic differences (Seminoff et al. 2015). At a minimum the species appears to be comprised of at least seven populations based on mtDNA analyses by Dutton et al. (2014 as cited in Seminoff et al. 2015).

Demographic data availability is limited and somewhat variable for nesting sites in this species. 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).

Satellite telemetry shows that Central West Pacific nesting females migrate to areas within and outside of the species. 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

Central West Pacific green sea turtles cover a large geographic area and 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 this area, 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 travel into the range of the East Indian/West Pacific green sea turtle. Although 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.

Central West Pacific green sea turtle nesting occurs at low levels throughout the geographic distribution of the population, with isolated locations having high nesting activity (Figure 22). The highest numbers of nesting females 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 FWS 1998a; 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).

Central West Pacific green sea turtles departing nesting grounds 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 as cited in Seminoff et al. 2015). 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 range of the species. Aerial sea turtle surveys show that an in-water population exists around Guam (DAWR 2011 as cited in Seminoff et al. 2015). 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).

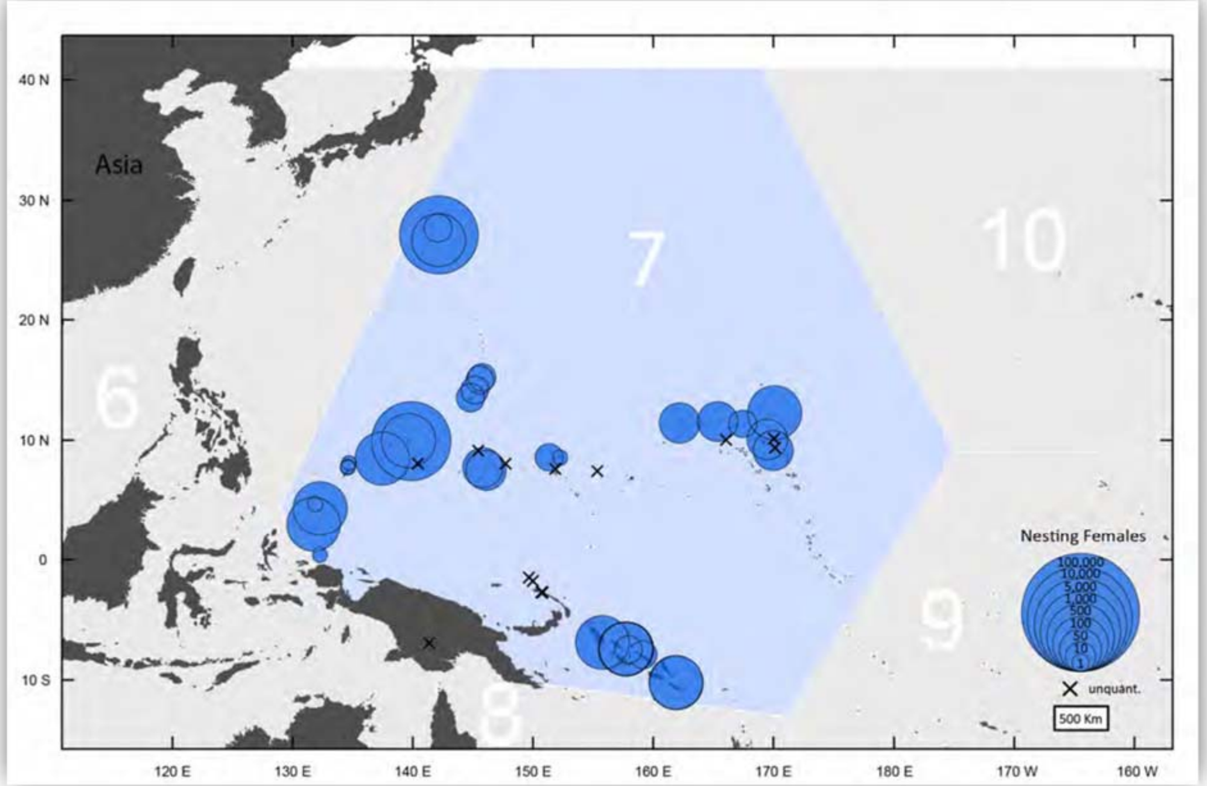

Figure 22**.** Nesting distribution of Central West Pacific green sea turtles (blue-shaded region). Size of circles indicates estimate nester abundance.

Status

Central West Pacific green sea turtles are listed as endangered. The species exhibits low nesting abundance, with an estimated total nester abundance of 6,518 females at 51 documented nesting sites (Seminoff et al. 2015). There are a number of unquantified nesting sites, possibly with small numbers of nesting females; however, specifics regarding these sites is unknown. The largest nesting site is in the FSM, and that particular site hosts approximately 22% of the total annual nesting females.

The limited available information suggests a nesting population decrease in some portions of the species like the Marshall Islands, or unknown trends in other areas such as Palau, Papua New Guinea, the Marianas, Solomon Islands, or the FSM (Maison et al. 2010). There 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. (2008a) reports an estimated annual population growth rate of 6.8% per year for the Chichi-jima 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 sea 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 species (Seminoff et al. 2015).

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

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; 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 as cited by Seminoff et al. 2015). 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 sea 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 as cited in Seminoff et al. 2015; NMFS and FWS 1998a). 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 as cited in Seminoff et al. 2015) 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 sea turtles. The mortality rates recorded for these 33 interactions were high, with only five turtles identified as alive upon release (McCoy 2007a as cited in Seminoff et al. 2015). 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, 2016b, 2017a, 2018d).

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 as cited in Seminoff et al. 2015). 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 2017c).

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 as cited in Seminoff et al. 2015) 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 sea turtles captured in these sets were 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 2008a). 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 FWS 1998a). However, no bycatch studies have been undertaken to quantify the level of incidental capture by commercial fishing operations in the Solomon Islands (Project GloBAL 2009), the CNMI (Project GloBAL 2009b as cited in Seminoff et al. 2015), or Guam (Project GloBAL 2009a as cited in Seminoff et al. 2015). 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 as cited in Seminoff et al. 2015), sea turtles interactions with both fisheries have been commonly observed (Kumoru 2008). However, the level of mortality is unknown.

NMFS estimates that between 0 and 8% of the green turtles caught in the shallow-set longline fishery could be Central West Pacific green sea turtles (P. Dutton pers. comm. November 20, 2017). This means that the HI SSLL fishery may have captured one Central West Pacific green sea turtle since 2004. NMFS estimates that between 0 and 18% of the green turtles caught in the HI DSLL fishery could be from the Central West Pacific species (P. Dutton pers. comm. June 29, 2018). This means that between 0 and 12 Central West Pacific green sea turtles were captured in the HI DSLL fishery since 2004. NMFS estimates that between 0 and 25% of the green sea turtles caught in the American Samoa longline fishery were likely part of the Central West Pacific green sea turtle (P. Dutton pers. comm. July 5, 2018). This means that between 0 and 67 Central West Pacific green sea turtles were captured by the American Samoa longline fishery since 2004.

4.2.3.5 Southwest Pacific Green Sea Turtle

Differential Distribution

The Southwest Pacific green sea turtle 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 *General Adult Foraging and Diving Behaviors of Pacific Green Sea Turtles* above for the description of feeding and diving behavior.

Population Dynamics

Structure

Southwest Pacific green sea turtles are spatially and genetically separated from other green sea turtle species in the Pacific Ocean (Seminoff et al. 2015). These observations, with other data led NMFS and FWS to conclude that the Southwest Pacific green sea turtle is a discrete species (81 FR 91097). Genetic data suggest that the species is comprised of at least four populations: nGBR, sGBR, Coral Sea and New Caledonia (Dethmers et al. 2006, Jensen 2010, and Dutton, NMFS, unpublished data as cited in Seminoff et al. 2015).

The spatial structure of Southwest Pacific green sea turtles is diverse and there appears to be no connectivity among the four primary populations. Seminoff et al. (2015) identified 12 total nesting sites, although perhaps more so than other green sea turtle species, 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, the SRT did the same. 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.

Nesting occurs at moderate to high levels within the Southwest Pacific green sea turtle with some isolated locations having extremely high nesting activity. The largest nesting assemblages are located in the nGBR (Chaloupka et al. 2008a). 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. 2004a; NMFS and FWS 2007a; Chaloupka et al. 2008a; 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).

Demographic information for nesting turtles is widely available for nesting beaches in the Australian component of the species. 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. 2004a). 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

This species has some of the oldest lineages found in *C. mydas*, and is characterized by high genetic and spatial diversity. Nesting is widely dispersed throughout the region, there is more than one major nesting site, and nesting is not completely limited to islands. 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 species, and some nesting sites are 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).

Green turtle nesting is widely dispersed throughout the Southwest Pacific Ocean (Figure 23). The bulk of this species 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). 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). Nesting sites also occur on the Coral Sea Islands, New Caledonia, and Vanuatu.

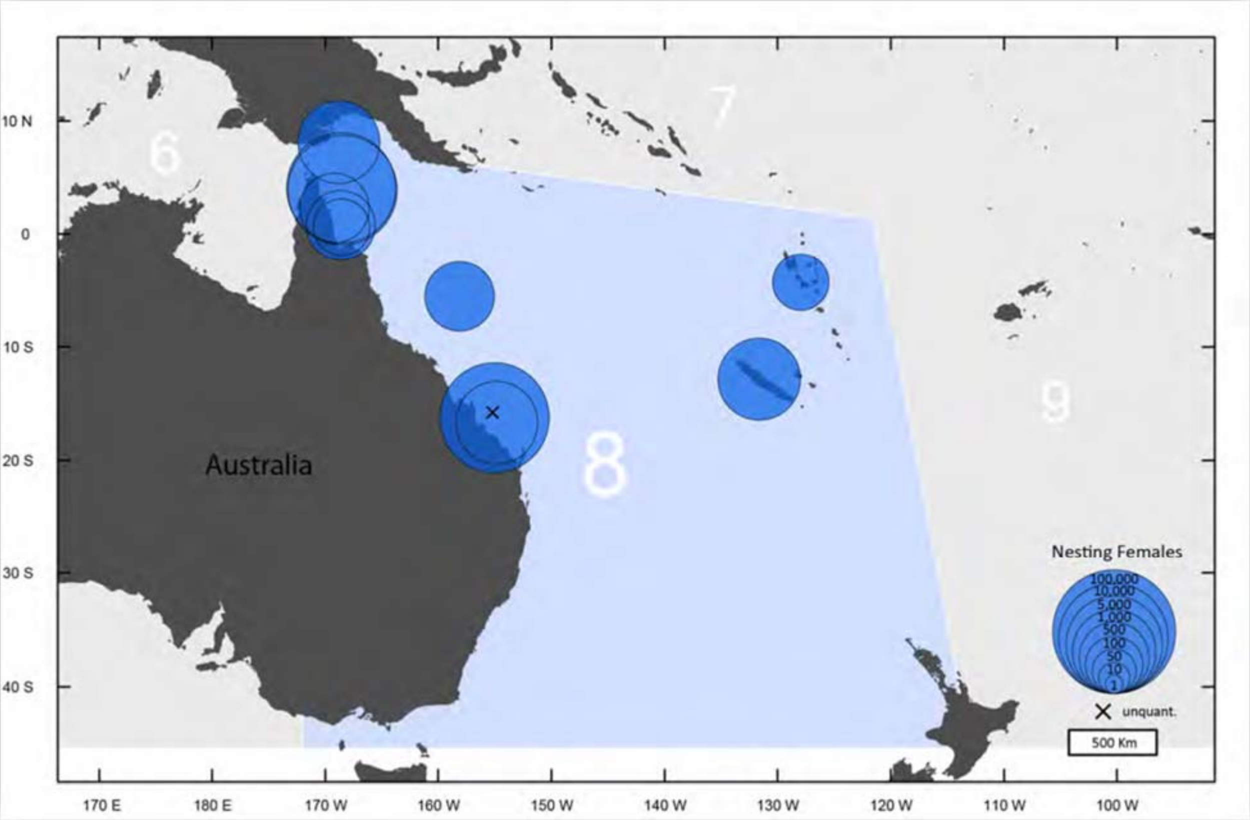

Figure 23. Nesting distribution of Southwest Pacific green sea turtles. Size of circles indicates nesting estimated nester abundance. Locations marked with '·' indicate nesting sites lacking abundance information.

Status

Southwest Pacific green sea turtles are listed as threatened. The species exhibits high nesting abundance, with an estimated total nester abundance of 83,058 adult female (based on the 2015 status review). There are only two nesting areas (Raine Island and Heron Island) 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 total nester abundance is estimated to be 83,058 (Seminoff et al 2015).

Threats to the Species

Natural Threats

The potential effects of diseases and endoparasites, as described for other species of green sea turtles, also exist for green turtles found in the Southwest Pacific green sea turtles. 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 occurs on other sea turtle species (primarily loggerheads) (Environment Australia 2003 as cited in Seminoff et al. 2015). 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 as cited in Seminoff et al. 2015), 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 green sea turtles are vulnerable to harvest throughout Australia and neighboring countries such as New Caledonia, Fiji, Vanuatu, Papua New Guinea, and Indonesia (Limpus 2009). Cumulative annual harvest of green sea turtles that nest in Australia may be in the tens of thousands, and it appears likely that historic Native harvest may have been in the same order of magnitude (Limpus 2009). Annual harvest in the southern East Coast is estimated to be 500–1000, mostly large females (Limpus 2009). Estimated annual harvest is even larger for the northeast part of the country, primarily Torres Strait which may have a large harvest turtles (4,000) and eggs (unquantified). Annual Indigenous harvest from northern and Western Australia may be several thousand turtles per year plus non-permitted egg harvest (Limpus 2009). Harvest from neighboring countries (New Caledonia, Fiji, Vanuatu, Papua New Guinea, Indonesia) may be as high as several thousand juvenile or adults per year and may threaten the sustainability of the Australian green turtle stocks (Limpus 2009).

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 species) 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.

Southwest Pacific green sea turtles 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 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. (2013a) 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. 2013a). 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 and 8% of the green sea turtles caught in the HI SSLL fishery could be Southwest Pacific green sea turtles (P. Dutton pers. comm. November 20, 2017). This means that one turtle from this species may have been captured by this fishery since 2004. NMFS estimates that between 0 and 25% of the green sea turtles caught in the HI DSLL fishery could be Southwest Pacific green sea turtles (P. Dutton pers. comm. June 29, 2018). This means that 0-17 turtles captured in the HI DSLL fishery since 2004 were Southwest Pacific green sea turtles. NMFS estimates that between 11.6 and 46.5% of the green turtles caught in the American Samoa longline fishery could be Southwest Pacific green sea turtles (P. Dutton pers. comm. July 5, 2018). This means that between 31 and 126 green sea turtles captured in the American Samoa longline fishery since 2006 were likely Southwest Pacific green sea turtles.

Southwest Pacific green sea turtles are also 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 Green Sea Turtle

Differential Distribution

The western boundary for the East Indian/West Pacific green sea turtle is 84°E longitude from 40°S to where it coincides with India near Odisha, northeast and into the West Pacific Ocean to include Taiwan extending east at 41°N to 146°E longitude, south west to 4.5°N, 129°E, then south and east to West Papua in Indonesia (at 135°E) and the Torres Straits in Australia (at 142°E longitude). The southern boundary is 40°S latitude, encompassing the Gulf of Carpentaria. Green turtle nesting is widely dispersed throughout the range of the species, 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 *General Adult Foraging and Diving Behaviors of Pacific Green Sea Turtles* above for description of feeding and diving behavior.

Population Dynamics

Structure

East Indian/West Pacific green sea turtles are spatially and genetically separated from other green sea turtle species in the Pacific Ocean (Seminoff et al. 2015). These observations, with other data led NMFS and FWS to conclude that the East Indian/West Pacific green sea turtle is a discrete species (81 FR 91097). Genetic data suggest that the species is comprised of at least 16 populations: 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; 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 species at some foraging areas. The demography of East Indian/West Pacific green turtles 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 species. The size of nesters throughout the species 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 as cited in Seminoff et al. 2015; 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 as cited in Seminoff et al. 2015; Cheng et al. 2009; Jensen 2010; Waayers 2010; Chen et al. 2010; Dethmers 2010; Muhara and Herlina 2012 as cited in Seminoff et al. 2015; 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 East Indian/West Pacific green sea turtle. 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 species, providing a level of population resilience through habitat diversity. The nesting season varies throughout the species, 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 as cited in Seminoff et al. 2015), 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 species, 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 24) (Limpus 2009). Currently, the East Indian/West Pacific green sea turtle 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.

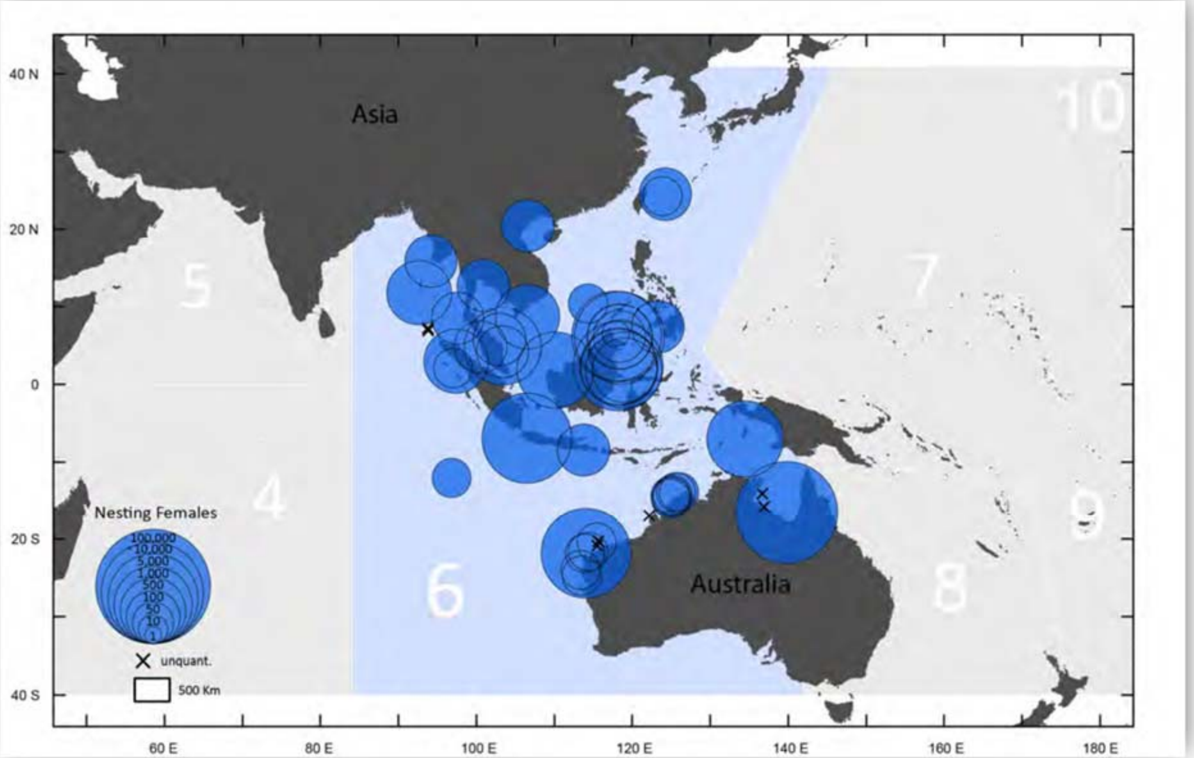

Figure 24**.** Nesting distribution of East Indian/West Pacific green sea turtle (blue-shaded area). Size of circles indicates estimated nester abundance. Locations marked with '·' indicate nesting sites lacking abundance information.

The in-water range of the East Indian/West Pacific green sea turtle is similarly widespread. 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 as cited in Seminoff et al. 2015). 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, 2006b).

Status

East Indian/West Pacific green sea turtles are listed as threatened. The species exhibits high abundances with an estimated total nester abundance of 77,009 females at 50 nesting sites. The largest nesting site is the in northern Australia and supports about 25,000 nesting females.

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 2010c).

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 as cited in Seminoff et al. 2015), Western Australia (Raidal and Prince 1996; Aguirre and Lutz 2004), and on PhuQuoc in Vietnam (Ministry of Fisheries 2003). Epidemiological studies indicate rising incidence of this disease (George 1997), thus the above list will likely grow in the future.

The best available data suggest that current nest and hatchling predation on the East Indian-West Pacific green sea turtle is prevalent. Depredation of nests by feral animals is also widespread in many South Asian areas (Sunderraj et al. 2001 as cited in Seminoff et al. 2015; 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 1996). 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 the range of this species (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 percentage 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 as cited in Seminoff et al. 2015).

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 as cited in Seminoff et al. 2015). It was believed this decline could be attributed to egg harvesting, although turtles were also threatened from incidental capture in fishing gear (Mortimer 1991a as cited in Seminoff et al. 2015). 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 FWS 2007a). 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 as cited in Seminoff et al. 2015).

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 as cited in Seminoff et al. 2015). However, this apparent decline could be reflective of a decline in egg collection efforts rather than a decline in egg production. Egg harvesting in Indonesia occurred for decades 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 (2010c) 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 as cited in Seminoff et al. 2015).

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; 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 as cited in Seminoff et al. 2015; 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 as cited in Seminoff et al. 2015; 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 and Yunupingu 2004).

NMFS estimates that between 0-10% of the green sea turtles caught in the HI SSLL fishery may be East Indian/West Pacific green sea turtles (P. Dutton pers. comm. November 20, 2017). This means that one East Indian/West Pacific green sea turtle may have been captured by the HI SSLL fishery since 2004. NMFS estimates that between 0 and 28% of the green sea turtles caught in the HI DSLL fishery may be East Indian/West Pacific green sea turtles (P. Dutton pers. comm. June 29, 2018). This means that 0 and 19 green sea turtles captured by the HI DSLL fishery since 2004 were likely East Indian/West Pacific green sea turtles. NMFS estimates that between 0 and 11.4% of the turtles caught in the American Samoa longline fishery could be East Indian-West Pacific green sea turtles (P. Dutton pers. comm. July 5, 2018). This means that between 0 and 31 turtles were captured in the American Samoa longline fishery since 2006 were likely East Indian/West Pacific green sea turtles.

4.2.3.7 Summary of the Status of Green Sea Turtles

In this section of this biological opinion, we explained that Central North Pacific, Eastern Pacific, Southwest Pacific, and East Indian-West Pacific green sea turtles are threatened; and Central West Pacific and Central South Pacific green sea turtles are endangered. We used our knowledge of the species' demography and population ecology to capture the primary factors that appear to determine the green sea turtle's population dynamics (Figure 25).

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

In our analysis of each green sea turtle species, we concluded most of these species face the same general threats. For instance, hatchling predation is encountered by all green sea turtles; however, the predatory species may change depending on geographic location. Those dynamics are consistent with the status and trend of the count data: 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 25). Anthropogenic threats can be direct, indirect, and may contain negative or positive associations between variables in relation to the stressor-response relationships illustrated in the casual loop. These threats are represented in our diagram by a path (arrow) that illustrates a negative relationship with the life stages they affect as represented by the negative sign at the terminus of the arrow where these threats meet the particular life stage of concern. A variety of conservation efforts have, and continue to occur, and these are represented with another arrow to the threats, and demonstrate a decrease in the threats as evidenced by the positive sign at the terminus of the arrow. Each threat to the species reduces to population to some degree, however the overall trend in the species' trajectory is positive.

While these green sea turtle species face both natural and anthropogenic threats at all life cycle stages, the number of recruits into the population appears to balance the loss from the various stressors as graphically represented by the diagram. Our causal loop diagram (Figure 25), 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 turtles.

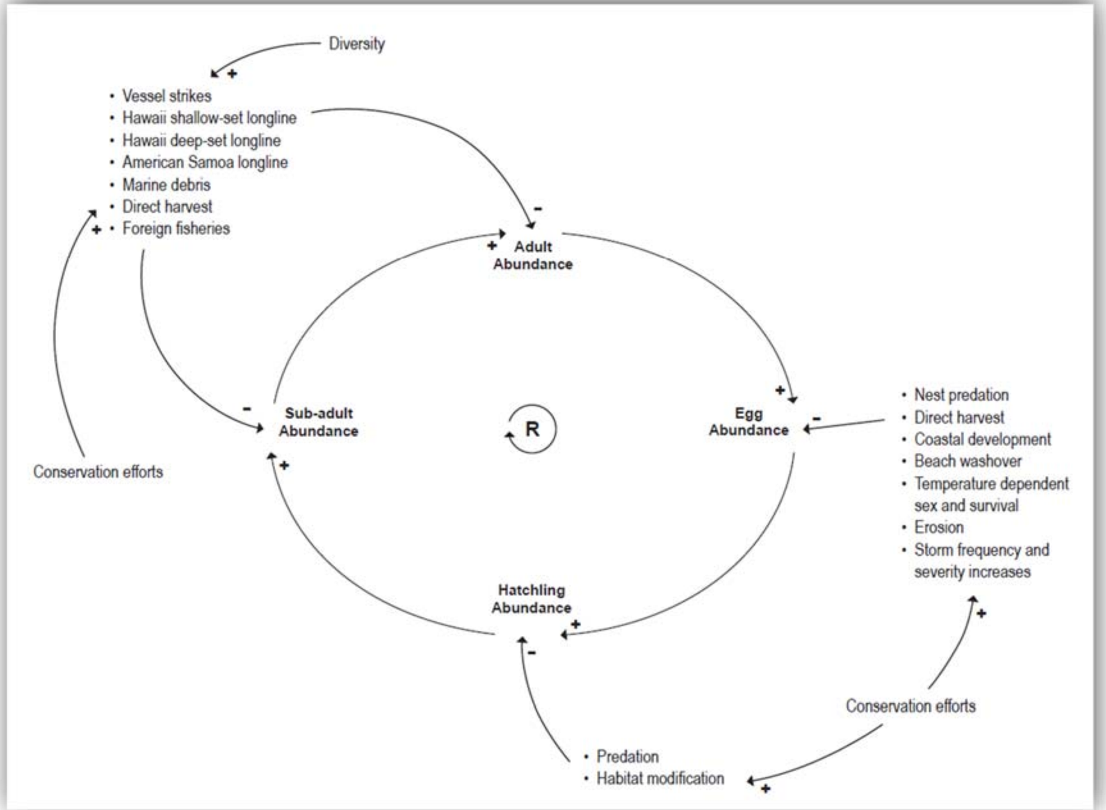

Figure 25. Green sea turtle casual loop diagram depicting various stressor-response relationships for each life cycle stage applicable to each species under consideration.

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 one or more of these turtles as a separate species (or DPS); however a substantial amount of information has become available on the population but the Services have not fully assembled or analyzed the new information (NMFS and FWS 2014).The data appear 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). 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 suggest 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. 2007a; 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; McMahon et al. 2007; NMFS and FWS 2014).

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 (Rodriguez-Zarate et al. 2013). Genetic diversity was low among these nesting sites indicating a population collapse likely due to localized over exploitation (Rodriguez-Zarate 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. (2010a) identified two distinct regional management units in the east Pacific (arribada and solitary nesters) based on a meta-analysis of genetic and other relevant data on olive ridley life history and biogeography. They believed the arribada and solitary nesting assemblages warranted separate management considerations, given there were differences in genetic diversity, trends, and abundance between the two types of nesting behaviors.

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 FWS 2007d, 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 were 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. (2007b) 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 26 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).

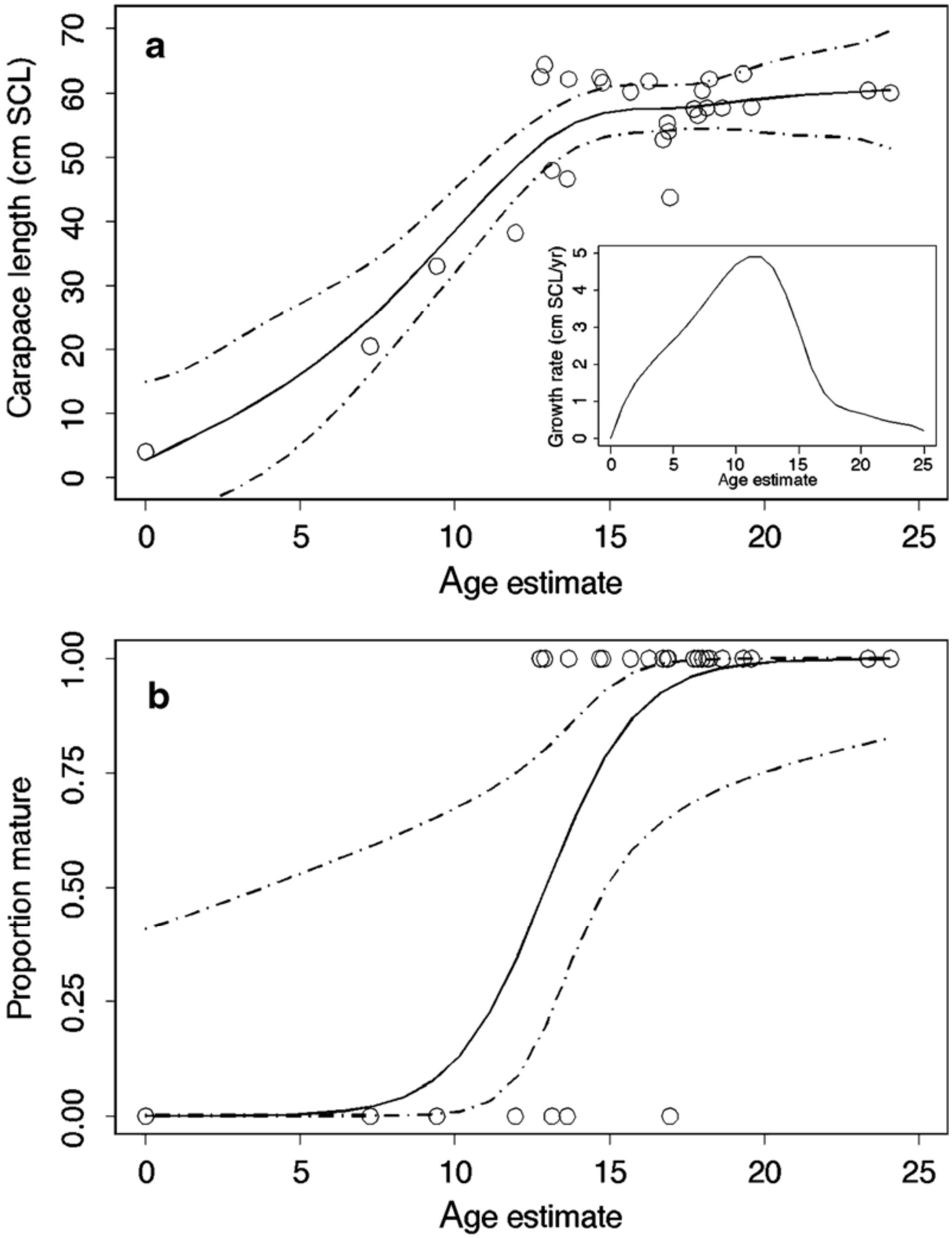

Figure 26. Panel (a) shows the expected size-at-age growth function (solid curve) fitted using a generalized smoothing spline model (Gu 2002) with 95% credible interval shown by the dashed curves. Age estimates shown by open dots and an estimate for expected olive rid- ley hatchling size is also included. 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 (NMFS and FWS 1998d; Avens and Snover 2013). 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. 2007a). 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 were needed to establish these rates. Additionally, Whiting et al. (2007a) 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 Hernandez-Echeagaray et al. 2012), approximately 30-31°C for nesting populations in Costa Rica, and less than 29°C in Gahirmatha, India (reviewed by Wibbels 2007). Pivotal temperatures likely vary within and among populations and generalizations should be applied with caution. Studies on sex ratios of olive ridley hatchlings are few and non-existent for juvenile and adults. Hernandez -Echeagaray et al. (2012) found a slight female-bias sex ratio (55%) for the 2010- 2011 nesting season at La Escobilla, Mexico. Sex ratios may also change over the nesting season. In Mexico, a female-biased hatchling sex ratio was found at most nest sites (La Escobilla was not included) at the beginning of the nesting season, and a male-biased ratio at the end of the season (Sandoval-Espinoza 2011 as cited in 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 (Gaos et al. 2006; Dornfeld and Paladino 2012). Conversely, survivorship is low on high density arribada nesting beaches because of density-dependent mortality (Cornelius et al. 1991). 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 (Bezy 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 18) (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%). Martin 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 18. 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

Location	**Annual Number**	**Trend**
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 19) (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 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 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 19. 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 20) (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 and FWS 2014).

Table 20. 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 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 2018 there were 686 olive ridley interactions in the HI DSLL fishery and from this the estimated mortality is 650 (NMFS 2019b). 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 no mortality (NMFS unpublished data). 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 2013). Since 2001 no olive ridley sea turtles have been captured in the California Oregon drift gillnet fishery and only one has been observed since 1990 (NMFS unpublished data).

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 27).

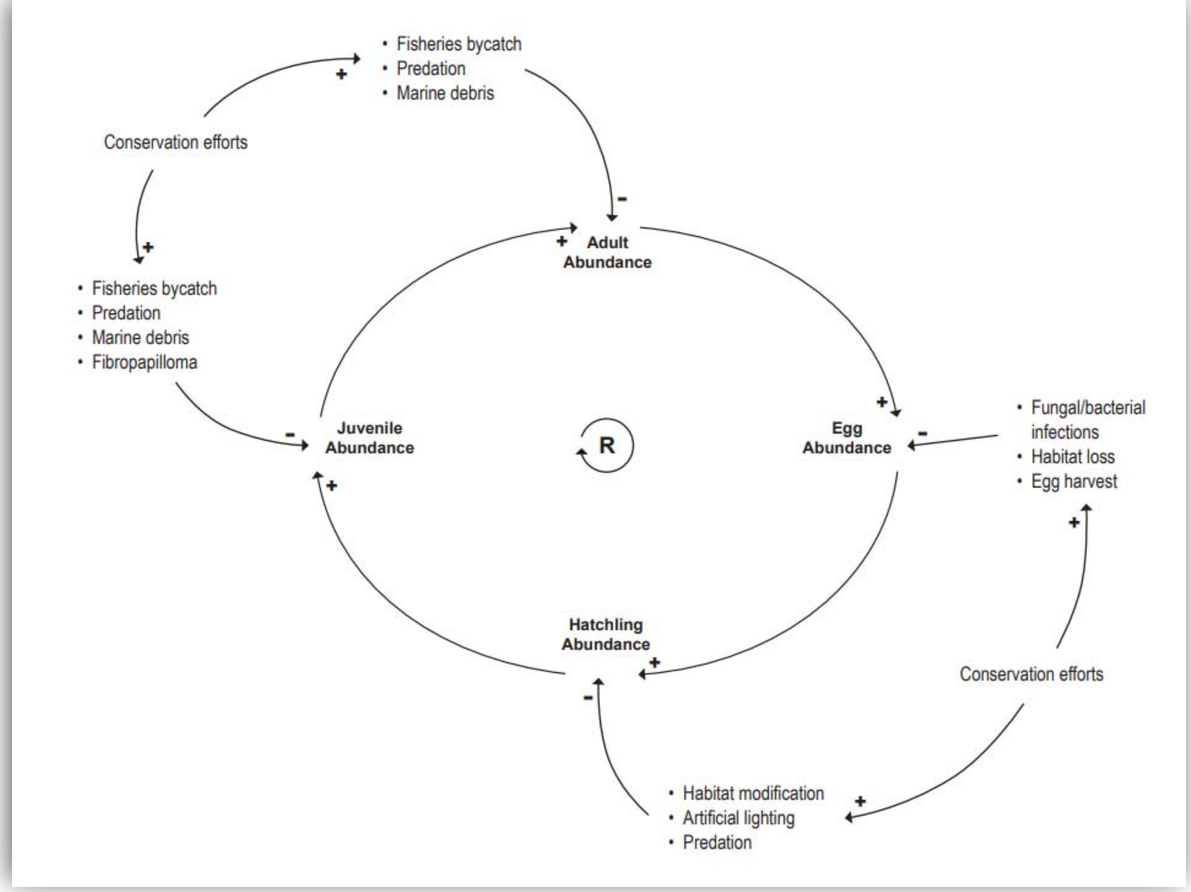

Figure 27. Olive ridley sea turtle causal loop diagram depicting various stressor-response relationships for each life cycle stage.

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. 2017). 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 28) (2009).

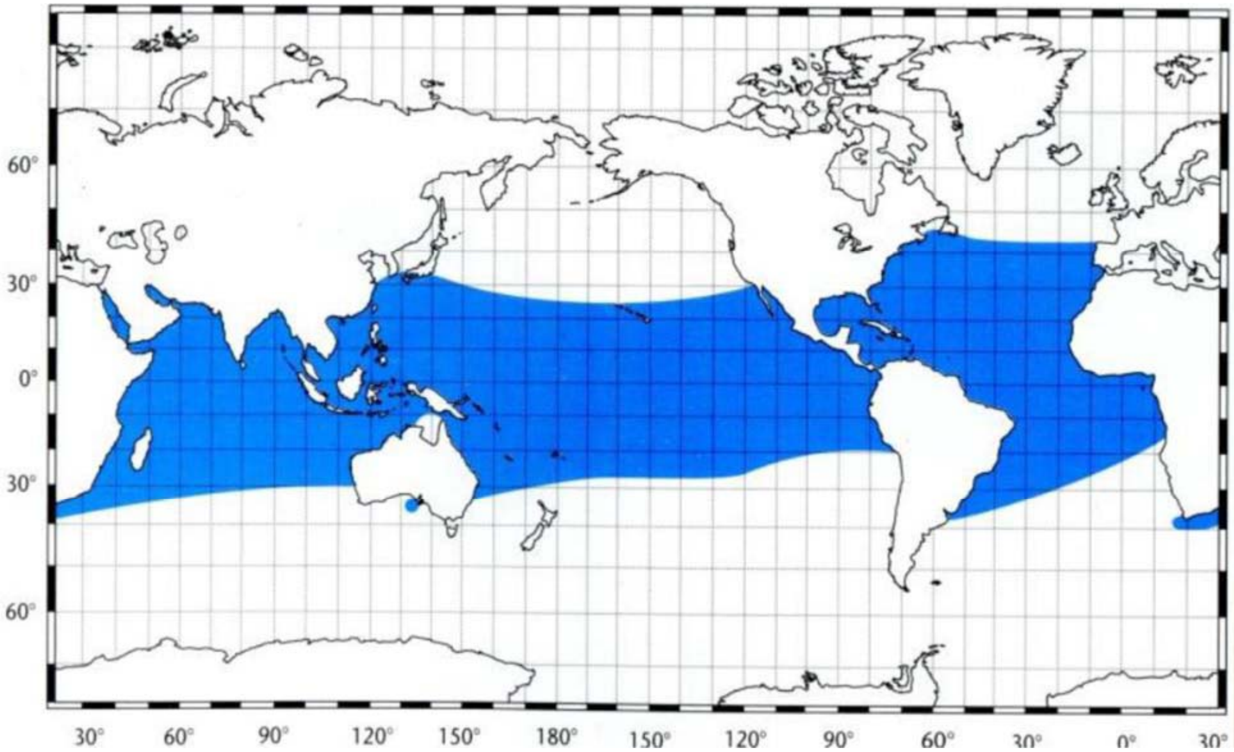

Figure 28. Geographical distribution of the oceanic whitetip shark (Last and Stevens 2009; Young et al. 2017).

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; Nakano et al. 1997; Bonfil et al. 2008; Clarke et al. 2011a; Hall and Roman 2013; Tolotti et al. 2013; Young et al. 2017).

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. 2016; Young et al. 2017). 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. 2017; 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. 2017). 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. 2017).

While little is known about oceanic whitetip migration patterns, several studies have provided valuable insights which are discussed in detail by Young et al. (2017). 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. 2017). 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 29 below (Kohler et al. 1998; Bonfil et al. 2008; Young et al. 2017).

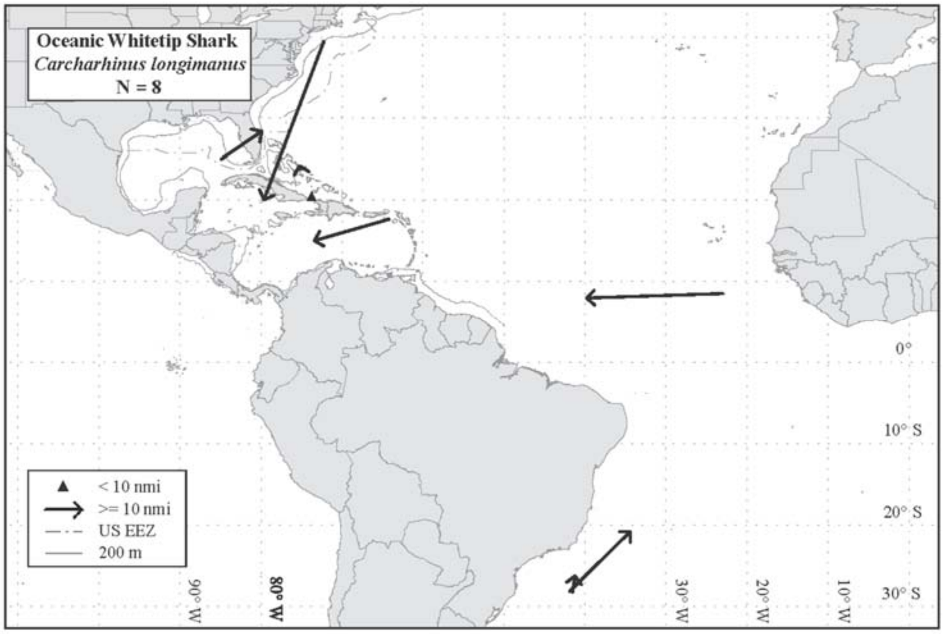

Figure 29. Movement distributions of oceanic whitetip sharks from mark recapture studies from the NMFS Co-operative Shark Tagging Program during 1962-93, NMFS unpublished data, and Young et al. 2017.

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. 2017). 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. 2017). 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. 2017).

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. 2017). The distribution illustrated by Garcia-Cortes et al. (2012) are highly influenced by the effort of the fleet as the data are related to total catches rather than catch per unit effort (CPUE) (Young et al. 2017).

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. 2017). 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. 2017).

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. 2017). 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. 2017). 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. 2017).

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. 2017; 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. 2017).

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. 2017). Camargo et al. (2016) compared the mitochondrial control region in 215 individuals from the Atlantic and Indian Oceans which identified 12 haplotypes (Figure 30) (Young et al. 2017). 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 (Camargo et al. 2016; Young et al. 2017). 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. 2017). 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. 2017). Lastly, it should be noted that this study only used mitochondrial markers, meaning male-mediated gene flow is not reflected (Young et al. 2017) although other species in the Carcharhinus genus are known to exhibit male-mediated gene flow between populations (Portnoy et al. 2010).

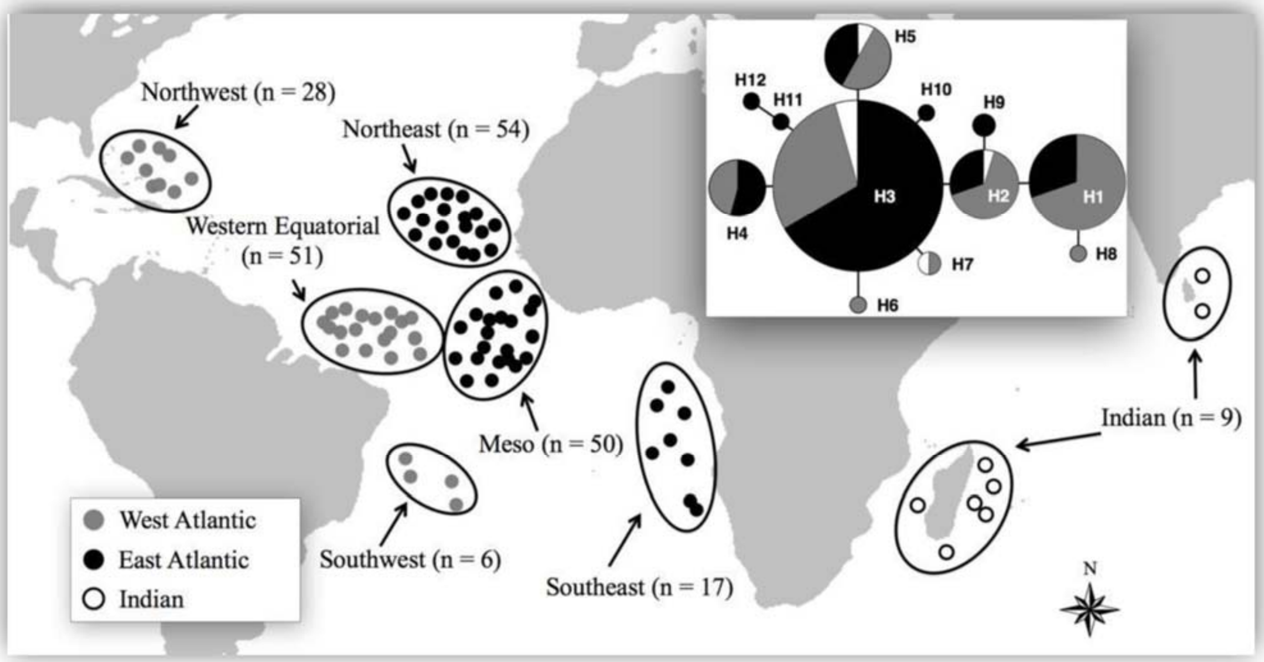

Figure 30. Geographic distribution of C. longimanus samples relaying analyzed network haplotypes and compilation of the mitochondrial DNA control regions. Source: Camargo et al. 2016; Young et al. 2017.

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. 2017). 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. 2017).

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. 2017). 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 ($P \leq 0.02$) (Young et al. 2017). 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. 2017). 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. 2017). Significant inter-basin population structure was apparent (Figure 31); 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. 2017).

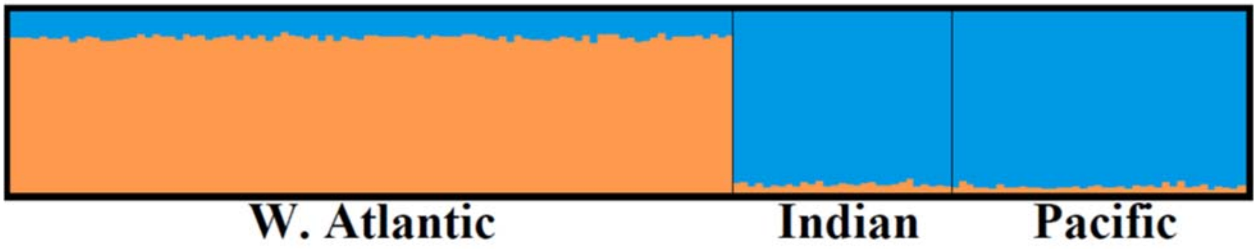

Figure 31. DISTRUCT plots summarizing STRUCTURE results of all genotyped samples: K = 2. Which clearly indicates strong sorting of two clusters between the Western Atlantic and the Indo-Pacific. Sources: Ruck 2016; Young et al. 2017.

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. 2017).

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. 2017). 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. 2017). 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. 2017). 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. 2017). Camargo 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; Camargo et al. 2016; Ruck 2016; Young et al. 2017).

Overall, the data do not show male-mediated gene flow within the Atlantic but relies solely on mtDNA to determine population structure (Young et al. 2017). Young et al. (2017) 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. 2017).

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. 2017). 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 32 below shows the various growth curves for the oceanic whitetip shark.

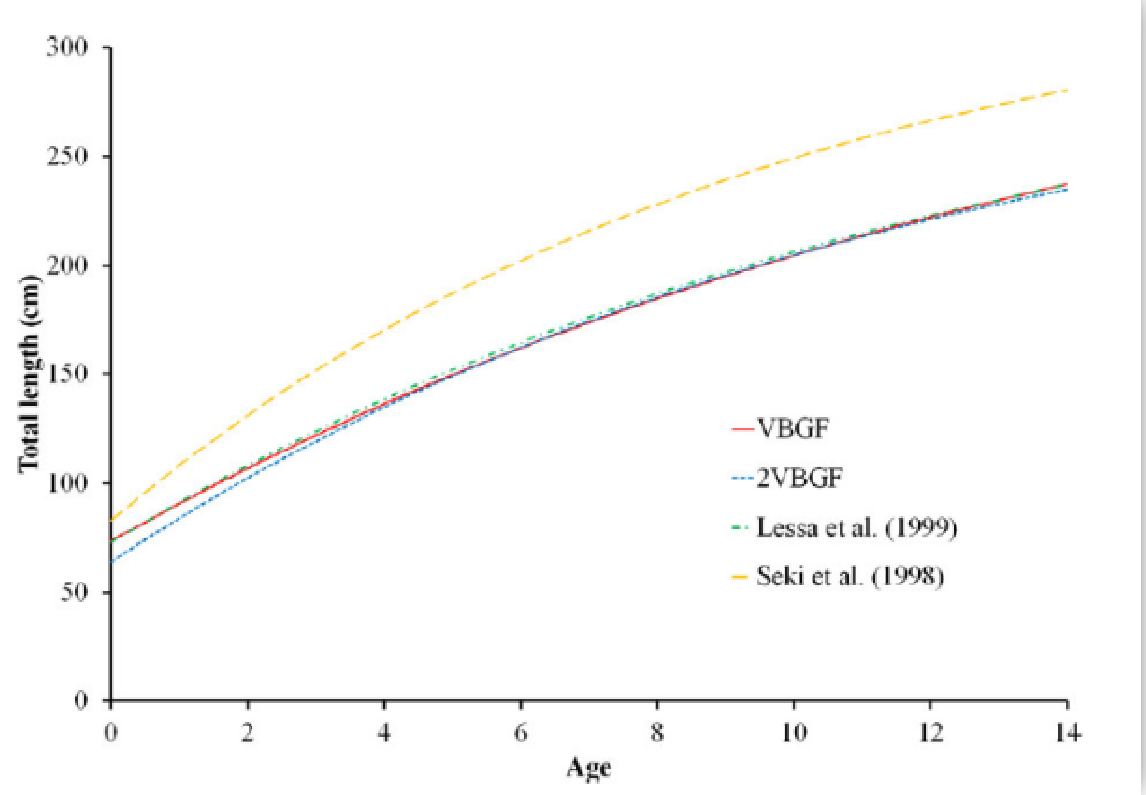

Figure 32. Comparison of the growth curves of the oceanic whitetip shark in different regions, from Seki et al. (1998), Lessa et al. (1999) and the current study. VBGF = von Bertalanffy growth function. 2VBGF was only used in the present study and VBGF was used in the other studies. Source: Joung et al. 2016; Young et al. 2017.

Since the status review was completed in 2016, an additional study compared von Bertalanffy growth curve values found in Figure 33 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 33) for total weight (TW): TW= (.386e-4)*FL^((2.75586)) (n = 587; both sexes).

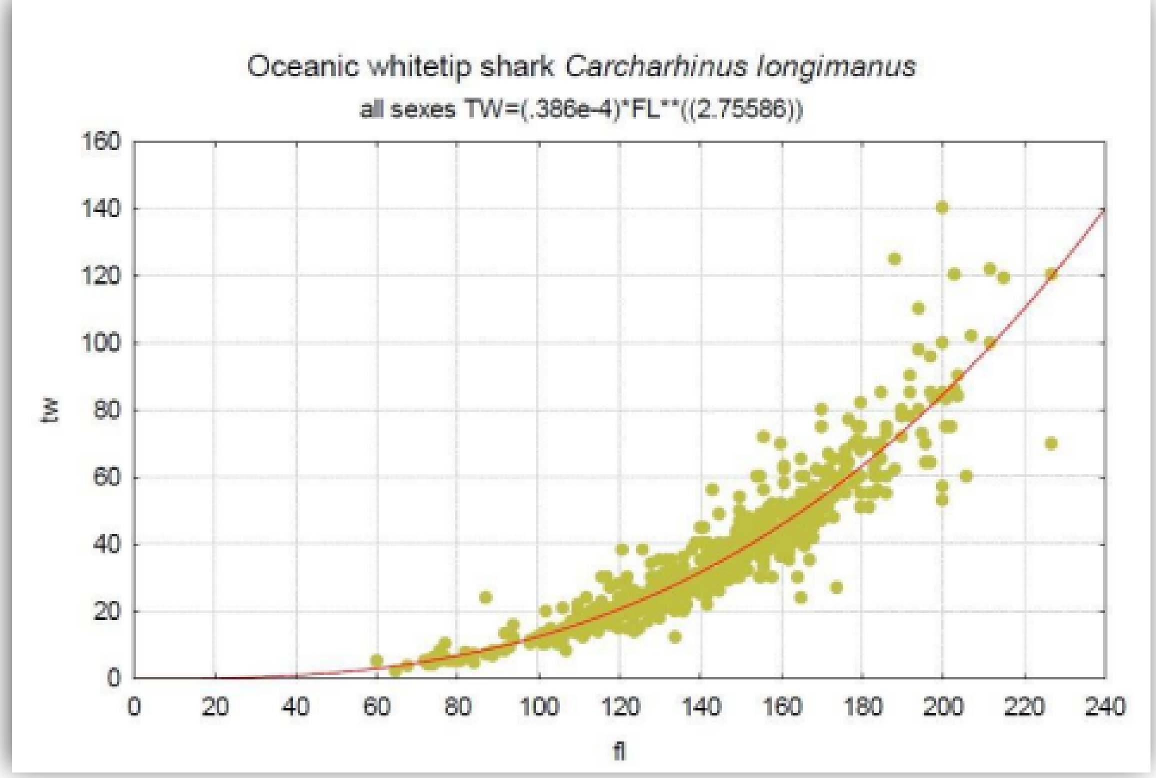

Figure 33. Length-weight scatterplot and relationship for oceanic whitetip shark (all sexes, n=587). Source: Romanov and Romanova 2009; Young et al. 2017.

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 suggest 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 21 below provides a summary of life history characteristics reported in published literature to date.

Table 21. 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. 2017.

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, 2008)
	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 (2015)
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[4] 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 (2015)
Generation Time	7 years	Cortés (2002)
	11.1 years	Smith et al. (2008)

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[4] 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 (2008)
	r= 0.094 (0.06-0.137)	Cortés et al. (2010)[5]
	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. 2017). 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. 2017). 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. 2017). 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. 2017). Young et al. (2017) recognizes that regulatory mechanisms alone may not equate effectiveness in achieving their intended purpose. Complex issues limit the effectiveness of well-intended statutes and legislation within the range of this species (Young et al. 2017).

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

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. 2017).

4.2.5.4 Threats to the Species

Threats to the oceanic whitetip shark are summarized in the status review (Young et al. 2017). However, the most significant threats identified for the species are fisheries bycatch and exploitation for the fin trade. As Young et al. (2017) 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. 2017). 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. 2017).

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. 2017). 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. 2017).

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. 2017). 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. 2017).

Anthropogenic Threats

Global trends in the abundance of the oceanic whitetip shark is lacking based on poor quantitative data (Smale 2008; Young et al. 2017). 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. 2017). 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 are 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. 2017). Several indices relating to abundance are available to infer population trends in certain regions.

Young et al. (2017) 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 are 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. 2017). 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 standards, establish probable declines in whitetip populations between 25–90% (Young et al. 2017). 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. 2017). 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. 2017). 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 34, Figure 35, and Figure 36) (Clarke et al. 2011a, 2011b, 2012; Lawson 2011; Rice and Harley 2012; Rice et al. 2015; Young et al. 2017). The data show 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. While the data cover 1992 to 2009, industrial longline and purse seine fishing in the western and central Pacific Ocean commenced in the 1950s and 1980s, respectively.

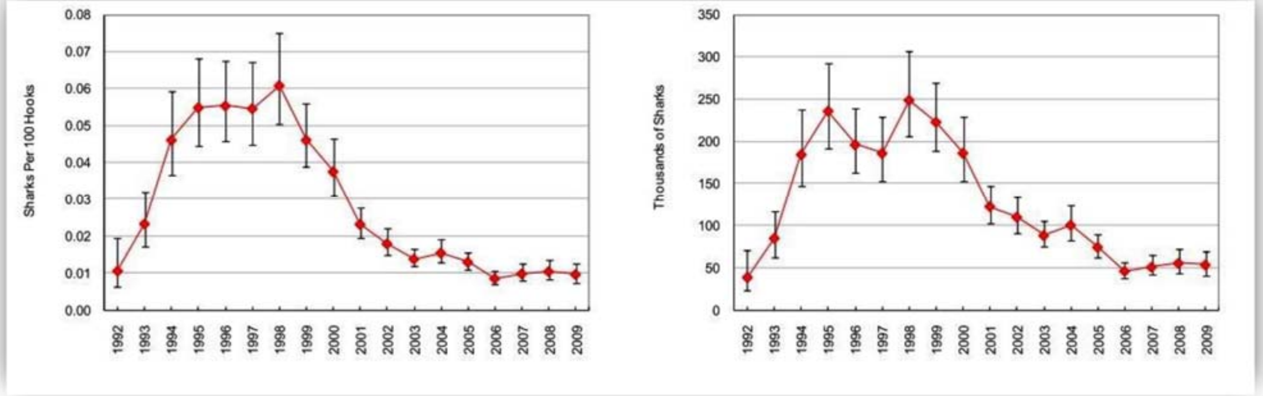


Figure 34. Estimates of longline catch rates (left) and catches (right) of oceanic whitetip sharks in the WCPFC Statistical Area east of 130°E. Source: Lawson 2011; Young et al. 2017.

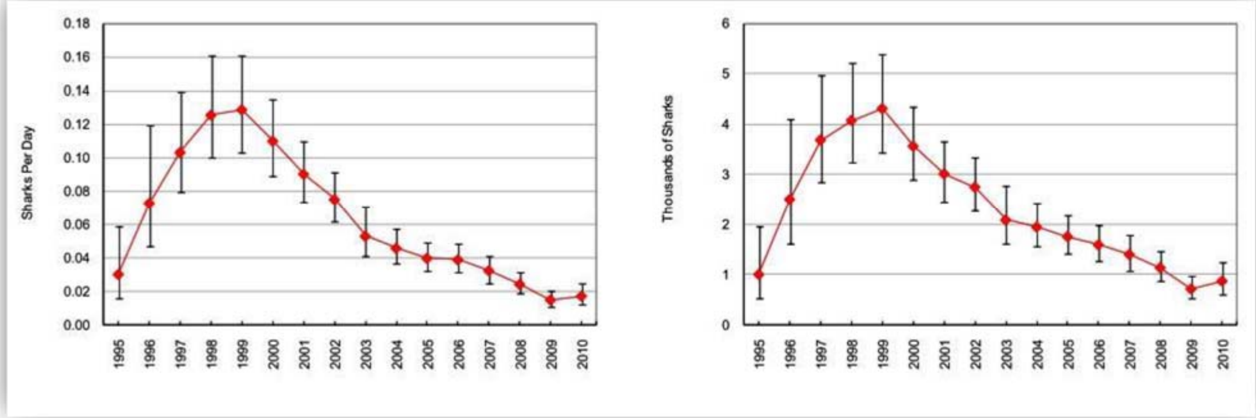


Figure 35. Estimates of purse seine catch rates (left) and catches (right) of oceanic whitetip sharks in the area from 20°S to 20°N and 130°E to 210°W. Source: Lawson 2011; Young et al. 2017.

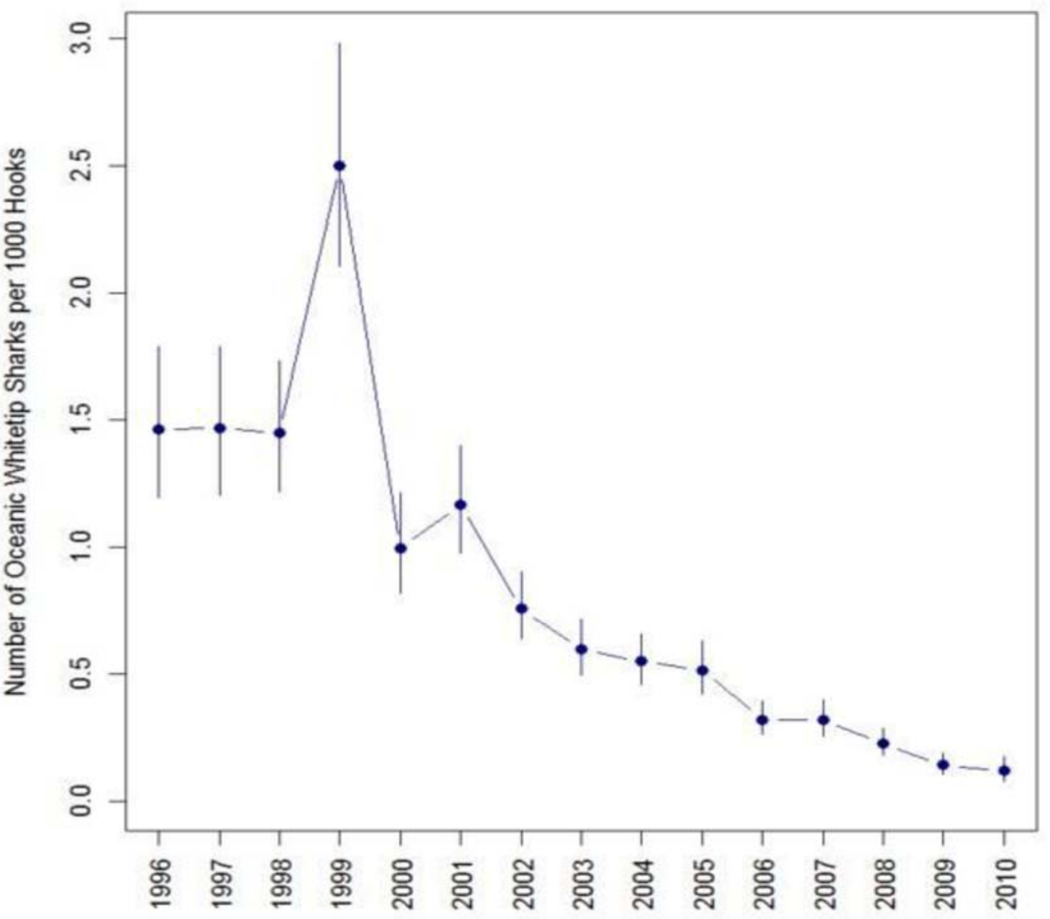


Figure 36. Standardized catch rates using a quasi-Poisson formulation of a generalized linear model for oceanic whitetip sharks in the Western and Central Pacific Source: Clarke et al. 2011a; Young et al. 2017.

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. 2017). 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. 2011a; Rice and Harley 2012; Young et al. 2017). 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. 2011a; Rice et al. 2015; Young et al. 2017). 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. (2018b) for multiple regions of the WCPFC using longline observer data. Table 22 shows statistics relevant to the oceanic whitetip shark north of 10N, between 10S and 10N, 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. (2018b). It should be noted that these data already include 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. 2018b).

Table 22. Median oceanic whitetip shark catch estimates ('000 individuals) north of 10N, between 10S and 10N, and South of 10S. Source: Peatman et al. 2018b.

	Oceanic Whitetip Shark (reported in '000 individuals)		
Year	**North of 10N**	**10S to 10N**	**South of 10S**
2003	51.2	96	32.6
2004	51.0	87.9	25.8
2005	50.7	57.3	2.03
2006	41.5	45.3	18.3
2007	46.8	48.5	15.5
2008	46.2	39.4	14.4
2009	47.6	45.9	18.1
2010	38.1	41.5	25.2
2011	46.9	42	21.5
2012	29.7	44.6	17.2
2013	16.1	25.6	12.4
2014	18.2	21	10.1
2015	23.6	26.9	10.7
2016	25.0	21.5	11.1
2017	15.9	14.7	10.3
Totals	548.5	658.1	263.5

Significant population declines have been summarized by Young et al. (2017) 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. 2017). 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. 2017). 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. 2017).

More recent data conflict with the expectation of the Hall and Roman (2017) 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 37 and Table 23) which are most likely a result of decreased number of FADs and are indicative of population declines in the area (Peatman et al. 2017).

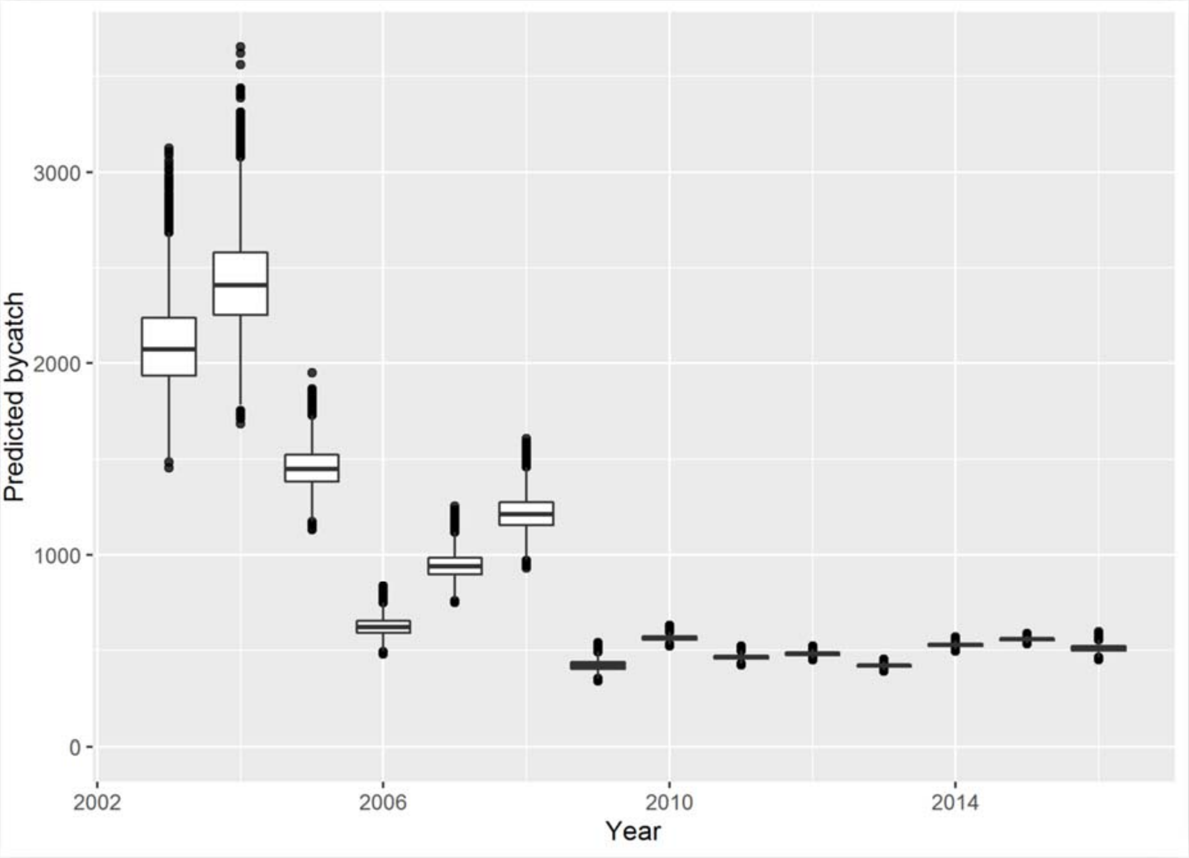


Figure 37. Predicted total annual oceanic whitetip bycatch (numbers) by year for large-scale purse seine fleets. Source Peatman et al. 2017.

Table 23. 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 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 (Peatman et al. 2017). An update was provided by Peatman et al. (2018a) for the 2017 fishing season and reported 721 sharks, resulting in total median bycatch estimate of 13,882 from 2003 to 2017. As Peatman et al. (2018a) discusses, observer coverage was only 20% and the bycatch estimates should be considered preliminary. However, at this time, this is considered the best scientific data available for this fishery and Region.

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.

From 2008-2015 the U.S. purse seine fleet operating in the WCPO caught approximately 1539 oceanic whitetip sharks (WCPFC Regional Observer Program unpublished data; NMFS 2019 in prep).

Finally, the observed and estimated total number of oceanic whitetip shark interactions in the U.S. American Samoa longline fishery from 2010-2017 are shown in Table 24 (NMFS unpub.

data). Observer coverage data for 2018 was not included. The total number of observed interactions between 2010 and 2017 was 918 and the estimated number of interactions is 5,020 sharks.

Table 24. 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 2010-2017. Source: McCracken 2019 in review

Year	Observed Interactions	Estimated Interactions
2010	124	1176
2011	107	319
2012	68	470
2013	87	407
2014	104	464
2015	168	827
2016	197	899
2017	63	458

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. 2017). Full details of the literature debate are summarized by Young et al. (2017). 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. 2017). 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 2012a, 2014a; Young et al. 2017). 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. 2017).

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. 2017). However extremely low CPUE in the region and high 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. 2017). 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. 2017). 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. 2017). 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. 2017). 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. 2017). 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. 2017).

Young et al. (2017) 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. 2017). 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. 2017). 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. 2017).

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. 2017; 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. 2017). Young et al. (2017) 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 38).

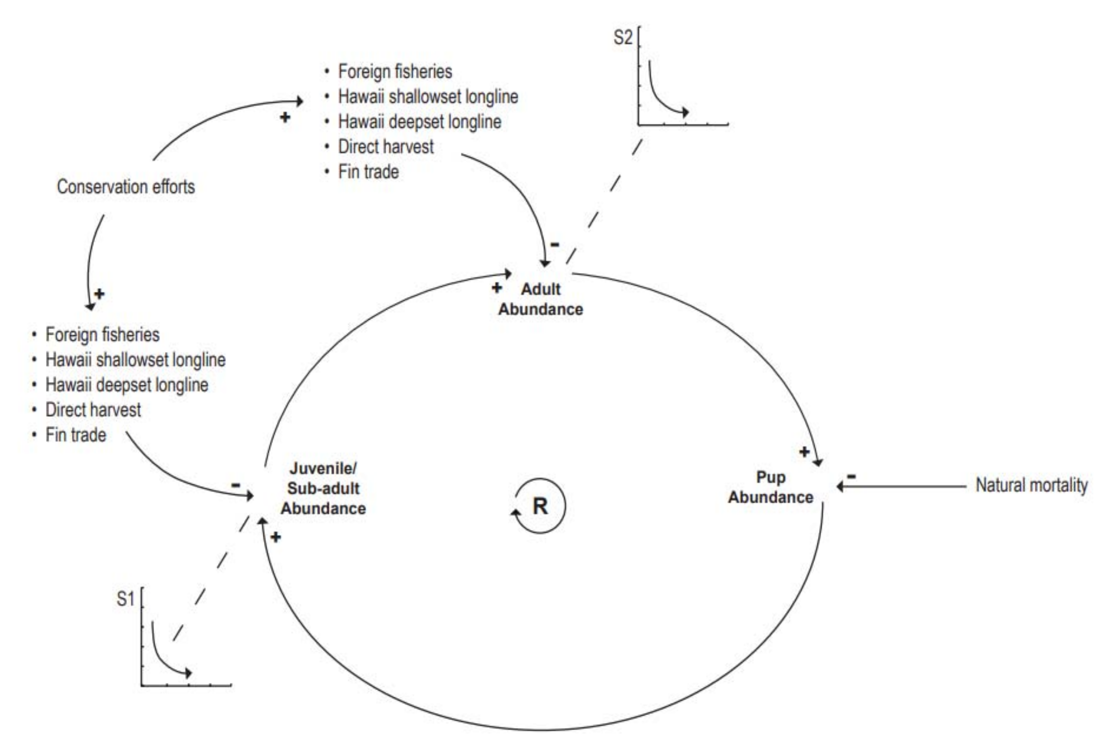


Figure 38. Oceanic whitetip shark causal loop diagram depicting various stressor-response relationships for each life cycle stage.

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. 2017). Overutilization, including fisheries bycatch, and inadequate regulatory mechanisms are the most concerning threats that may contribute to the extinction risk of the species.

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. (2017), geographic distribution maps were created and recently refined based on current distribution knowledge of the giant manta ray (Lawson et al. 2017). These maps (Figure 39) 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.

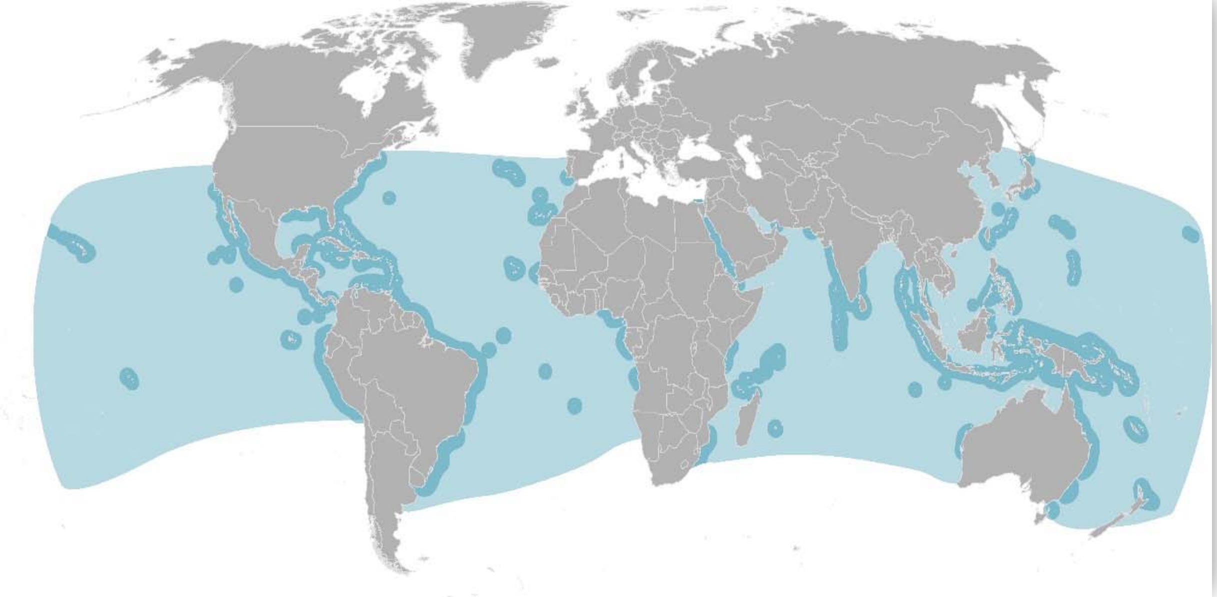


Figure 39. Distribution map for the giant manta ray. Extent of occurrence is depicted by light blue and the area of occupancy is noted in darker blue. Source: Lawson et al. 2017.

Despite this large range, sightings are often sporadic with the exceptions of (Table 25) 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 25. 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

Country	**Specific location**
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 as cited in Miller and Klimovich 2016; 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 as cited in Miller and Klimovich 2016; 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 Yucatan 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 Yucatan 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 Yucatan 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; Couturier 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 as cited in Millerand Klimovich 2016; 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. (1999) 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 (Kitchen-Wheeler 2013) as mantas do not provide any parental care to their offspring after birth. Some data suggest 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. (2011) and Marshall and Bennett (2010) 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 (CITES 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 2010; Kitchen-Wheeler 2013; Marshall et al. 2018) 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. 2011, 2018). 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 26). 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 26. 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 (2011)
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)

Location	Recorded Individuals	Subpopulation Estimate	Reference
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 40). Years of higher abundance of the species were correlated with lower El Nino 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 these data are available. Giant manta rays are reportedly targeted in fisheries in Indonesia, Philippines, India, Thailand, Mozambique, Tonga, Micronesia, Peru, Ghana, and previously in Mexico and possibly the Republic of Maldives.

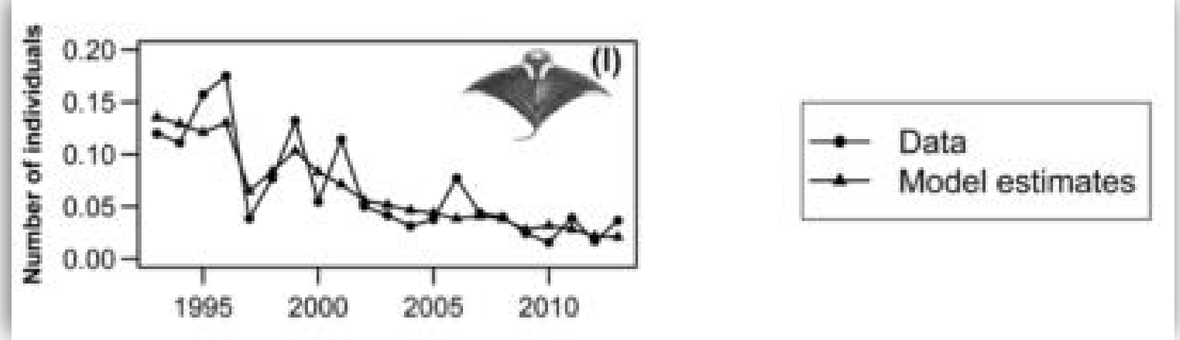


Figure 40. Observed and modeled estimates of mean annual numbers of giant manta rays in Cocos Island National Park, based on diver observations from 1993 to 2015 (Source: White et al. 2015).

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

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

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

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. as cited in Millerand Klimovich 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 2013).

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 as cited in Miller and Klimovich 2016).

Opportunistic hunting of manta rays has been reported in Tonga and Micronesia (B. Newton and J. Hartup pers. comm. cited in CMS 2014). While the extent of this fishing and associated impacts on the local manta ray populations are unknown, given the reportedly opportunistic nature of the 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 are 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. as cited in Miller and Klimovich 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 (1989) 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 FAD purse-seine sets, however, it does not appear to be caught in large numbers (Amande 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 (Amande 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 (data from 214 observed vessels, 833 trips, and 3,032 hauls (shark bottom longline observer program report).

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 2006), 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 (southeast gillnet observer program reports).

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 (Amande 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 IOTC 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 1981)), 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 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. (2018b), summarize observer data from 2003 to 2017 to estimate the catch and catch composition of the longline fisheries of the western and central Pacific Ocean. Observer coverage levels in the region are generally less than 5% and observer coverage can be expressed in a variety of units (e.g. trips with observers on board, hooks with observer onboard, hooks observed, Peatman et al. 2018b). Observer coverage over the whole Convention Area (Figure 41) tends to be consistent from 2003–2010 (1 to 1.5%) before reaching a maximum of ~4.5% in 2013 and then varying between 2 and 4% up to 2017 (Peatman et al. 2018b). Since the 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.

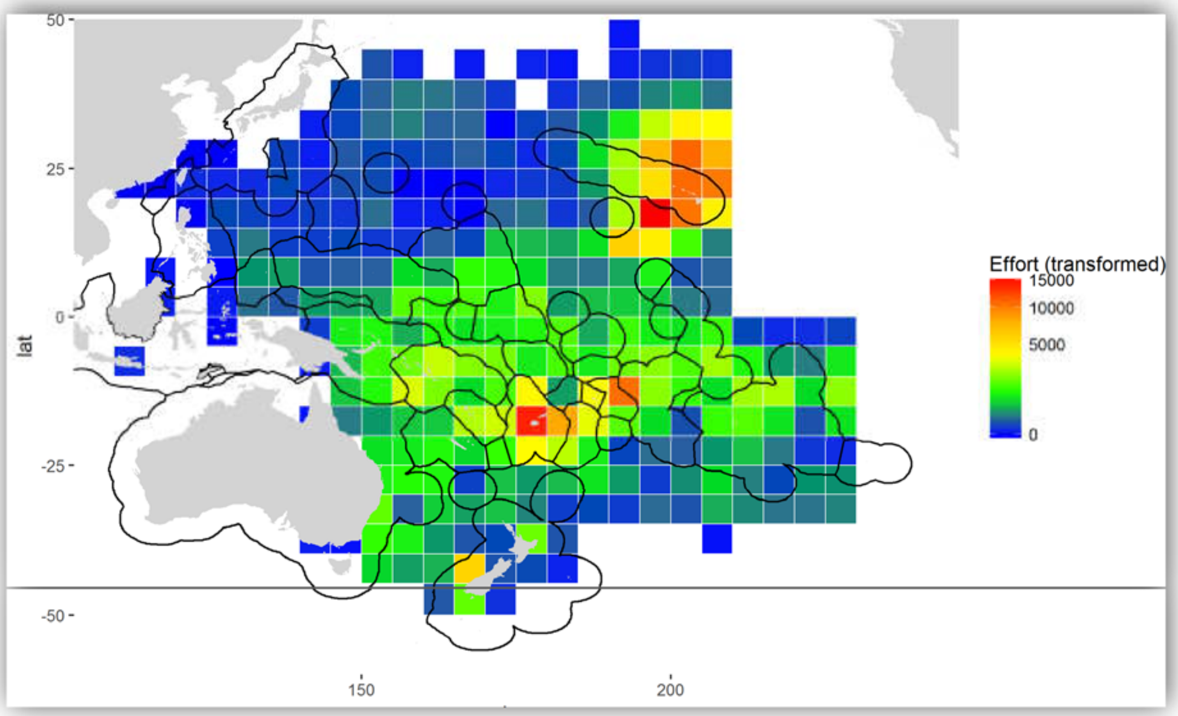


Figure 41. Observed effort in number of hooks (square root transformed) for longliners during the 2003-2016 time period in the WCPFC convention area.

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

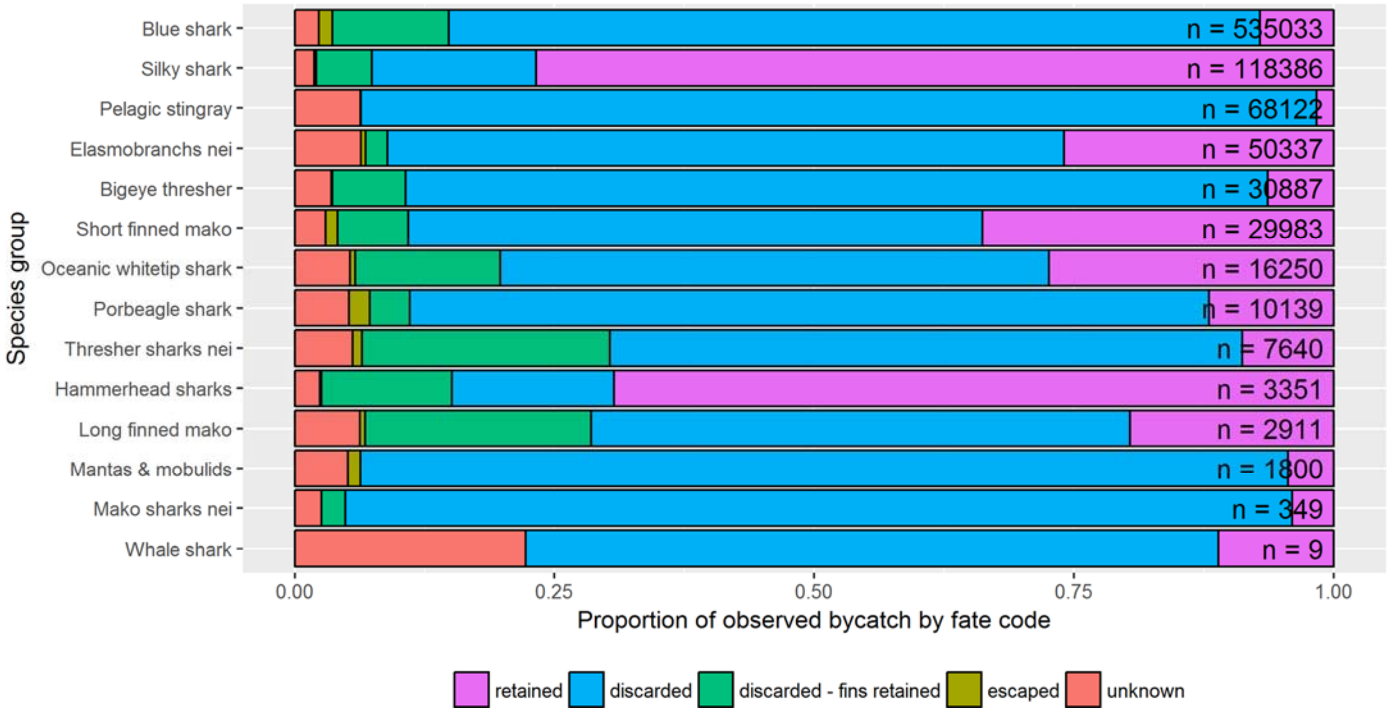


Figure 42. Recorded fate of observed sharks and rays catch by species/species group, as a proportion of total observed catch (number of fish) for the species/species group in the longline fisheries. The number of records is provided (Source: Peatman et al. 2018b).

As evident in the above figure, very few mantas or mobulas were retained, the majority captured were discarded. In Figure 43 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).

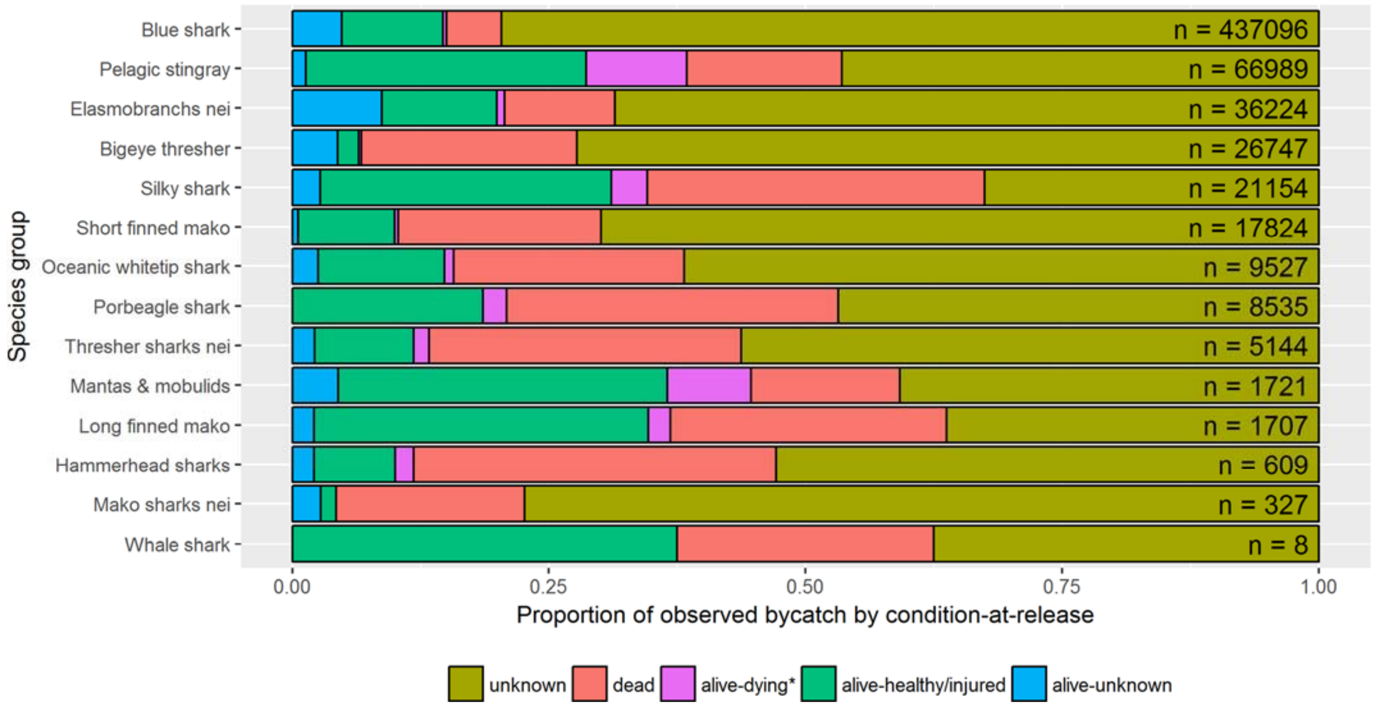


Figure 43. Recorded condition at release of observed sharks and rays catch by species/species group, as a proportion of total observed catch (number of fish) for the species/species group in the longline fisheries. The number of records is provided for each species/group. Note – alive-dying* is individuals that were alive but considered unlikely to survive. Source: Peatman et al. 2018b).

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, HI SSLL 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 from 2004-2018 (NMFS Observer Program unpublished data). Observer data (100% coverage) for the HI SSLL fishery recorded 21 interactions with giant manta rays from 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 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 as cited in Miller and Klimovich 2016). Bycatch per set ranged from 0.001 individuals (in log associated sets) to 0.027 individuals (in school associated sets) (Hall unpublished data as cited in Miller and Klimovich 2016).

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 Nino 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 (California/Oregon Drift Gillnet Fishery Catch Summaries).

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 as cited in Miller and Klimovich 2016; 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 et al. 2017). However, driven by the international trade in gill plates, fisheries targeting mantas have expanded and pose a serious threat to the giant manta ray (CITES 2013). 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. 2017), 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. 2017) 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 U.S. $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, (CITES 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 (Manta Trust 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 (Queensland shark control program 2018). 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 (Queensland shark control program 2018).

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 et al. 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 2015a, 2015b).

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 27. 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 27. 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, Indonesia	*Manta spp.*	Shark and Ray Sanctuary Bupati Decree 2010

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. (2017), 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. 2017).

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 indicate 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 44, 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.

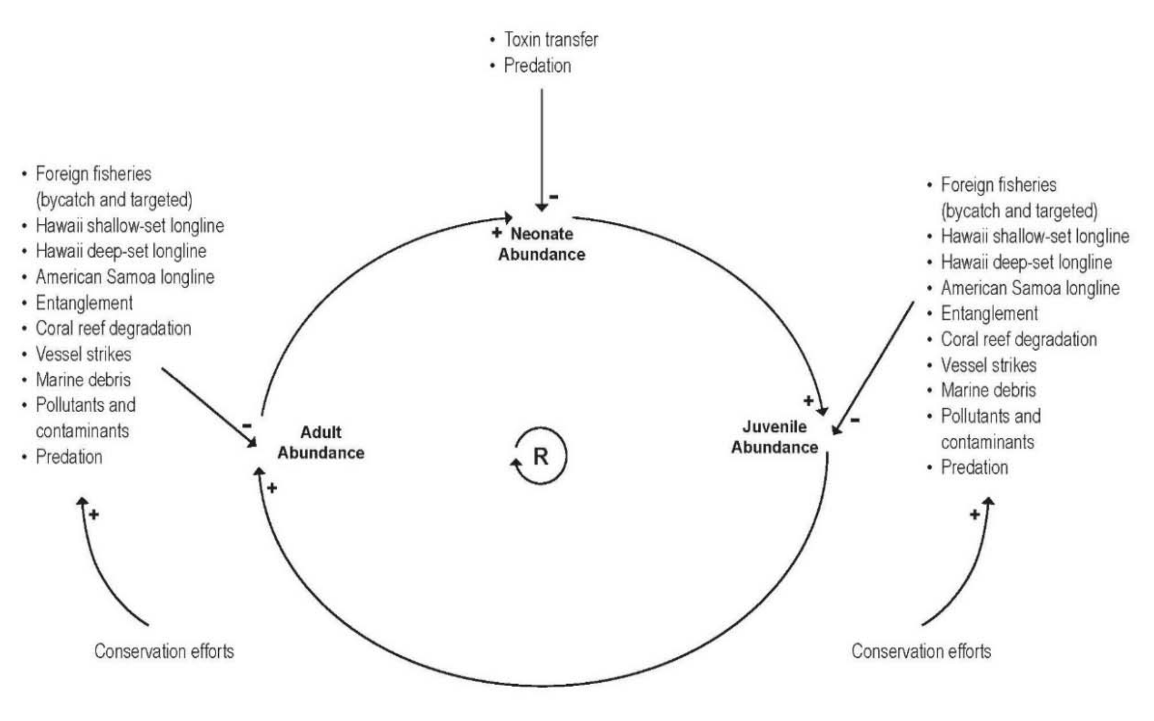


Figure 44. Giant manta ray casual loop diagram depicting various stressor-response relationships for each life cycle stage.

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; 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 45). 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; Elorriaga-Verplanken et al. 2016a; Carretta et al. 2017b; 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. 199; 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 46). Lambourn et al. (2009) suggests the northernmost location this species has been documented was Katcemak Bay in Alaska.

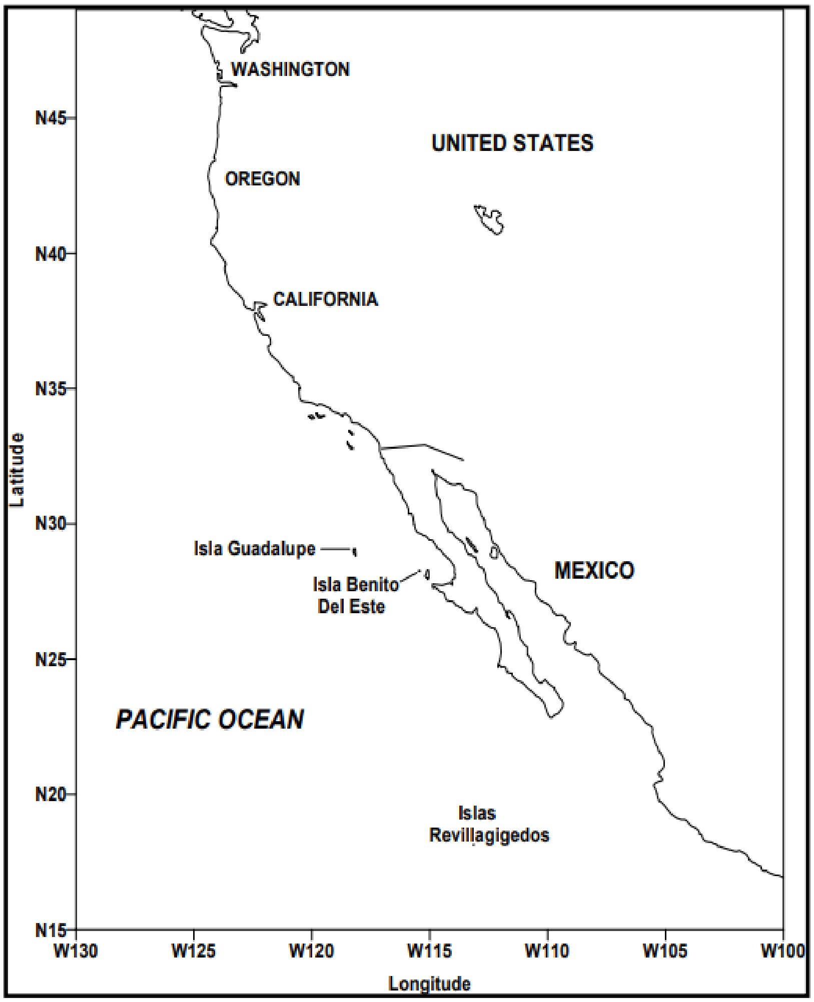


Figure 45. Geographic range of the Guadalupe fur seal, showing location of the two rookeries at Isla Guadalupe and Isla Benito Del Este. Source: Carretta et al. 2017b

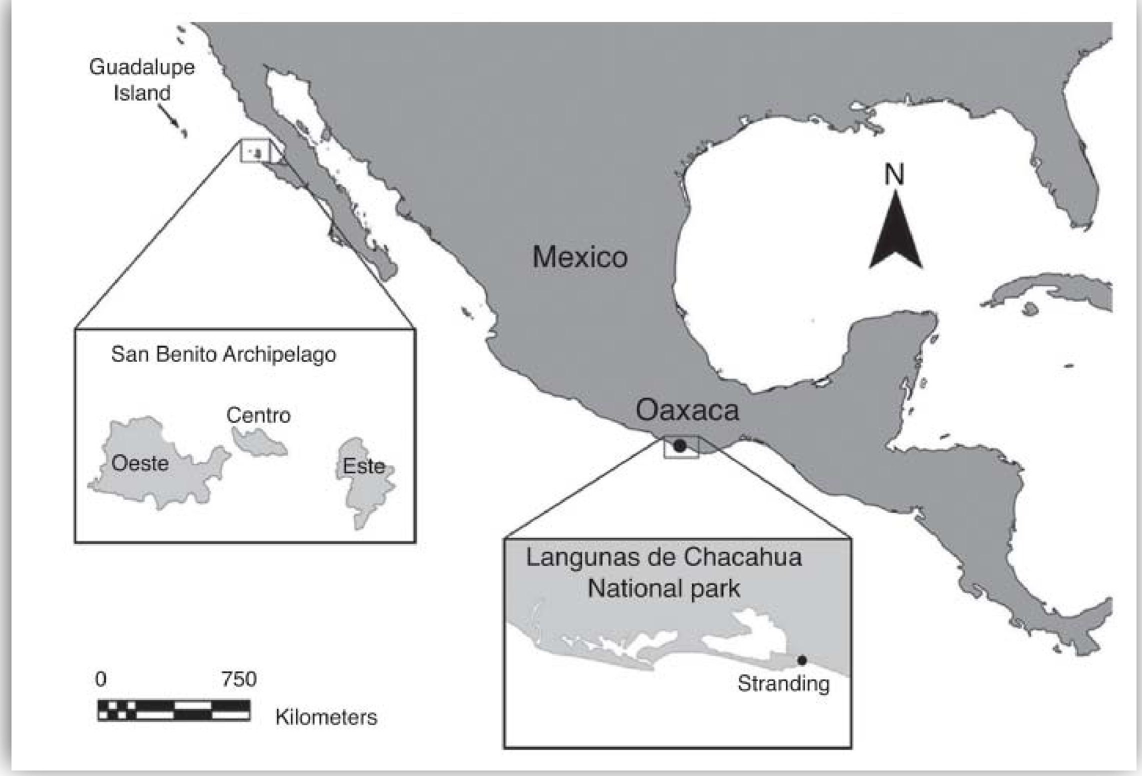


Figure 46. Distribution of Guadalupe fur seals, Arctocephalus townsendi, indicating the breeding (Guadalupe Island) and recolonization (San Benito Archipelago) sites, as well as the atypical stranding in Parque Nacional Lagunas de Chacahua in Oaxaca, Mexico. Source: Villegas-Zurita et al. 2015.

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 as cited in Carretta et al. 2017b). 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 50 m 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 47).

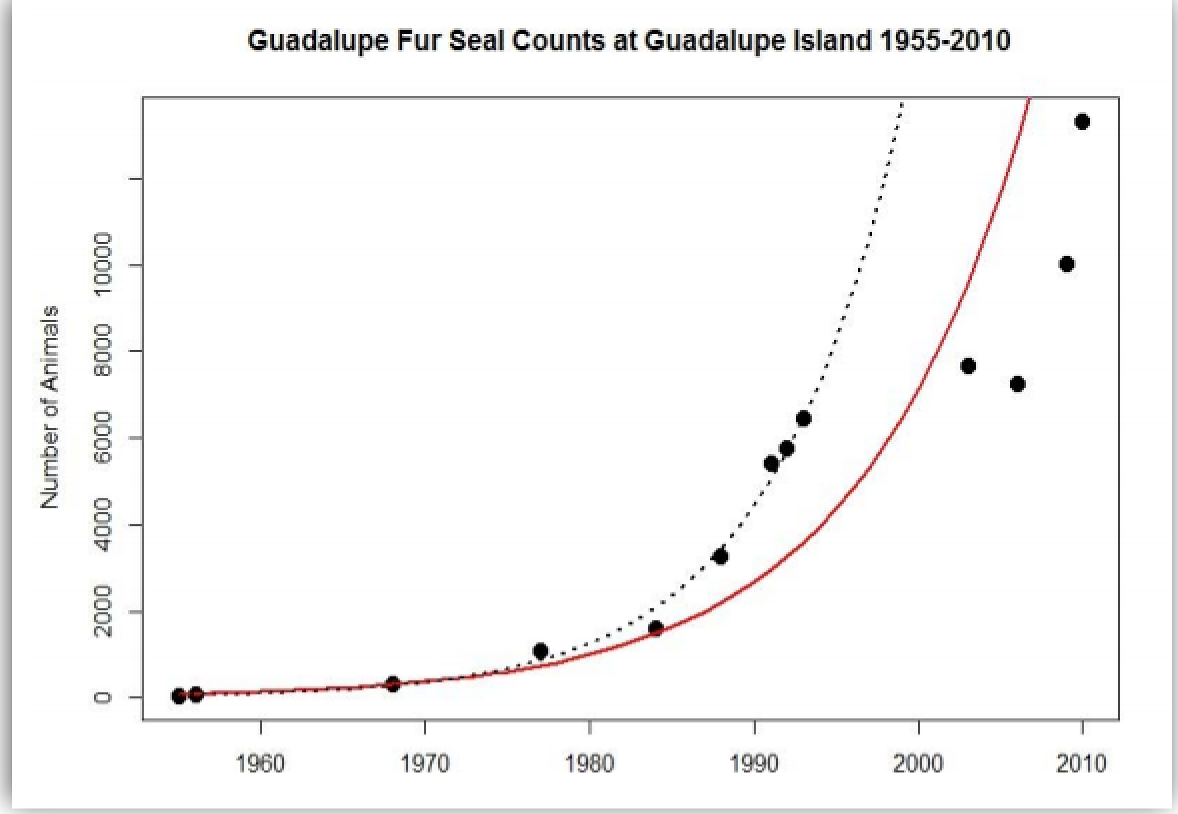


Figure 47. Counts of Guadalupe fur seals at Guadalupe Island Mexico, and the estimated population growth curves derived from counts made during the breeding season. Direct counts of animals are shown as black dots. An estimated annual growth rate of 13% is based on counts made between 1955 and 1993 (black dashed line). The estimated growth rate over the period 1955-2010 is approximately 10% annually (solid red line). Source: Carretta et al. 2017b.

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 2017c; 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 48 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).

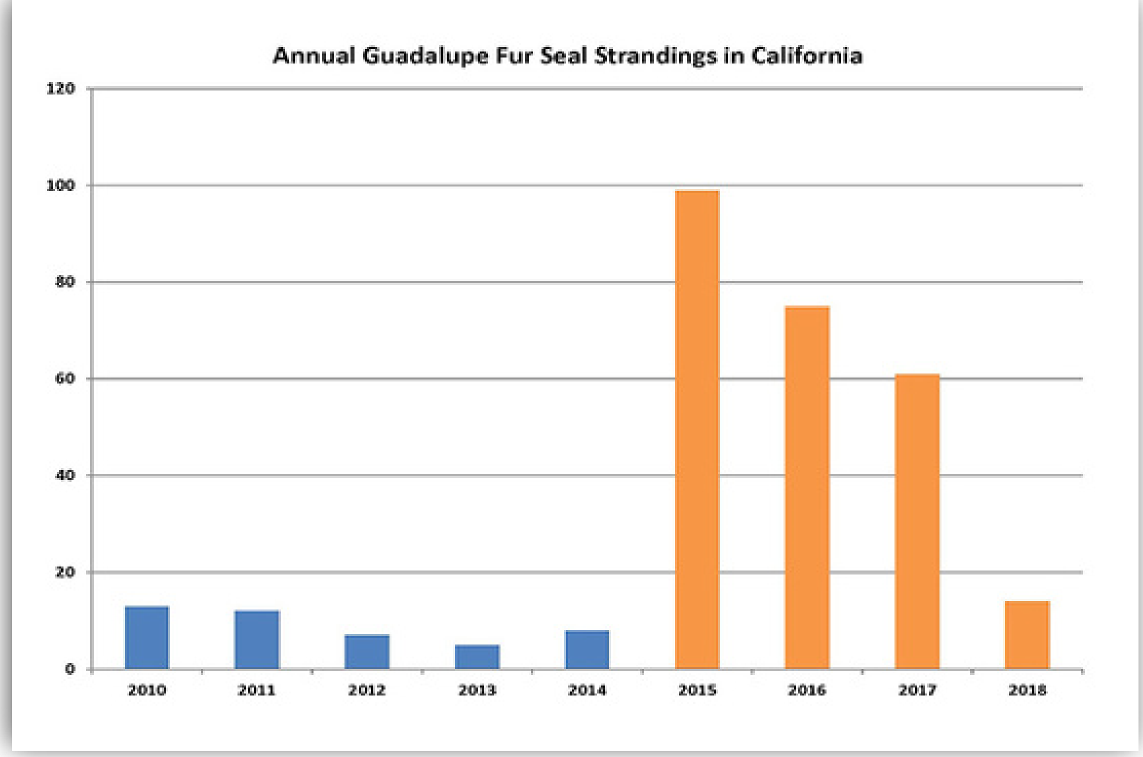


Figure 48. Annual Guadalupe fur seal strandings in California from 2010-2018.

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. 2017a). According to Carretta et al. (2017b) 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 HI SSLL 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 49).

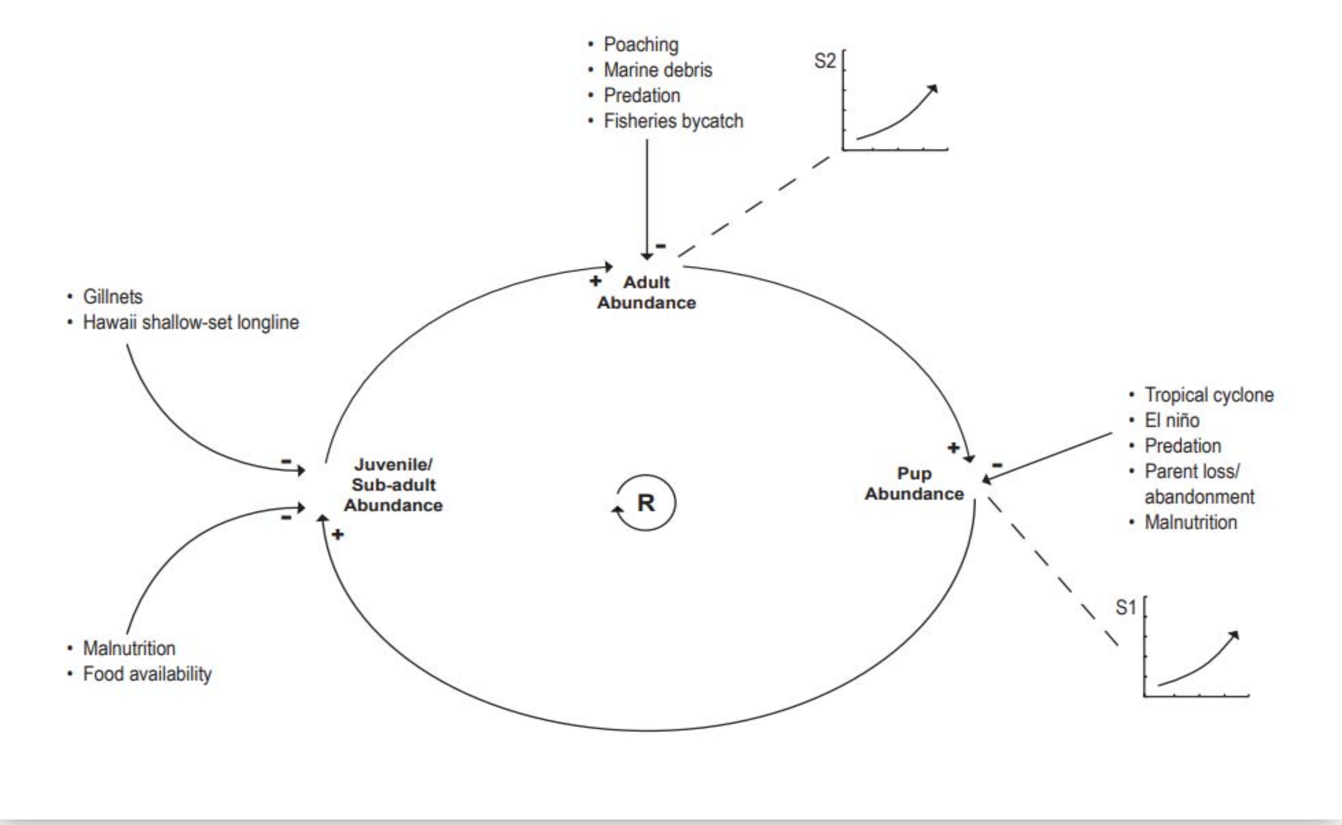


Figure 49. Guadalupe fur seal causal loop diagram depicting various stressor-response relationship for each life cycle stage.

Our graphic illustrates the population behavior of Guadalupe fur seals in S1 and S2, which is based on available rookery count data (Figure 49). 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 a biological opinion includes the past and present impacts of all state, federal or private actions and other human activities in the *Action Area*, anticipated impacts of all proposed federal projects in the *Action Area* that have already undergone formal or early section 7 consultation, and the impact of state or private actions which are contemporaneous with the consultation in process (50 CFR 402.02). The Consultation Handbook further clarifies that the environmental baseline is "an analysis of the effects of past and ongoing human and natural factors leading to the current status of the species, its habitat (including designated critical habitat), and ecosystem, within the *Action Area*" (FWS and NMFS 1998). The purpose of describing the environmental baseline in this manner in a biological opinion is to provide context for effects of the proposed action on listed species.

The past and present impacts of human and natural factors leading to the status of the 12 ESA-listed marine species addressed by this biological opinion within the *Action Area* include fishery interactions, vessel strikes, climate change, pollution, marine debris, and entanglement. The environmental baselines for these species addressed by this biological opinion are described below.

Information in this section is summarized from the several past biological opinions on the HI SSLL fishery and other fisheries that occur within the same *Action Area* as the HI SSLL fishery (NMFS 2004a, 2004b, 2005, 2012b, 2014b). We also used the 2017 pelagics report (WPRFMC 2018), the green sea turtle 5-year status review (Seminoff et al. 2015), the 2009 loggerhead sea turtle status review (Conant et al. 2009), the 2011 loggerhead sea turtle listing (76 FR 58868), the olive ridley sea turtle 5-year status review (NMFS and FWS 2014), 2017 status review report for oceanic white tip sharks (Young et al. 2017), 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 (McCarty 2001; IPCC 2014). Climate change will likely have its most pronounced effects on vulnerable species whose populations are already in tenuous positions (Williams et al. 2008). As such, we expect the risk of extinction for ESA-listed species to rise with the degree of climate shift associated with global warming. Increasing atmospheric temperatures have already contributed to documented changes in the quality of freshwater, coastal, and marine ecosystems and to the decline of endangered and threatened species populations (Mantua et al. 1997; Karl et al. 2009).

Marine species ranges are expected to shift as they align their distributions to match their physiological tolerances under changing environmental conditions (Doney et al. 2012). Climate-related shifts in range and distribution have already been observed in some marine mammal populations (Silber et al. 2017). Marine mammal species often exhibit strong dependence on or fidelity to particular habitat types, oceanographic features, and migration routes (Sequeira et al. 2018). Specialized diets, restricted ranges, or reliance on specific substrates or sites (e.g., for pupping) make many marine mammal populations particularly vulnerable to climate change (Silber et al. 2017). Marine mammals with restricted distributions linked to water temperature may be exposed to range restriction (Learmonth et al. 2006; Isaac 2009). MacLeod (2009) estimated that, based upon expected shifts in water temperature, 88% of cetaceans would be affected by climate change, 47% would be negatively affected, and 21% would be put at risk of extinction. Hazen et al. (2012) examined top predator distribution and diversity in the Pacific Ocean in light of rising sea surface temperatures using a database of electronic tags and output from a global climate model. Hazen et al. (2012) predicted up to a 35% change in core habitat area for some key marine predators in the Pacific Ocean, with some species predicted to experience gains in available core habitat and some predicted to experience losses. 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 (Pike 2014; Bone et al. 2015).

Significant impacts to marine mammals and sea turtles from ocean acidification may be indirectly tied to foraging opportunities resulting from ecosystem changes (Busch et al. 2013; Haigh et al. 2015; Chan et al. 2017). Nearshore waters off California have already shown a persistent drop in pH from the global ocean mean pH of 8.1 to as low as 7.43 (Chan et al. 2017). The distribution, abundance and migration of baleen whales reflects the distribution, abundance and movements of dense prey patches (e.g., copepods, euphausiids or krill, amphipods, and shrimp), which have in turn been linked to oceanographic features affected by climate change (Learmonth et al. 2006). Ocean acidification may cause a shift in phytoplankton community composition and biochemical composition that can impact the transfer of essential nutrients to predators that eat plankton (Bermudez et al. 2016). Blue whales, as predators that specialize in eating krill, are likely to change their distribution in response to changes in the distribution of krill (Payne et al. 1986, 1990; Clapham et al. 1999). 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 (Newson et al. 2009; Patricio et al. 2017; Jensen et al. 2018). 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. 2015). 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, 128 unknown/indeterminable) (Franke and TRP Staff 2015). However, preliminary (unpublished) data from Allen et al. (2015) 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 (Mazaris et al. 2008; Reina et al. 2008; Robinson et al. 2008; Fuentes et al. 2009a). 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 (Fuentes et al. 2009b, 2010a, 2010b, 2011; Azanza-Ricardo et al. 2017).

Other climatic aspects, such as extreme weather events, precipitation, ocean acidification and sea level rise also have potential to affect marine turtle populations. Changes in global climatic patterns will likely have profound effects on the coastlines of every continent, thus directly impacting sea turtle nesting habitat (Wilkinson and Souter 2008). In some areas, increases in sea level alone may be sufficient to inundate turtle nests and reduce hatching success by creating hypoxic conditions within clutches (Caut et al. 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 50). 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 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 50. The *Action Area* is shown by the red rectangle, and consists mostly of high seas areas, although some effort still occurs within the U.S. EEZ.

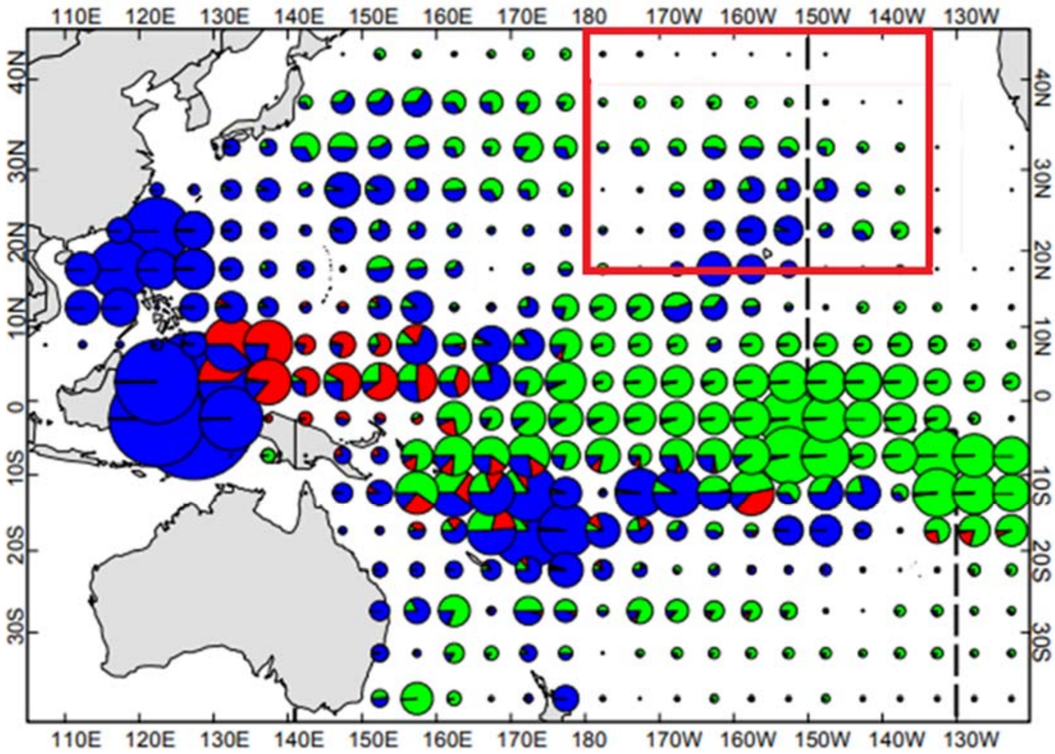


Figure 50. Distribution of longline effort for distant water-fleets (green), foreign-offshore fleets (red) and domestic fleets (blue) for the period of 2000-2016. Source: Williams et al. 2017. The *Action Area* for the HI SSLL fishery where fishing occurs is highlighted by the red rectangle and consists mostly of international waters. The dashed black line at 150W represents the boundary between the WCPFC (west of the line) and the 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, likely 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, important international CMMs have resulted in more recent years. 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 leatherback sea turtles 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 28). 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 28). There were 325 green sea turtles reported with an estimate of 6,500 green sea turtles caught in the region from 1989-2016 (Table 28). 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 28).

Table 28. Turtle interactions reported for the ABNJ workshop from 16 countries from 1989-2015 and expanded based on 5% coverage in the WCPFC.

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

Table 29 and Table 30 show estimates for sea turtle interactions north of 10°N 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. (2018b). 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 these data already include 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 10°N (Peatman et al. 2018b). Lastly, the data have wide confidence intervals for sea turtles (Peatman et al. 2018b). Peatman (2018b) 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 31 and Table 32.

Table 29. Low, median, and high (95%) confidence intervals for loggerhead sea turtles in the region north of 10°N. Adapted from Tables 19 and 32 in Peatman et al. 2018b. (‘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

Year	Low	Med	High (95%)
2013	322	804	1987
2014	765	1785	4168
2015	1220	2636	5899
2016	819	1905	4398
2017	140	555	2178
Totals	6,619	16,263	41,180

Table 30. Median estimates for olive ridley, green, and leatherback sea turtles reported for the region north of 10°N. Adapted from Table 32 in Peatman et al. 2018b.

Year	Olive Ridley Sea Turtle	Green Sea Turtle	Leatherback Sea Turtle	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

Year	Olive Ridley Sea Turtle	Green Sea Turtle	Leatherback Sea Turtle	Turtles nei
2017	3561	2421	299	4
Totals	55,169	22,089	9,923	5,220

Table 31. 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. 2018b	6,619-41,180	2003-2017	473-2941

Table 32. 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. 2018b	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. 2007b; Swimmer et al. 2017). Between 2004 and 2018, the HI SSLL fishery captured 105 leatherback sea turtles, and an estimated 21 leatherback sea turtles have died from their capture based on the post-hooking criteria (Ryder et al. 2006; NMFS unpublished data). Since 2012 the anticipated level of interactions in the HI SSLL fishery has been 26 annually and 54 have been captured since 2012 (Table 33). The HI DSLL fishery is anticipated to interact with up to 72 leatherback turtles (with 27 anticipated mortalities) over a three-year period (NMFS 2014b). Between 2005 and 2018, an estimated 146 interactions have occurred in the HI DSLL fishery, with 85 occurring since 2012 (Table 34) (WPRFMC 2018). Since 2005, the Hawaii longline fisheries combined have reduced their estimated mortality to an average of seven annually (Table 33).

Table 33. 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 et al. (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. 2007b) 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 26 loggerhead sea turtles died out of the 177 loggerhead sea turtles that were captured. Two loggerhead sea turtles came up dead on the line.

Table 34. 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 et al. (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)**
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)	-

From 2004-2018 the HI SSLL fishery captured 10 each of olive ridley and green sea turtles, with no mortalities of either (NMFS unpublished data). Since 2005, the HI DSLL fishery has caught approximately 686 olive ridley sea turtles, and there have been an estimated 650 mortalities (NMFS 2019b). Since 2005, the HI DSLL fishery has caught approximately 71 green sea turtles, and there have been an estimated 66 mortalities (NMFS 2019b).

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 Nino events in order to reduce bycatch of loggerhead turtles that move further north on the warm El Nino 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 35. 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 et al. (2006) post-hooking mortality criteria.

Fishery	**ITS Olive Ridley C (m)**		**Olive ridley 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	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 2016a). 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 36. 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 et al. (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	10 yr	1 (1)	0 (0)	0(0)	0(0)	0(0)	0(0)	0(0)	-

Table 37. Hawaii deep-set longline fishery 3-year ITS from 2017 Supplemental opinion for Green sea turtle DPSs percentage assigned to each DPS in parentheses.

Green Sea Turtle Species	**Captures**	**Mortalities**
East Pacific (70)	12	12
Central North Pacific (12)	6	6
East Indian-West Pacific (8)	6	6
Southwest Pacific (7)	6	6
Central West Pacific (1)	3	3
Central South Pacific (1)	3	3

Table 38. Green sea turtle interactions by species (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 et al. (2006) post-hooking mortality criteria.

Green Sea Turtle Species	2012	2013	2014	2015	2016	2017	2018
East Pacific	0 (0)	4 (3)	11 (11)	3 (3)	4 (3)	11 (10)	10 (9)
Central North Pacific	0 (0)	1 (0)	2 (2)	1 (1)	1 (1)	2 (2)	2 (2)
East Indian-West Pacific	0 (0)	0 (0)	1 (1)	0 (0)	0 (0)	1 (1)	1 (1)
Southwest Pacific	0 (0)	0 (0)	1 (1)	0 (0)	0 (0)	1 (1)	1 (1)
Central West Pacific	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Central South Pacific	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 2016). 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. (2017) 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. 2017). 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. (2018b) for multiple regions of the WCPFC using longline observer data. Table 39 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. (2018b). It should be noted that these data already include 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. 2018b). Lastly, the data have a wide confidence interval for key shark species (Peatman et al. 2018b).

Table 39. Median oceanic whitetip shark catch estimates (‘000 individuals) north of 10°N. Source: Peatman et al. 2018b.

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
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 tons of sharks were caught within the EPO for 2016, however the data are 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. No stock assessments currently exist for the Eastern Pacific.

5.2.2.2 Other U.S. Fisheries Oceanic Whitetip Shark Bycatch

Table 40 lists the number of interactions of oceanic whitetip sharks that have occurred in the HI DSLL fishery between 2002 and 2017. Interactions with the HI SSLL fishery are addressed in the *Effects of the Action*. The total number of observed sharks was 5,815 individuals with an expanded estimate of 26,967 sharks over this time period (McCracken 2019 in review).

Table 40. Estimated take levels for oceanic whitetip sharks (observed and estimated numbers) captured in the Hawaii deep-set longline fishery between 2002 and 2017. Source: McCracken 2019 in review.

Year	Observed	Estimated Interactions
2002	840	3574
2003	524	2515
2004	718	2938
2005	341	1282
2006	331	1346
2007	262	1341
2008	144	741
2009	244	1236
2010	252	1198
2011	225	1176
2012	172	878
2013	196	973
2014	370	1670
2015	531	2654
2016	423	2188
2017	242	1257
Total	5815	26967

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. 2017). 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. (2017) 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 unpublished 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 (C. Fahy and C. Villafana pers. comm. March 7, 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, 3 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.

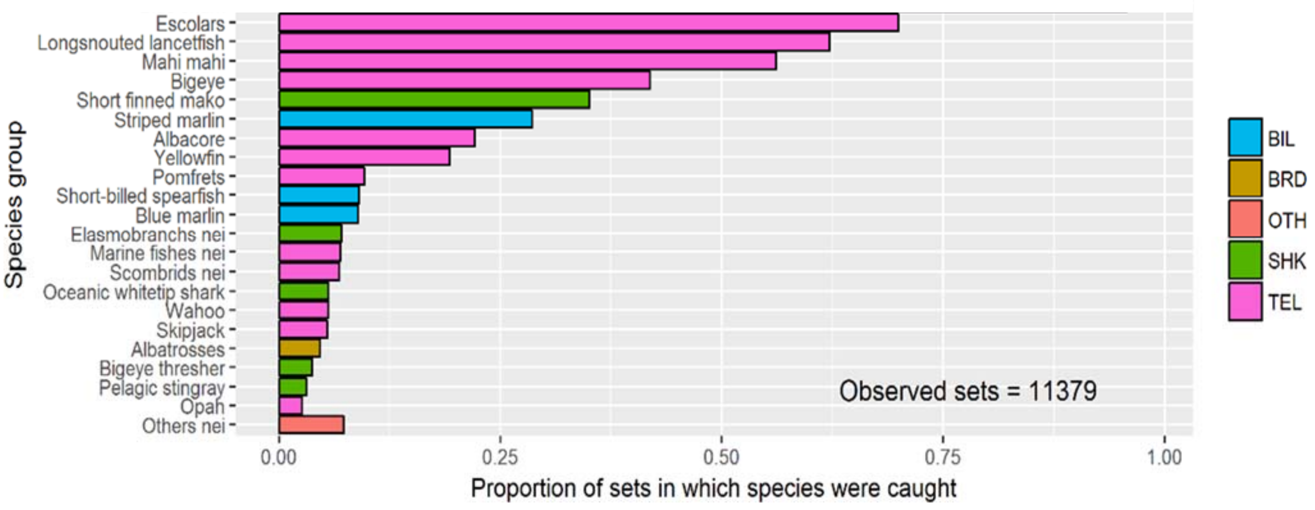
5.2.3 Giant Manta Ray

5.2.3.1 International Fisheries Giant Manta Ray Bycatch

As mentioned previously, the *Action Area* overlaps with two RFMOs, 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 are the best data available. A report from Peatman et al. (2018b), 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 PNG and Solomon Islands) therefore, they are included in the status section. Peatman et al. (2018b) also separate bycatch catch data and estimates at the regional level.

The catch and estimate data for North temperate fisheries (areas North of 10°N) 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. (2018b) combined mantas and mobulas into a larger category of elasmobranchs not included elsewhere. As evident in Figure 51 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

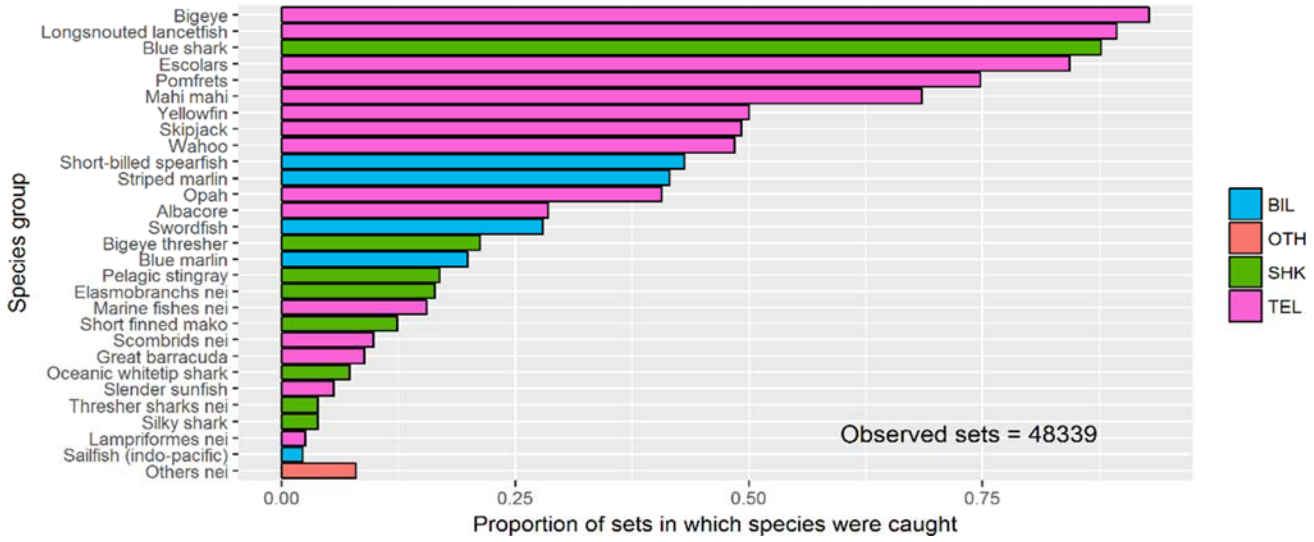


Figure 51. The proportion of longline shallow sets and deep sets in north temperate fisheries with observed catch against species/species group. Rarely observed species have been grouped in to ‘others nei’. Bar color denotes billfish (BIL), marine mammals (MAM), others nei (OTH), shark species (SHK), teleosts or fish (TEL), turtles (TTX) and tuna (TUN). Source: Peatman et al. 2018.

Peatman et al. (2018b) also provided annual catch estimates at the regional levels. These estimates are based on the limitations of the data set 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 41. Median shark and ray catch estimates (number of individuals) by species/species group and region. This region of North of 10°N. Source: Peatman et al. 2018b.

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 42 lists all observed and estimated giant manta ray, *Mobulidae spp*. and unidentified ray interactions in the DSLL fishery from 2004 to 2017. This table did not include data for 2018, therefore, the following computations for the giant manta ray interactions in the DSLL fishery are from 2004-2017.

Table 42. Observed and estimated annual takes using an estimator for 2004-2017 annual take levels based on a complex adaptive sample design to account for the 20% coverage in the DSLL fishery (McCracken 2019).

Year	**Giant Manta Ray**		**Mobulidae spp.**		**Unidentified Ray**	
	Observed	**Estimated**	**Observed**	**Estimated**	**Observed**	**Estimated**
2004	1	3	8	39	0	0
2005	2	7	0	0	0	0
2006	2	11	2	21	0	0
2007	2	5	6	31	1	1
2008	2	10	2	10	1	5
2009	4	23	3	19	3	20
2010	17	95	1	6	2	5
2011	1	5	2	9	0	0
2012	2	11	1	6	2	12
2013	1	5	0	0	4	21
2014	3	11	4	16	0	0
2015	2	10	5	25	4	21
2016	4	22	3	16	1	4
2017	0	0	5	26	1	7

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 observed 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*. Finally, we incorporated McCracken 2019 estimates in Table 47 to account for the 20% observer coverage. As a result, we estimated the HI DSLL fishery had 266 interactions with giant manta rays (218 observed plus 39 *Mobulidae* spp. and 9 unidentified rays) from 2004-2017. We estimated the HI SSLL fishery had 21 interactions with giant manta rays (17 observed plus 4 (unidentified) giant manta rays) from 2004-2017.

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). However, we do not have expanded fleet estimates for the giant manta ray, *Mobulidae* spp. and unidentified rays to account for all quarters of observer coverage from 1994 to 2004. Therefore this estimated number of interactions prior to 2004 is very low and is the minimum number of interactions possible with the giant manta ray. Overall, we estimate that the Hawaii based fisheries have had at least 329 interactions with giant manta rays from 1994-2017.

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 Nino 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, 2017b 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. 2017b 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 (C. Fahy and C. Villafana pers. comm. March 7, 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 Galapagos 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 2018e). 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 green sea turtle 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 43) due to vessel strikes which will continue to occur into the future (NMFS 2018e). 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 2017a).

Table 43. 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, 2010). Nearshore spills or large offshore spills that reach shore can oil beaches on which sea turtles lay their eggs, causing birth defects or mortality in the nests (NOAA 2003, 2010). Disruption of other essential behaviors, such as breeding, communication, and feeding may also occur.

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

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 observers 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 (Lutcavage et al. 1997; Laist et al. 1999).

Santos et al. (2015) found that a small amount of plastic debris was sufficient to block the digestive tract and cause death. 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. (2016) estimates that 52% of sea turtles globally have ingested plastic debris. Schuyler et al. (2016) 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 2015a, b).

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 2003b). These activities occur within the *Action Area* to varying degrees throughout the year. ESA-listed species have the potential to be impacted by increased levels of both background sound and high intensity, short-term sounds. Sources of anthropogenic noise are becoming both more pervasive and more powerful, increasing both oceanic background sound levels and peak intensity levels (Hildebrand 2004).

This section is divided into subsections addressing the impacts to listed species in the *Action Area* from the following major sources of anthropogenic sound sources: vessels and commercial shipping; seismic surveys; military 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 (NRC 2003b; Hildebrand 2009; Mckenna et al. 2012). Shipping constitutes a major source of low-frequency (five to 500 Hz) sound in the ocean (Hildebrand 2004), particularly in the Northern Hemisphere where the majority of vessel traffic occurs. While commercial shipping contributes a large portion of oceanic anthropogenic noise, other sources of maritime traffic can also impact the marine environment. These include recreational boats, whale-watching boats, research vessels, and ships associated with oil and gas activities.

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. 1995). There is also evidence that nearby vessels and construction noise affect haulout times during pupping season for some pinniped species (Jansen et al. 2010; Anderwald et al. 2013; Karpovich et al. 2015). 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.

5.4.3.2 Seismic Surveys

Offshore seismic surveys involve the use of high energy sound sources operated in the water column to probe below the seafloor. Seismic surveys primarily for scientific research have been conducted in the *Action Area* over the past several decades (NMFS 2018d).

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

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

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 green sea turtles, 65 leatherback sea turtles, 61 North Pacific loggerhead sea turtles, and 32 olive ridley sea turtles will be harassed during the survey around Hawaii (NMFS 2018d).

Elasmobranchs, like all fish, have an inner ear capable of detecting sound and a lateral line capable of detecting water motion caused by sound (Hastings and Popper 2005; Popper and Schilt 2008). Data for elasmobranch fishes suggest they are capable of detecting sounds from approximately 20 Hz to 1 kHz with the highest sensitivity to sounds at lower ranges (Casper et al. 2003, 2012; Casper and Mann 2006, 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 are 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.

5.4.3.3 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 2018e). 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 2018e).

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 2018e). 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 44). 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 44. 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 2018e).

Species	**Impulsive and Non-Impulsive Acoustic Stressors**		
	Harassment (TTS/ Behavioral)	**Harm (PTS)**	**Harm (Slight Lung Injury)**
Guadalupe Fur Seal	15 / 1,442	0	none
Central North Pacific Green Sea Turtle	20 / 1,831	7	1
Olive Ridley Sea Turtle	0 / 96	none	none
North Pacific Loggerhead Sea Turtle-	0 / 182	none	none
Leatherback Sea Turtle	0 / 193	none	none

5.5 Synthesis of Baseline Impacts

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

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

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

The stress regime created by the activities discussed in this *Environmental Baseline* continues to have a serious and adverse impact on leatherback sea turtles. Leatherbacks face high probability of extinction as a result of both environmental and demographic stochasticity. Demographic stochasticity, or chance variation in the birth or death of an individual of the population, is facilitated by the increases in mortality rates of leatherback populations resulting from the premature deaths of individual sea turtles associated with human activities (or that die as a result of being captured in fisheries) or incidental capture and mortality of individuals in various fisheries. The information available suggests that leatherback sea turtles have high a probability of becoming extinct in the Pacific Ocean unless they are protected from the combined threats of entanglement in fishing gear and overharvest. The number of individuals that continue to be captured and killed in fisheries in the *Action Area* contributes to the increased extinction risk of the species.

Of the other activities and their associated stressors, the propensity of vessel strikes to go unnoticed or unreported by vessel operators impedes an accurate assessment of the magnitude this threat poses to leatherback sea turtles. However, leatherback sea turtles occur in the pelagic waters within the *Action Area* where their density is sparse in comparison to nearshore turtle habitats where as a result of a higher density of turtles, there is an increased risk of a vessel strike. Therefore, we do not expect vessel strikes to contribute to the increased extinction risk of the species in the pelagic environment but in nearshore areas the risk of vessel strike likely increases.

Of the different types of pollution that can adversely affect leatherback sea turtles, the ingestion and entanglement risk of marine debris may pose the most serious threat especially since the Subtropical Convergence Zone (STCZ), a known area of marine debris aggregation (Kubota 1994; Pichel et al. 2007; Maximenko et al. 2012), is within the *Action Area*. Due to increased biological productivity in the STCZ, it has become a significant foraging and migration corridor for swordfish (*Xiphias gladius*) (Seki et al. 2002) and sea turtles (Polovina et al. 2004; Howell et al. 2008, 2010). Ingestion of plastic debris can block the digestive tract which can cause turtle mortality as well as sub-lethal effects including dietary dilution, reduced fitness, and absorption of toxic compounds (Lutcavage et al. 1997; Laist et al. 1999). Entanglement in marine debris could result in substantial wounds, including cuts, constriction, or bleeding on any body part. In addition, entanglement could directly or indirectly interfere with mobility, causing impairment in feeding, breeding, or migration. The number of individuals that continue to ingest and become entangled in marine debris in the *Action Area* contributes to the increased extinction risk of the species.

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

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

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

Of the different types of pollution that can adversely affect loggerhead sea turtles, the ingestion and entanglement risk of marine debris may pose the most serious threat especially since the STCZ, a known area of marine debris aggregation (Kubota 1994; Pichel et al. 2007; Maximenko et al. 2012), is within the *Action Area*. Due to increased biological productivity in the STCZ, it has become a significant foraging and migration corridor for swordfish (*Xiphias gladius*) (Seki et al. 2002) and sea turtles, specifically loggerhead sea turtles (Polovina et al. 2004; Howell et al. 2010, 2008). Ingestion of plastic debris can block the digestive tract which can cause turtle mortality as well as sub-lethal effects including dietary dilution, reduced fitness, and absorption of toxic compounds). Entanglement in marine debris could result in substantial wounds, including cuts, constriction, or bleeding on any body part. In addition entanglement could directly or indirectly interfere with mobility, causing impairment in feeding, breeding, or migration. The number of individuals that continue to ingest and become entangled in marine debris in the *Action Area* contributes to the increased extinction risk of the species.

We understand little about the impacts of noise or the underwater noise thresholds for sea turtles. However, their hearing ranges are generally limited and sea turtle ears are considered more primitive than that of marine mammals and less capable of detecting sound pressures (Popper et al. 2014). Since loggerhead sea turtles in the pelagic waters within the *Action Area* are generally moving and their exposure to a moving source of sound would be relatively short lived, ocean noise in the action area is not expected to contribute to the increased extinction risk of the species. The stress regime created by activities discussed in this *Environmental Baseline* continue to be a threat for the Central North Pacific and East Pacific 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 green sea turtle, along with vessel strikes and nearshore fisheries in the main Hawaiian Islands. Even with increasing populations these species of green sea turtles 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 species 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; Aurioles-Gamboa 2015; Elorriage-Verplancken et al. 2016a, 2016b; Pablo-Rodriguez et al. 2016; NMFS 2017c; Juarez-Ruiz et al. 2018).

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.

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

Of the different types of pollution that can adversely affect giant manta rays, the ingestion and entanglement risk of marine debris may pose the most serious threat especially since the STCZ, a known area of marine debris aggregation (Kubota 1994; Pichel et al. 2007; Maximenko et al. 2012), is within the *Action Area*. Because giant manta rays must filter hundreds to thousands of cubic meters of water daily to obtain adequate nutrition (Paig-Tran et al. 2013), they can ingest microplastics directly from the water or indirectly through their contaminated planktonic prey (Setala et al. 2014). Microplastics can prohibit adequate nutrient absorption and physically damage the digestive track (Germanov et al. 2018), they can harbor high levels of toxins and persistent organic pollutants and transfer these toxins to the animal once ingested (Worm et al. 2017). If entangled in marine debris, the giant manta ray is at risk of severing of the cephalic and pectoral fin, severe injuries that can lead to a reduction in feeding efficiency and even death. The number of individuals that continue to ingest and become entangled in marine debris in the *Action Area* contributes to the increased extinction risk of the species.

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

The stressors discussed in this *Environmental Baseline* are also a threat for the oceanic whitetip shark. Oceanic whitetip sharks face a high probability of becoming endangered as a result of both environmental and demographic stochasticity. Due to their life-history characteristics, oceanic whitetip sharks are more susceptible to the effects of high fishing exploitation. The information available suggests that oceanic whitetip sharks have high a probability of becoming extirpated in the Pacific Ocean unless they are protected from the combined threats of incidental 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.

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

Of the different types of pollution that can adversely affect oceanic whitetip sharks, the ingestion and entanglement risk of marine debris may pose the most serious threat especially since the STCZ, a known area of marine debris aggregation (Kubota 1994; Pichel et al. 2007; Maximenko et al. 2012), is within the *Action Area*. Ingestion of marine debris may lead to injury or possibly starvation, and entanglement in debris could directly or indirectly interfere with the shark's mobility, causing impairment in feeding, breeding, or migration. However, due to the foraging behavior of sharks and the limited reports on entanglement, marine debris in the *Action Area* is not expected to contribute to the increased extinction risk of the species.

Sharks may respond to noises by avoiding, halting their activities, experience reduced hearing by masking, or become attracted to source noises. Avoidance is most likely, and a common natural reaction and considered low risk. Nevertheless, sharks are large agile animals and capable of swimming away safely from disturbances that would harm them, therefore, ocean noise in the *Action Area* is not expected to contribute to the increased extinction risk of 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 NMFS has previously concluded that the proposed action is not likely to adversely affect several listed species and areas designated as critical habitat for listed species, these listed resources are not considered in the analyses that follow. Species and critical habitat not likely to be adversely affected by the proposed action are discussed in section 4.1 of this biological opinion, *Status of Listed Resources Not Considered Further*.

In 2004, the Hawaii longline fishery was split into shallow and deep set sectors 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 or a redistribution associated with change in forage base that is possibly climate-related. 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 effects analysis focuses on interaction data available from the past 14–15 years. These data form the backbone to our effects analysis.

6.1 Potential Stressors

Potential stressors associated with the proposed action include:

1. capture (hooking) in fishing gear,
2. entanglement in fishing gear,
3. interactions with derelict fishing gear (lines and hooks that have been lost, abandoned or discarded into marine waters),
4. being struck by fishing vessels,
5. vessel noise, and
6. vessel pollution, which includes discharges of solid waste, oils, air emissions, etc.

This list is not exhaustive; however, it includes all of the stressors that may adversely affect endangered or threatened species, their ecological interactions, or critical habitat that has been designated for listed species. At any point in time, a single vessel might be the source for one or more of these potential stressors and listed individuals might be exposed to one or more of these stressors.

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.

6.2 Exposure Analyses

As discussed in the *Approach to the Assessment* section of this opinion, exposure analyses are central to our assessment of the effects of actions. Exposure analyses are designed to identify which listed resources are likely to co-occur with stressors caused by an action, the nature of that co-occurrence, and interactions that result from that co-occurrence. As part of these analyses, we try to estimate the number, age (or life stage), and gender of the individuals that are likely to be exposed and identify the populations or subpopulations those individuals represent.

With the limited data currently available, we cannot estimate the number of endangered or threatened marine animals that are likely to be exposed to the HI SSLL fishery. All of the exposures occur underneath the ocean's surface where they are unobserved. We cannot estimate the number of sea turtles that are hooked or entangled by longline gear but escape before they are observed. Instead, our exposure analysis focuses on hooking and entanglements that have been observed and reported.

6.2.1 Hooking and Entanglement of Listed Species

Data on hooking and entanglements in the HI SSLL fisheries almost certainly underestimate the actual number of interactions because they cannot account for individuals that were hooked or entangled but either escaped or were never brought aboard a ship (for example, because of a line break or a predator). Despite several efforts to assess the significance of unobserved catch the number of unobserved interactions (for example, Moyes et al. 2006; Murray 2011; and Warden and Murray 2011; Gilman et al. 2013), the difference between the number of observed interactions and the actual number of interactions remains unknown.

6.2.1.1 Sea Turtles

Since 2004 there have been 302 records of sea turtles being captured by the HI SSLL. Fishery observers document how each turtle was captured (hooked, entangled or both) and whether the turtle died or was returned to sea injured (Table 50). Green and olive ridley sea turtles, each, represented 3% of the interactions, leatherback sea turtles represented 35% of the turtle interactions, and loggerhead sea turtles more than half (59%) of the total observed these interactions.

Table 45. Number sea turtles, by species, captured dead and alive in the Hawaii shallow-set longline fishery between 2004 and 2018.

Observed turtle status	Green sea turtle	Leatherback sea turtle	Loggerhead sea turtle	Olive Ridley sea turtle	Grand Total
Hooked or entangled	10	105	175	10	293
Dead	-	-	2	-	2
Grand Total	10	105	177	10	302
% of Total	3	35	59	3	100

NMFS estimates post release mortality of sea turtles captured by longline gear based on the turtles condition when released and six injury categories (Ryder et al. 2006; Swimmer and Gilman 2012). 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 ("comatose").

Table 46 describes the number of sea turtles captured alive in the HI SSLL fishery by injury category after applying the Ryder et al. (2006) post release mortality categories while Table 47 presents the same data as proportions. We use the values in Table 52 to assign injury categories for situations when the hooking location is unknown as a result of the line parting before the observer could determine where the animal was hooked. Further, these proportions are applied to the future interaction estimates to estimate mortality. 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.

Table 46. Numbers of all sea turtles by species and injury category that were captured alive in the Hawaii shallow-set longline fishery between 2004 and 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 47. Proportions of sea turtles by species and injury categories in the Hawaii shallow-set longline fishery between 2004 and 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 (e.g. 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 48 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 49 describes the proportion in each category. Overall, external hooking accounts for the largest category of interactions (85%). Thirty-seven percent of interactions resulted in the animal being externally hooked and released without gear. Whereas, 48% of the animals externally hooked were 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 48. 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 49. Proportion 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	-	-	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. Observer estimates suggest that leatherback sea turtles caught in this fishery are generally between 4 and 6 feet long; however, these estimates must be regarded with some caution due to the difficult conditions underwhich these estimates are made, and because there are few leatherback sea turtles captured by this fishery that have been actually measured. Nevertheless, best estimates suggest that the average size of a leatherback sea turtle captured in the HI SSLL fishery is about 145 centimeters (NMFS unpublished observer data; Siders 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, PNG, and the 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 (P. 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 (Siders 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 (Siders et al. 2018). Similarly, Benson et al. (2011) tagged turtles in the California foraging grounds at a ratio of 3:1 females to males. The IUCN recently used a sex ratio of 3:1 female to male for their assessment to account for adult males, which is based on the work of Tapilatu and Tiwari (2007 as cited in Tiwari et al. 2013).

Spatial and Temporal Patterns of Exposure

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). Siders et al. (2018) developed a model to predict the oceanic variables that most influence the spatial and temporal behavior of leatherback sea turtles (Figure 54). The model included over a dozen environmental attributes including data from Howell et al. 2015. The model (Siders et al. 2018) identified two areas where leatherback sea turtles had their highest probability of occurrence, northwest of the Hawaiian Islands (Q1-3) and just west of California (Q4). 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.

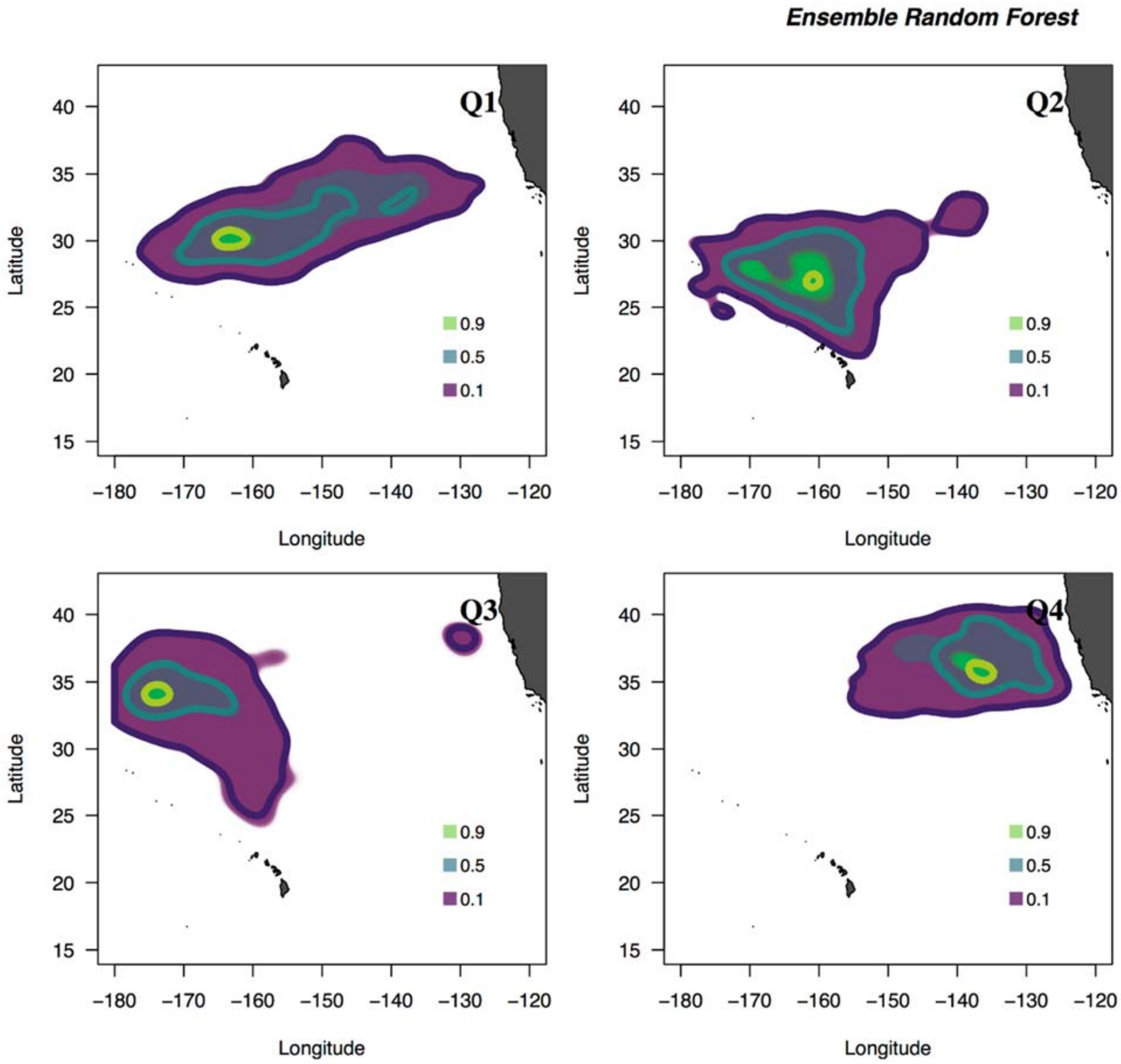


Figure 52. Areas of highest interaction between leatherback sea turtles and the Hawaiian shallow-set longline fishery by quarter. Source: Siders et al. (2018).

Table 50. Leatherback sea turtle interactions, interactions and sets by critical area, and total sets in the Hawaii shallow-set longline fishery by quarter (Siders et al. 2018; T. Jones pers. comm to A. 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 interactions using Bayesian statistical inference techniques as described in the *Approach to the Assessment* section of this biological opinion (McCracken 2018). The predictions were based on the recorded bycatch of leatherback sea turtles in the HI SSLL fishery. Table 51 contains the resulting predictions of future 1, 2, and 3-year interactions (mean and 95th percentile) for leatherback sea turtles in the HI SSLL fishery (from McCracken 2018).

Table 51. Estimated mean number of leatherback sea turtles expected to be reported as captured in fishing gear, with 95th percentile from McCracken 2018).

Period	Mean	95th 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 45). 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 have been hooked externally; however, in 2017 and 2018 two loggerhead sea turtles were retrieved dead. Table 52 lists the number of loggerhead sea turtles by injury category and release condition in the HI SSLL fishery between 2004 and 2018. Table 53 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 and amount of gear that remains is expected to affect survivability (Ryder et al. 2006). 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 52. 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

Table 53. 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 2017 and 2018, interactions with loggerhead sea turtles were the highest recorded in the HI SSLL fishery in the past 14 years. In 2017, 21 loggerhead sea turtles interactions occurred, and in 2018, 33 loggerhead sea turtle interactions occurred (see Figure 53). 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). Even though the HI SSLL fishery was only open part of the year, 2018 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 the HI SSLL fishery in any calendar year since 2004.

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 mean 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 53).

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 54).

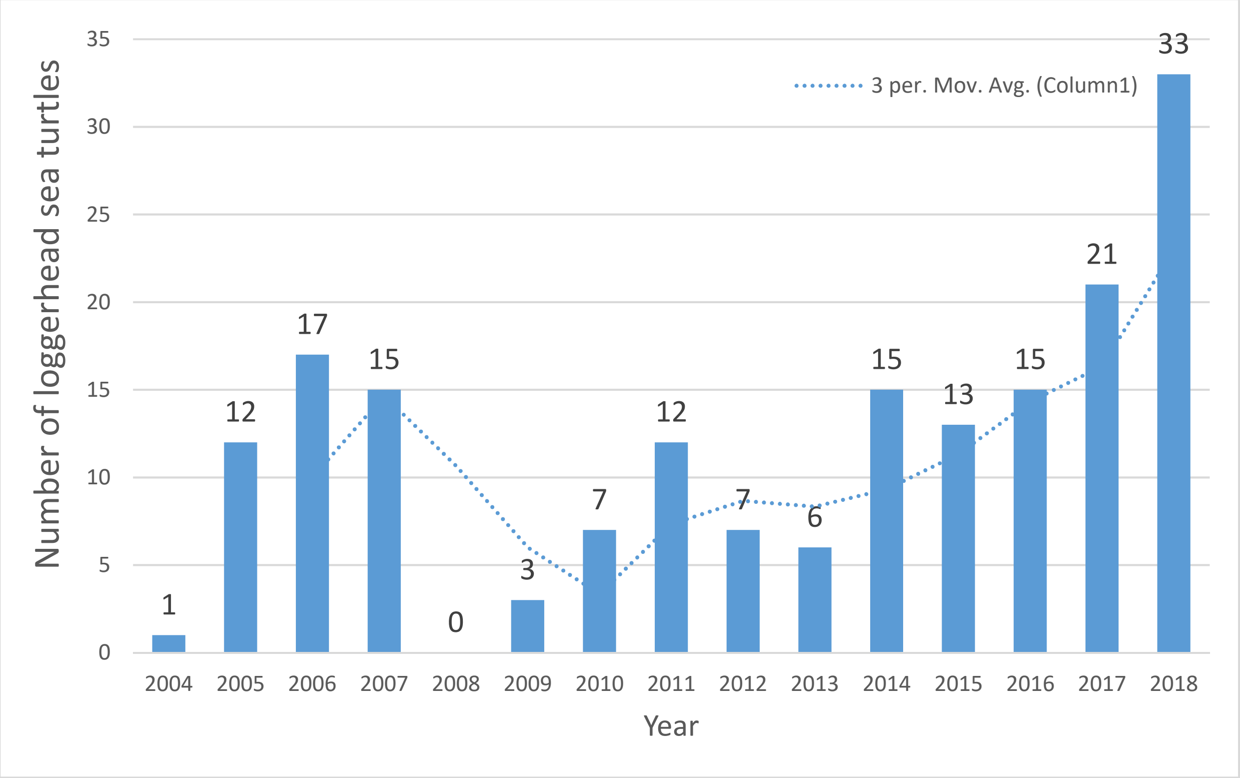


Figure 53. Loggerhead sea turtle interactions by year in the Hawaii shallow-set longline fishery between 2004 through 2018 with 3-year moving average.

Table 54. Number of loggerhead sea turtle interactions per vessel per year, 2004- 2018. 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, nine vessels were responsible for 44 interactions with loggerhead and leatherback sea turtles (Table 55). 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 collects data on interactions with 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 55. 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 with ≥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. 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 12) (P. 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 54). 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 (Polovina et al. 2000, 2001, 2004, 2006; Howell et al. 2008). 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%.

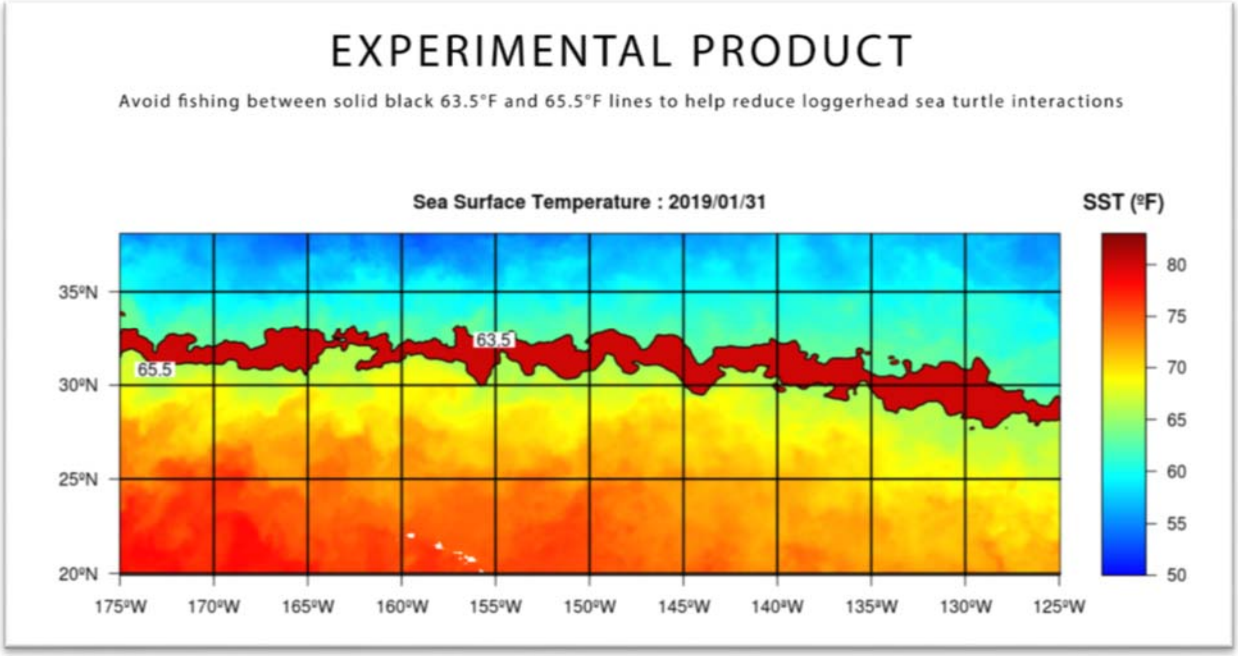


Figure 54. TurtleWatch map, January 31, 2019 (https://www.pifsc.noaa.gov/turtlewatch/today.png).

Predicted Future Exposure to the Fishery

As described previously in the leatherback sea turtle exposure section and the *Approach to the Assessment* section of this biological opinion, NMFS predicted of future interactions using

Bayesian statistical inference techniques (McCracken 2018). The predictions were based on the recorded bycatch of loggerhead sea turtles in the HI SSLL fishery. Table 56 lists NMFS' estimates of the number of loggerhead sea turtles that would be captured in 1-year, 2-year, and 3-years (mean and 95th percentile) in the HI SSLL fishery (from McCracken 2018).

Table 56. Mean and 95th percentile of loggerhead sea turtle posterior estimated exposures in terms of anticipated interactions with fishing gear (from McCracken 2018).

Period	Mean	95th 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 HI SSLL fishery re-opened in 2004, there have been a total of 10 observed interactions that have resulted in a hooking or entanglement. All 10 olive ridley sea turtles were recorded with sufficient information to evaluate post interaction mortality risk using the criteria of Ryder et al. (2006). Table 57 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 57. 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% are from the western/Indo-Pacific. Based on genetic studies, five of the 10 olive ridley sea turtles captured since 2004 are from the Eastern Pacific (P. 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 2007d, 2014). The olive ridley sea turtles that have interacted with the HI SSLL fishery have ranged in size from 48 to 64.5 cm SCL (NMFS unpublished observer data). Based on studies by Work and Balazs (2002, 2010), and Shanker et al. (2003), this suggest that olive ridley sea turtles captured in the HI SSLL fishery are immature turtles. At least one animal that was captured was likely an adult. Based on age estimation charts and size-at-age growth function in Zug et al. (2006), olive ridley sea turtles captured in the HI SSLL fishery likely range from 12 to 13 years old and could may range from subadult (60%) to 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, five individuals were captured in a small aggregation approximately 500 nm NE of Oahu in the months of March and April. Four of the five individuals captured occurred within 100 nm of each other and the fifth olive ridley was approximately 120 nm SE from those four. These five olive ridley sea turtles represent 83% (5/6*100) of the interactions documented in the months of March and April across all years for this species in this 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 for olive ridley sea turtles using the same method as described previously for leatherback sea turtles. Predictions were based on the recorded bycatch of olive ridley sea turtles in the HI SSLL fishery. Table 58 contains the resulting predictions of future annual, 2-year, and 3-year interactions (mean and 95th percentile) for olive ridley sea turtles in the HI SSLL fishery (from McCracken 2018).

Table 58. Mean and 95th percentile of olive ridley sea turtle posterior estimated exposure in terms of anticipated interactions with fishing gear (from McCracken 2018).

Period	**Mean**	**95th 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 captured 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 59 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 59. 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 HI SSLL fishery. The majority of the turtles captured are comprised of animals from the Eastern Pacific and the Central North Pacific (P. Dutton pers. comm. 17 December 2018). Estimated catch composition 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, Central West Pacific, Southwest Pacific, and the Central South Pacific. Although no animals from these four species have been captured in this fishery to date, the sample size (total number captured) is quite small and the potential exists based on the movement patterns of juvenile green sea turtles of these species that they likely occur in the action area (see Seminoff et al. 2015 for a discussion of genetics and tagging studies evaluated in determining discreteness). Therefore, we expect that they may be exposed to the HI SSLL fishery albeit in low numbers.

Table 60 lists the size of all ten green sea turtles captured in the HI SSLL fishery between 2004 and 2018. There is not enough information available to identify green sea turtles captured in this fishery to sex. Since green sea turtles do not mature in this region until reaching 80 cm (NMFS and FWS 1998b; Zug et al. 2002), we suspect that the largest green sea turtle captured may have been an adult and the remaining individual turtles were likely immatures. Two turtles, those in the 65 cm range, may have been sub adults. Table 60 lists the estimated ages 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 60. Straight carapace length measurements and estimated age for all green sea turtles captured in the Hawaii shallow-set longline fishery between 2004 and 2018 (Estimates are based on Zug and Balazs 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 do not allow for viable density mapping products to be produced. Interactions span a latitudinal distance of approximately 1,197 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 predicted future interaction levels for green sea turtles as described previously for leatherback sea turtles. The predictions were based on the recorded bycatch of green sea turtles in the HI SSLL fishery. Table 61 contains the resulting predictions of future annual, 2-year, and 3-year interactions (mean and 95th percentile) for green sea turtles in the HI SSLL fishery (from McCracken 2018).

Table 61. Mean and 95th percentile of green sea turtle posterior estimated exposure in terms of anticipated interactions with fishing gear (from McCracken 2018).

Period	**Mean**	**95th percentile**
Annual	1.4	5
2-year	2.8	8
3-year	4.1	10

6.2.1.2 Oceanic Whitetip Shark

Between 2004 and 2018, 875 oceanic whitetip sharks were caught in the HI SSLL fishery (Figure 55; NMFS Observer program unpublished data).

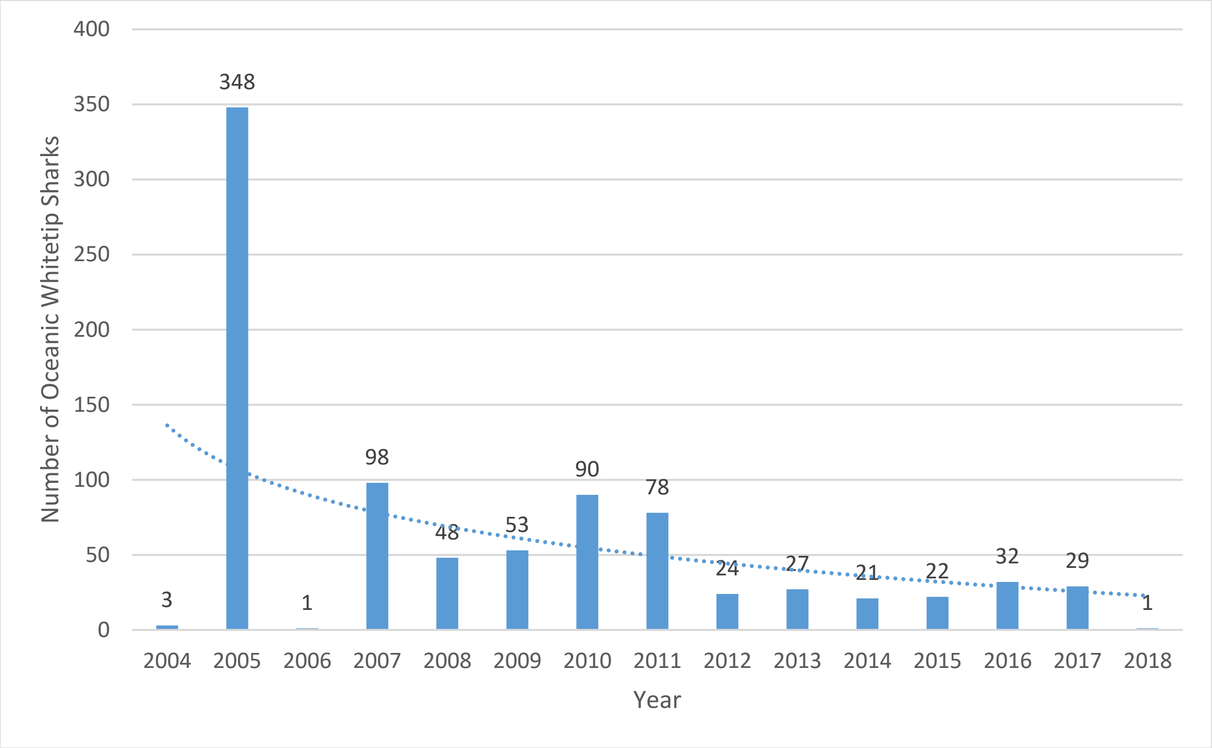


Figure 55. Observed oceanic whitetip shark interactions, and logarithmic trend, in the Hawaii shallow-set fishery, 2004-2018

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 (FAO 2012; Rice and Harley 2012; Young et al. 2017). HI SSLL fishery data, also incorporated in the Status Review (i.e. Young et al. 2017), supports these conclusions as the number of interactions have decreased substantially in our action area over time as shown in Figure 55, and occurred before the Papahānaumokuākea 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. (2017) 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 unpublished 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 62 (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. 2017).

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[6]. Two occurred before the designation in 2005, and two afterward in 2009. The Monument was expanded to include the 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 interactions 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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[6] 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 62. 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[7]	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 and condition at landing, efficiency of maintaining fishery operations, concern for the safety and stress on 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.).

When feasible, observers collect actual length measurements. Currently the protocol is to measure every third fish, regardless of species (NMFS 2017b). The actual number of oceanic

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[7] NMFS conducted species specific research in 2017 that contained atypical handling and release procedures to attach scientific instrumentation to sharks.

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 32), the data show 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 exist 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%). Sharks can be visually sexed with ease by looking at the ventral surface (i.e. underside) for claspers, which are indicative of 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 appears to interact with more females than males, at a ratio of approximately 3:2.

Spatial and Temporal Patterns of Exposure

We reviewed all data on oceanic whitetip sharks captured in the HI SSLL fishery between 2004 and 2018. Clear spatial and temporal patterns in interactions with oceanic whitetip sharks are evident both before and after the Monument declaration occurred. Most sharks (97%) were caught between April and July, with peak captures in May and June (Figure 56).

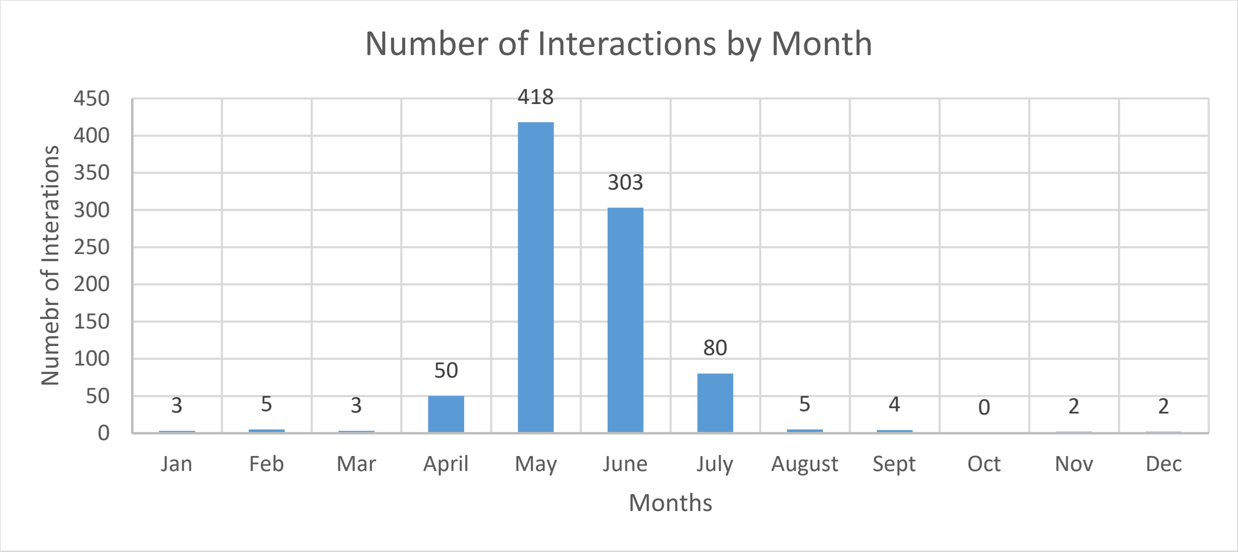

Figure 56. Total number of oceanic whitetip shark interactions by month in the Hawaii shallow-set longline fishery, 2004 through 2018 (n=875).

Using haul back locations, we mapped all sets where interactions occurred using GIS (Figure 57). 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 58), 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. This function allows for the creation of a hot spot or heat map, which illustrates the number of interactions or effort in an area corresponding to those particular values. According to ESRI, the company that created and administers the ArcGIS program, kernel density is based on the quartic kernel function described by Silverman (1986; ESRI 2018). By adjusting these ranges, the resulting maps appear smoother with edges that are more fluid. As the density increases, the color shifts from a cooler blue to a more dynamic warmer tone, also known as a color ramp. Therefore, blue areas have lower interaction or effort rates than those areas with orange or reds. The key was adjusted to illustrate low versus high density values.

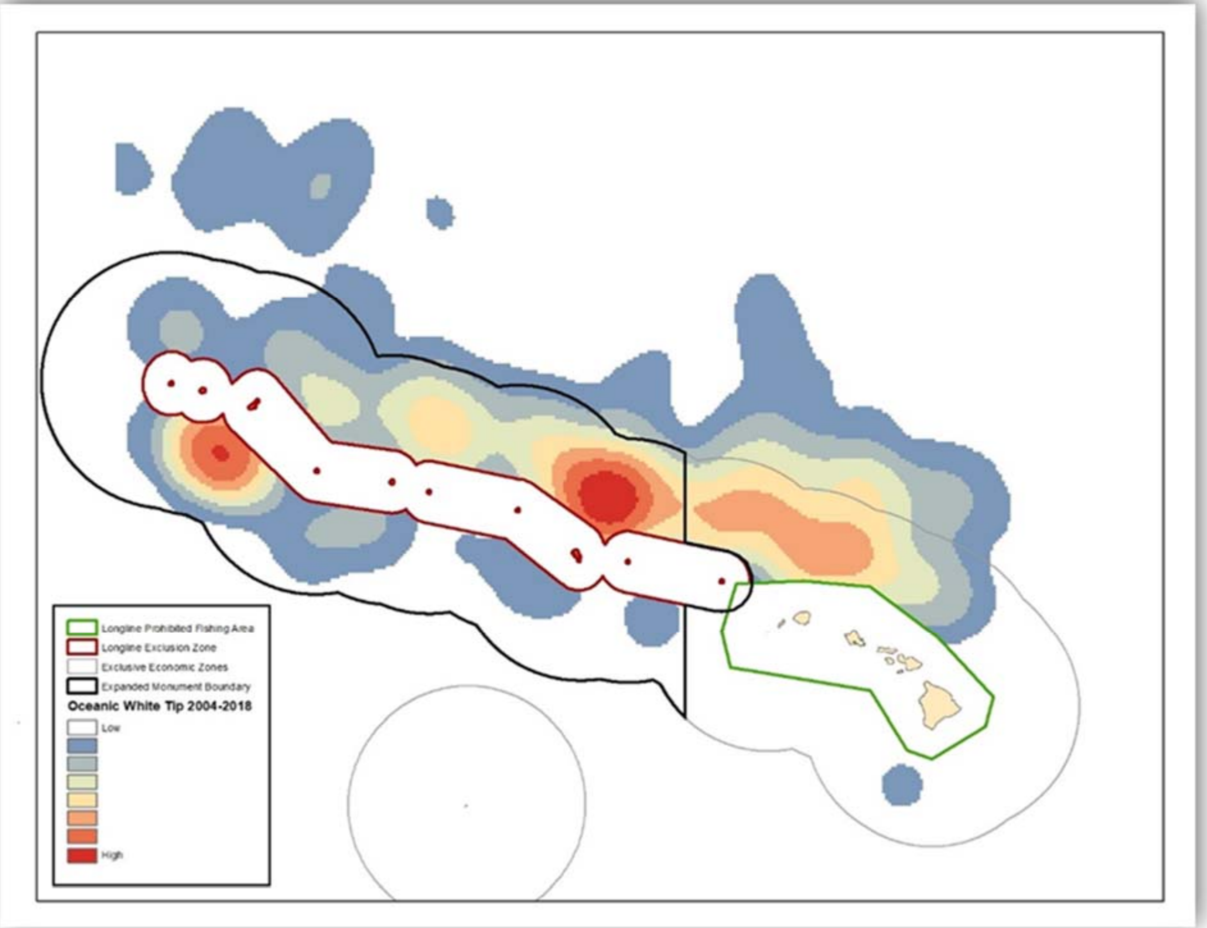

Figure 57. Geographical representation of aggregate oceanic whitetip shark interactions with the Hawaii shallow set longline fishery between 2004 and 2018.

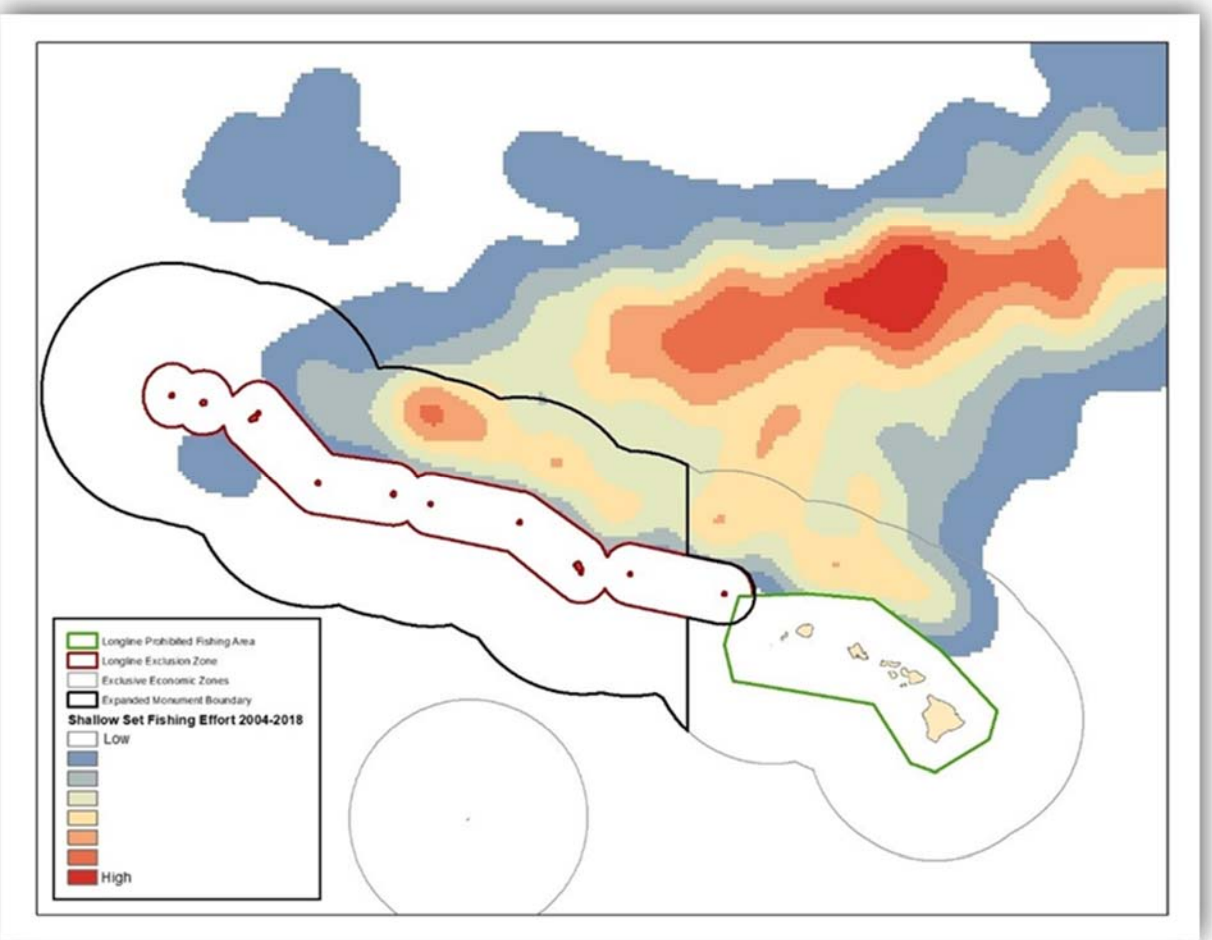

Figure 58. Snapshot of aggregated Hawaii shallow set longline fishing effort between 2004 and 2018.

As Figure 56 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 shark interactions are occurring geographically (Figure 57) when compared to aggregate fishing effort (Figure 59). 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 57, 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 60 depicts the aggregated effort with the Monument removed, as these areas can no longer be fished by the fleet.

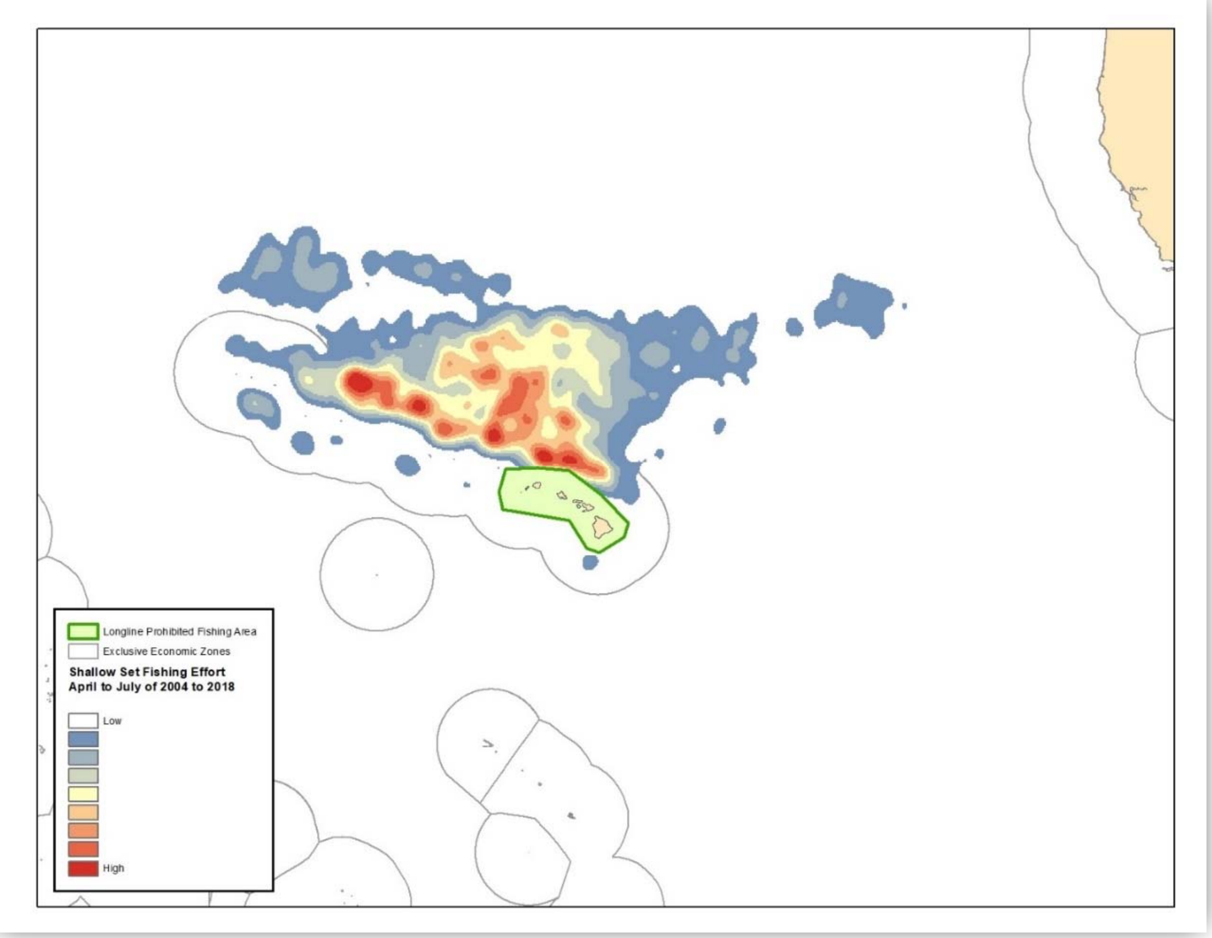

Figure 59. Aggregated fishing effort from 2004 to 2018, during only the months of April, May, June and July.

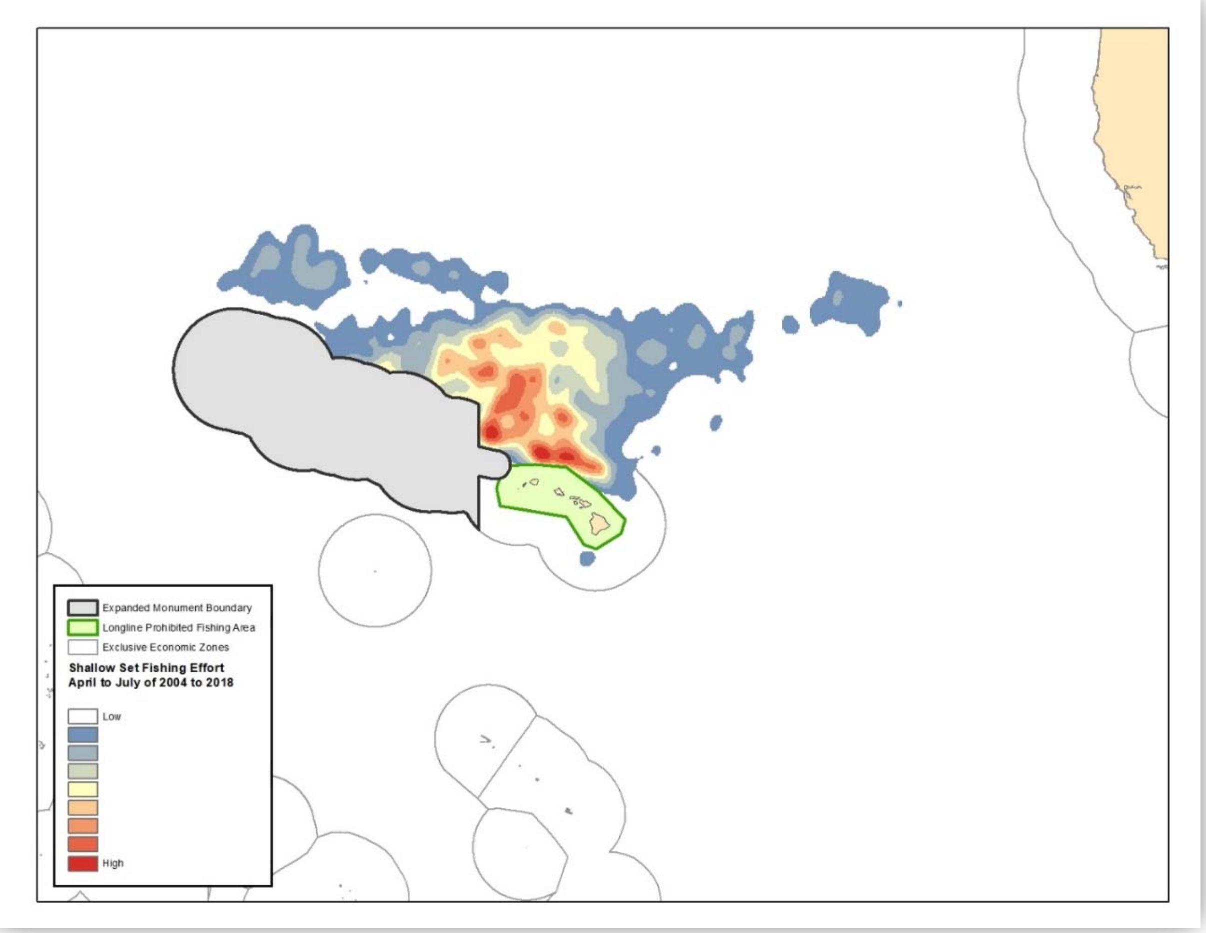

Figure 60. Aggregated fishing effort from 2004 to 2018, during the months of April, May, June and July excluding the Monument.

Between 2004 and 2018, a total of 48 vessels caught oceanic whitetip sharks in the HI SSLL fishery. Table 63 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 63). 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 63. 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 64 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 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. These three vessels range between 0 and 78.13% of the interactions any given year. However, it should be noted the lower range has a low number of interactions for that year. We draw your attention to the number and percentage of interactions for 2017 and 2018, as this fishing effort will be discussed later.

Table 64. 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
2015	2	22	9.09	0.03	0.28

Year	Top 3 Vessels: Total Number of Interactions	Total Number of Interactions by Year	Percent Interactions	95% CI: Lower	95% CI: Upper
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 61. 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).

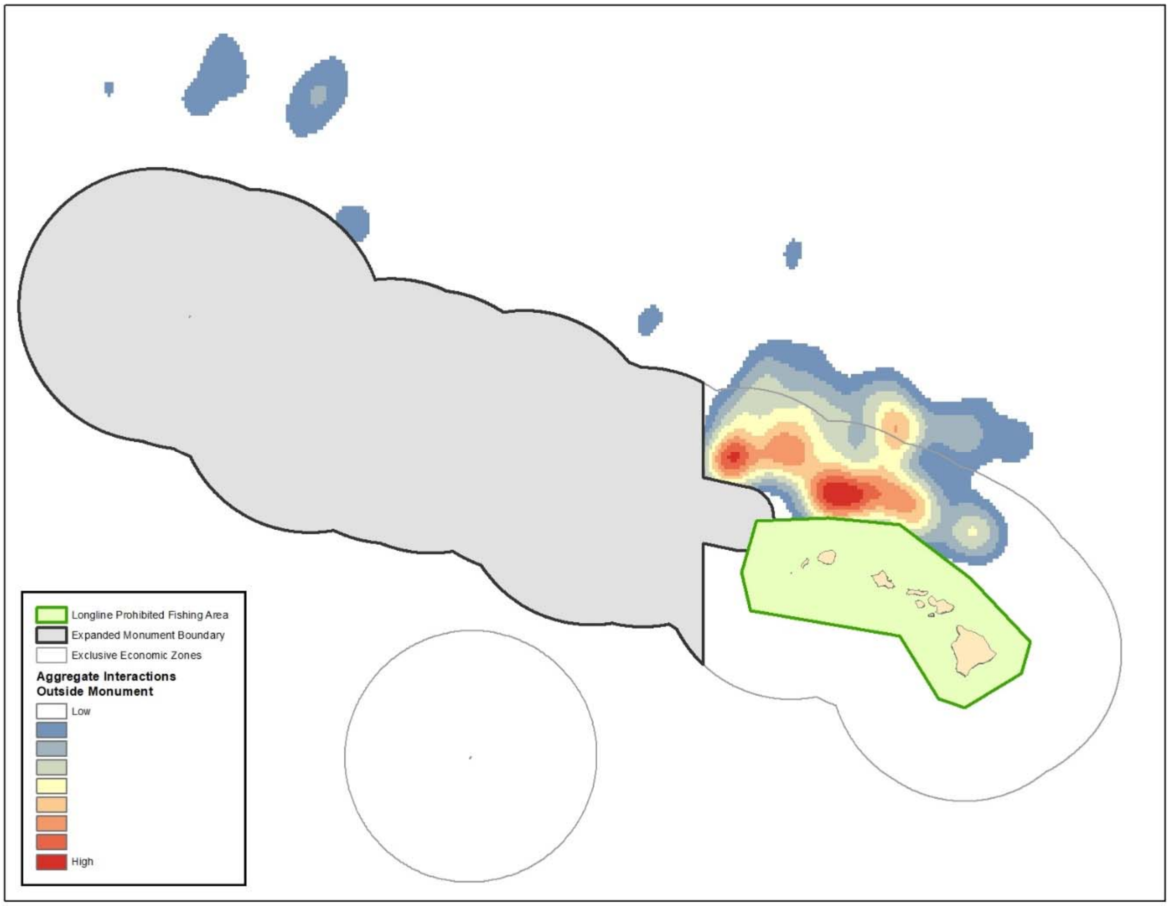

Figure 61. Aggregate oceanic whitetip shark interactions from 2004 to 2018 that occurred outside of the Monument (n=391).

Interaction data were 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 62. Spatial analysis also reveals a hotspot on the north side of the MHI (Figure 63). 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 62) when compared to aggregated effort (Figure 2). Please note, Figure 56 is the total aggregate of interactions and includes records from the post monument designation.

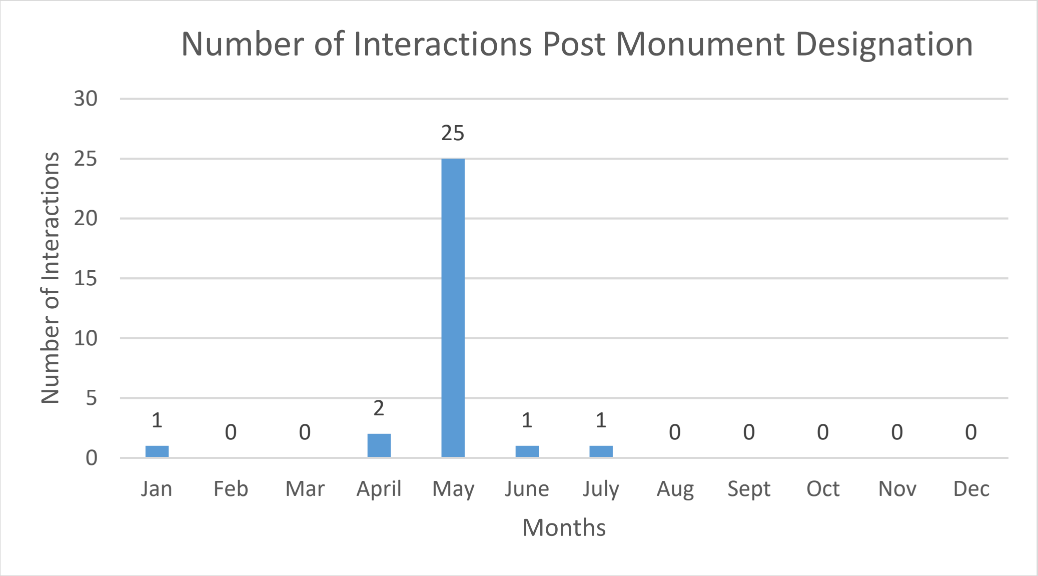

Figure 62. Number of oceanic whitetip shark interactions after the Monument designation; from August 2016 to 2018, displayed by months (n= 30).

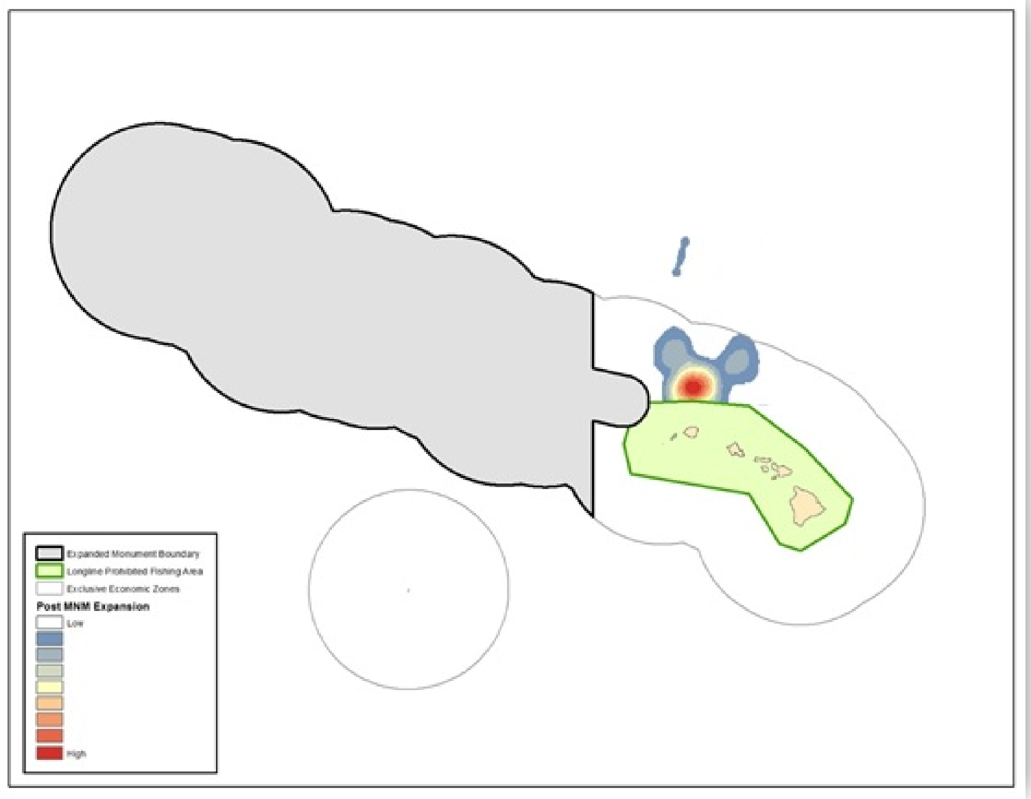

Figure 63. Interaction heat map with oceanic whitetip sharks (n=30) post Monument designation (August 2016 to May 2018).

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 64). This shows that this same area is being fished over multiple years during the same time period by multiple vessels.

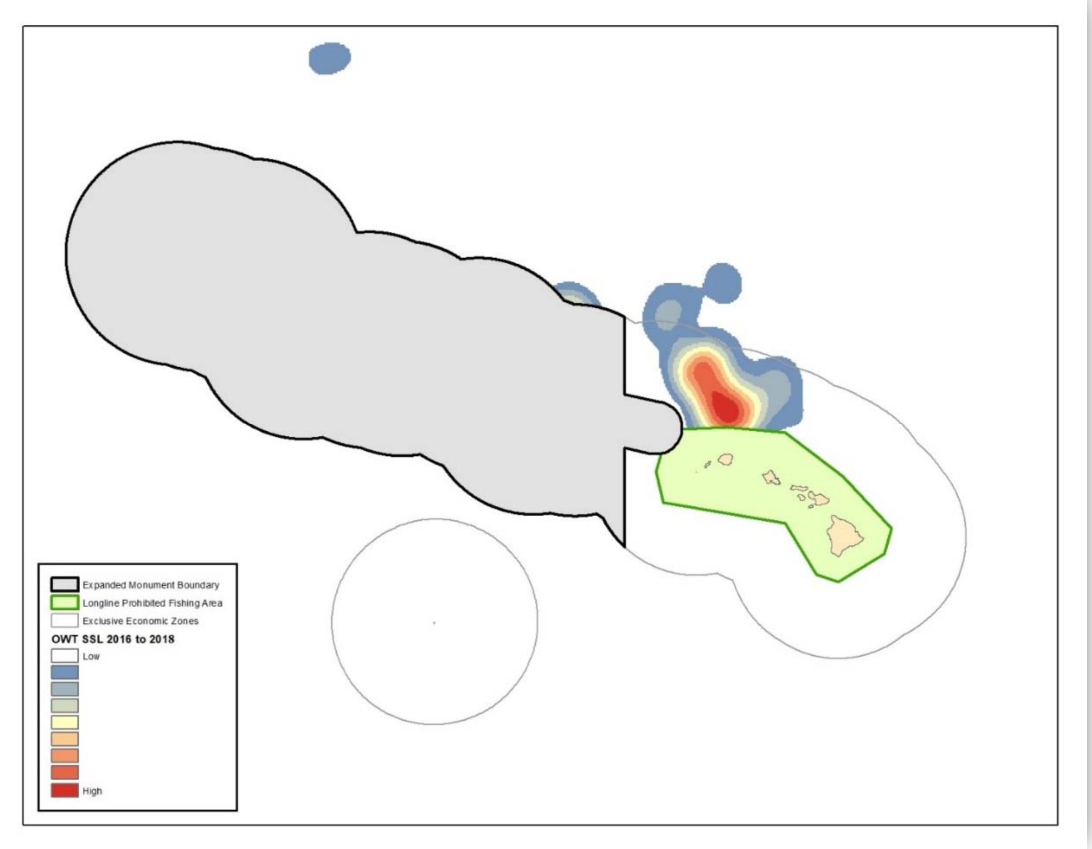

Figure 64. Post monument designation with outside interactions incorporated showing all interactions that occurred outside of the Monument from 2016 to 2018 (n=49).

We then looked at the fishing effort data for pre- and post- Monument expansion (Figure 65 and Figure 66) to compare if fishing effort was focused in this area or distributed elsewhere. Figure 65 does not depict the Monument boundaries, as these data refer 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 these 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.

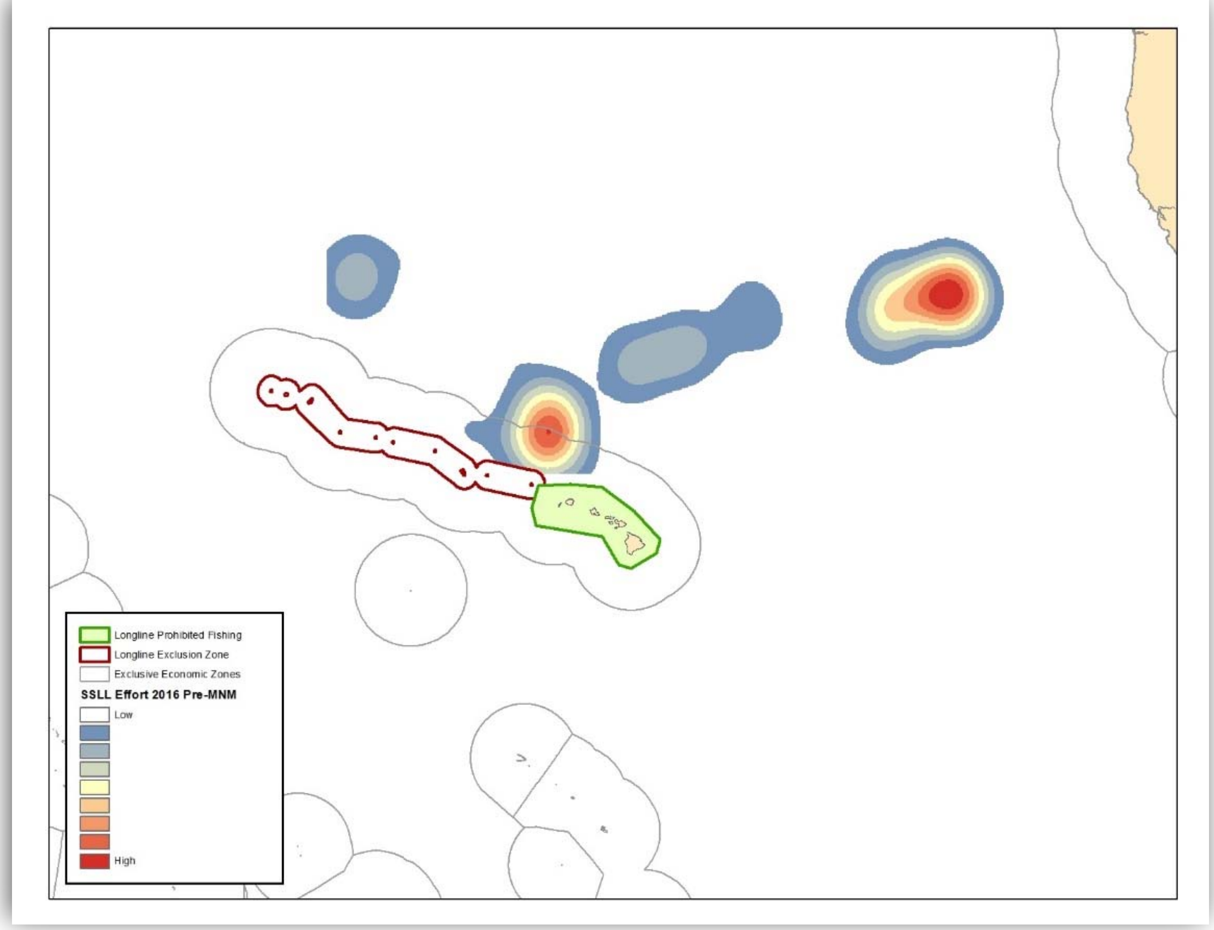

Figure 65. Aggregated fishing effort in 2016 prior to the Monument designation.

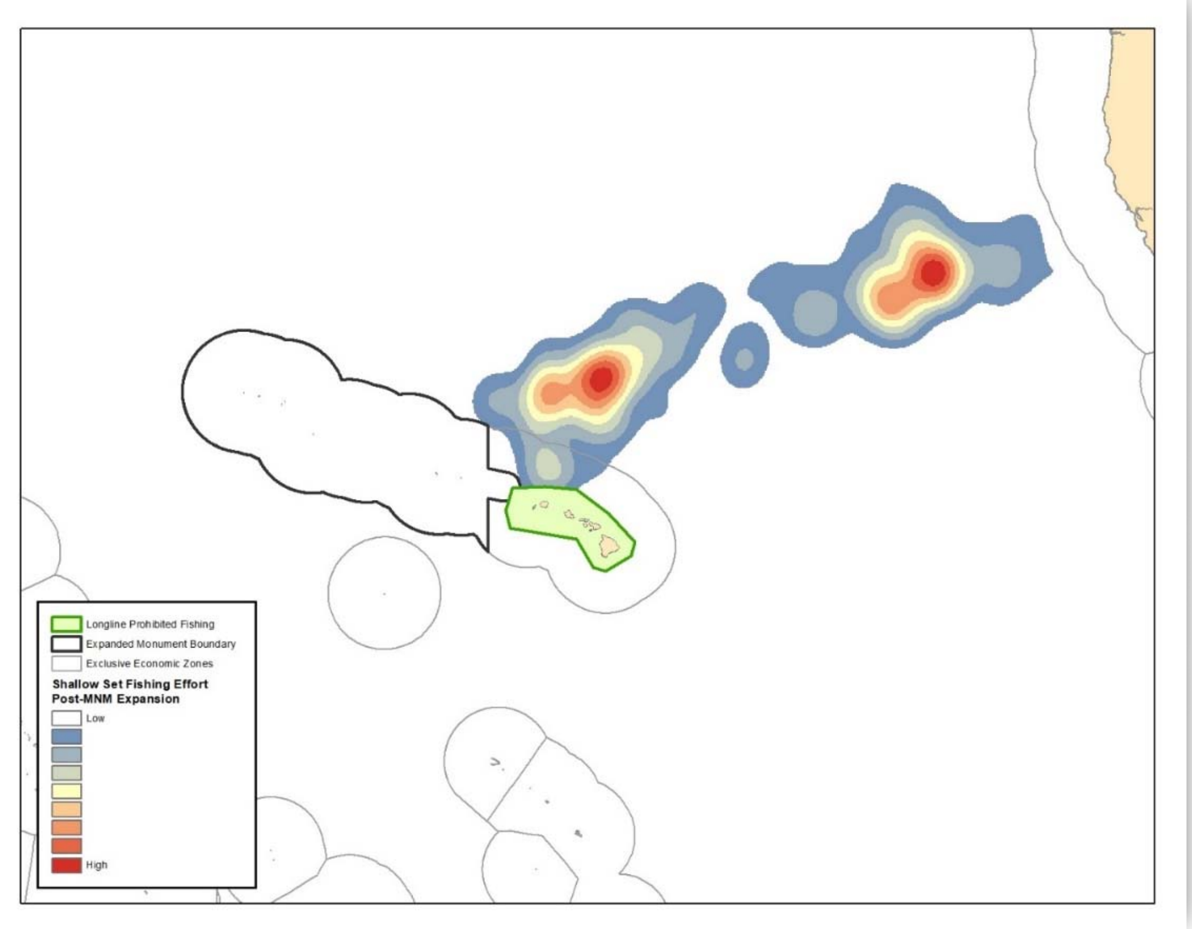

Figure 66. Aggregated fishing effort post Monument expansion; August 26, 2016 to May 2018.

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 58 and Figure 67. 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 62).

Predicted Future Exposure to the Fishery

NMFS developed predictions of future interaction levels (McCracken 2018) using Bayesian statistical inference techniques as described in the *Approach to the Assessment* section of this biological opinion. The predictions were based on the recorded bycatch of oceanic whitetip sharks in the HI SSLL fishery. Table 65 contains the resulting predictions of future annual, 2-year, and 3-year interactions (mean and 95th percentile) for oceanic whitetip sharks in the HI SSLL fishery (McCracken 2018).

Table 65. Mean and 95th 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	95th 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 66.

Table 66. Mean and 95th 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

6.2.1.3 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 67. 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 manta, they are instructed to use the unidentified ray classification.

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 68). Both methods yielded the similar results but the Wilson score method provides us with confidence intervals (Table 69) 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 69) 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) from 2004-2018.

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 67). Giant manta rays are known to aggregate around bathymetric features such as seamounts and ridges (Kashiwagi et al. 2011).

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 69). 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.

Table 67. Number of interactions and caught and release condition of rays captured in the HI SSLL fishery between 2004 and 2018. *The caught and release dispositions are for the giant manta ray, *Mobulidae* spp. and unid. ray only.

Number of interactions with ray species and their release 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

Table 68. Estimated proportions of the different species of *Mobulidae* rays captured in the shallow-set long line fishery using observer data and the 95% confidence interval for those proportions.

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 69. 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 68).

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

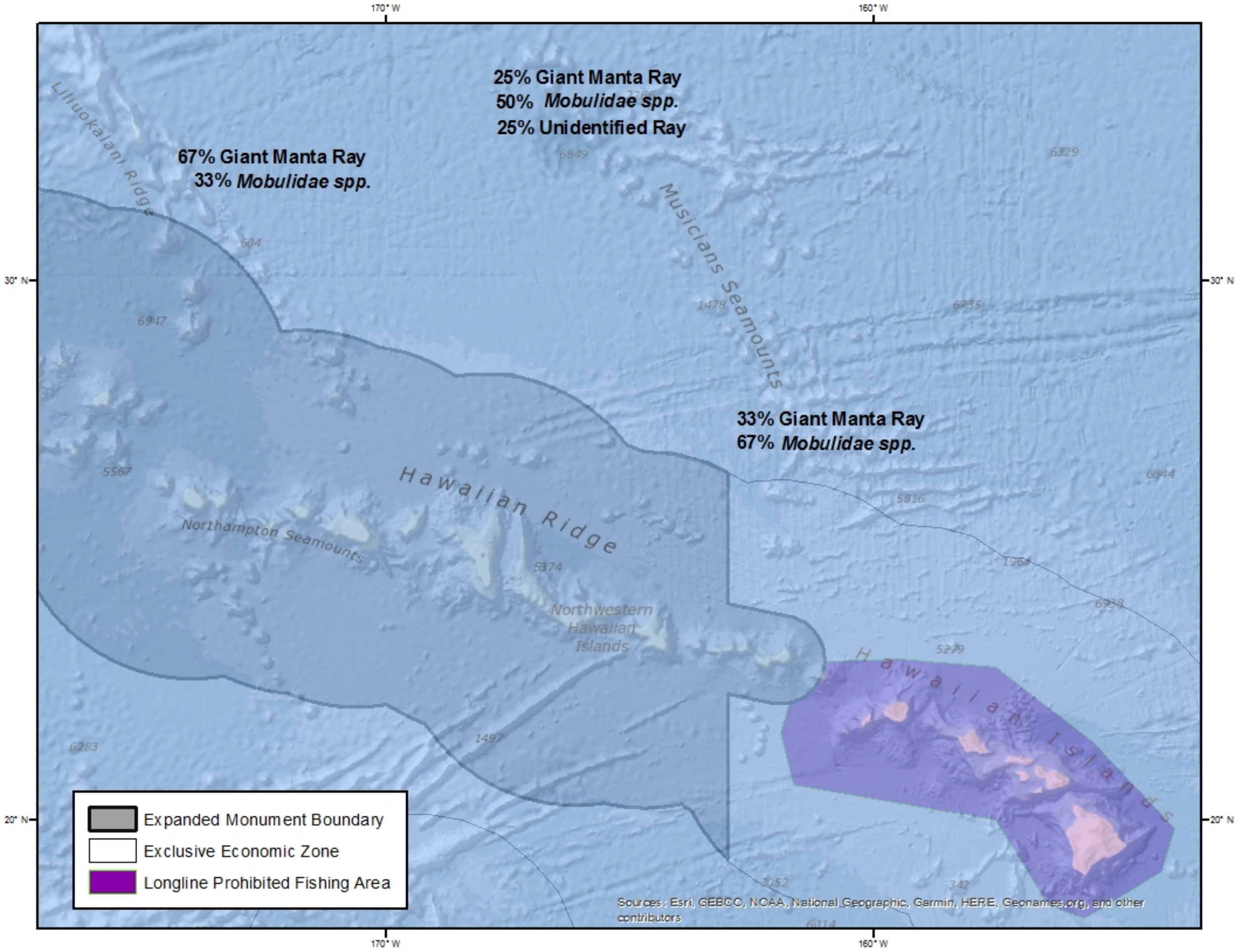

Figure 67. General location of interactions with giant manta ray, manta/mobula and unidentified ray and the percentage of each classification in each area.

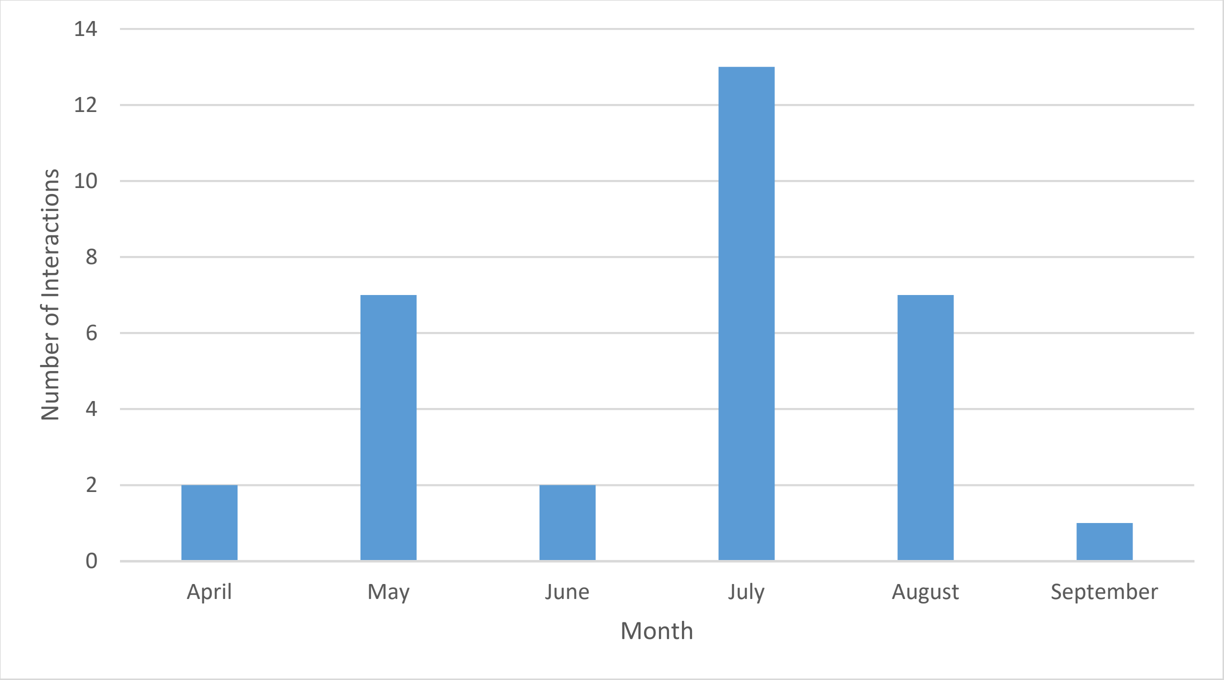

Figure 68. Giant manta ray and *Mobulidae* spp. interactions in the HI SSLL fishery by month, combined data from 2004 through 2018.

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. The predictions were based on the recorded bycatch of giant manta ray and the *Mobulidae* spp. classification in the HI SSLL fishery). Table 70 contains the resulting predictions of future annual, 2-year, and 3-year interactions (mean and 95th percentile) for giant manta ray and Table 71 contains the predictions for the *Mobulidae* spp. classification in the HI SSLL fishery (McCracken 2018).

Table 70. Mean and 95th 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 71. Mean and 95th 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. We calculated 95% confidence intervals for the proportions and assigned the 15 unidentified *Mobulidae* spp. (see Table 69) 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 72. 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 72. Anticipated future exposure levels for giant manta rays in the Hawaii shallow-set longline fishery. Values are derived by combining estimates from Table 69 and Table 70.

	Mean Estimates of giant manta rays exposures			Upper Estimates of giant manta ray exposures		
No. of Years	**Modeled (Bayes) estimate**	**Prorated estimate of unknown Mobulidae**	**Adjusted estimate**	**Upper 95% Credible Interval**	**Upper 95% CI**	**Adjusted upper estimate**
1	3	4	7	8	5	13
2	5	4	9	13	5	18
3	8	4	12	17	5	22

6.2.1.4 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 (NSI) determinations as all gear was removed from the animals and the individuals appeared healthy upon release (J. 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 (J. 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 (J. Carretta pers. comm. 2018).

Guadalupe fur seals 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 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 73 and Table 74). 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 73. Estimated proportions of the different species of *Pinnipedia* captured in the shallow-set long line fishery using observer data and 95% Confidence Intervals for those proportions. 95% Confidence Intervals for those proportions.

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 74. 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 73).

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 seal 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 these 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 to what age group these animals may have belonged. 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 may be 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 (McCracken 2018) using Bayesian statistical inference techniques as described in the *Approach to the Assessment* section of this biological opinion). The predictions were based on the recorded bycatch of Guadalupe fur seals, unidentified pinniped and *Otariidae*, 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 Guadalupe fur seals in the HI SSLL fishery (from McCracken 2018). Table 76 lists values determined for unidentified pinnipeds and Table 77 displays values for unidentified *Otariidae*.

Table 75. Mean and 95th 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	95th percentile
annual	2	6
2-year	4	9
3-year	5	12

Table 76. Mean and 95th 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 77. Mean and 95th 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 mean proportions and upper and lower 95% confidence intervals for those proportions, and assuming these proportions are representative of the unidentified animals, we assigned 4 of the 6 unidentified pinnipeds to Guadalupe fur seals (95% CI = 2, 5) . The resulting future anticipated exposure estimates are listed in Table 78. 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 78. Anticipated future exposure levels for Guadalupe fur seals in the Hawaii shallow-set longline fishery. Values are derived by combining estimates from Table 74 and Table 75.

	Mean Estimates of Guadalupe fur seal exposures			Upper Estimates of Guadalupe fur seal exposures		
No. of Years	**Modeled (Bayes) estimate**	**Prorated estimate of unknown Otariids**	**Modeled (Bayes) estimate**	**Upper 95% Credible Interval**	**Upper 95% CI**	**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.

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 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 are 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 affect size, growth rate and reproductive success. Lastly, Deakos et al. (2011) report that videos show two reef manta rays in Hawaii, which were entangled in mooring lines, perish and become immediately consumed by sharks. Although mooring lines are not used in this fishery, the material is similar to polypropylene float line.

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

If a hook does not become lodged or pierce an organ, it can pass through to the colon, or even be expelled through the turtle (E. Jacobson in Balazs et al. 1995a). In such cases, sea turtles 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 being lodged, the chances are good that less damage has been done. Tissue necrosis that may have developed around the hook may also be passed along through the turtle as a foreign body (E. Jacobson in Balazs et al. 1995a).

6.3.2.2 Oceanic Whitetip Shark

Sharks are incidentally captured when they bite 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 hook animals 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, and after their release due to exhaustion and injury. Furthermore, handling procedures can cause additional damage (e.g. cutting the jaw, tail, gaffing, etc.), stress, or death.

A gut-hooked shark is at risk of severe damage to vital organs and excessive bleeding. Campana et al. (2009) found in a post-release mortality study that 33% of tagged blue sharks with extensive trauma such as a gut-hooking perished. Campana et al. (2009) attribute rapid post-release mortality of sharks to occur 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 as cited in Miller and Klimovich 2016; 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 and 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 CMMs for the oceanic whitetip shark. Pursuant to CMM 2011-04, NMFS has implemented regulations (50 CFR 300.226) requiring vessels to release any oceanic whitetip shark that is caught as soon as possible after the shark is brought alongside the vessel, and to do so in a manner that results in as little harm to the shark as possible. In accordance 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 rays are seldom boarded, and similar to leatherback sea turtles, observers and fishers are instructed to cut the line as close the hook as possible. Occasionally, the branchline breaks during an interaction and the majority of the line may remain attached to the animal. If entangled in trailing line, the giant manta ray is at risk of severing of the cephalic and pectoral fin, which are considered severe injuries that can lead to a reduction in feeding efficiency and even death. 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 wrap around an animal, constricting its movement, cause it to become snagged or accumulate drag. Crew are directed to remove external hooks from seals. 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

We used Ryder et al. (2006) to estimate post-hooking survival and mortality of turtles captured and released alive. 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 (Swimmer and Gilman 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 79.

Table 79. 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. 2008b	65-82%		captivity	-
Hays et al. 2003	31%		PTT	-
Parker et al. 2005a	20-40%	Depends on hooks status deep vs light	PTT	-
Chaloupka et al. 2004b	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 establish a mortality estimate based on whether the turtle was lightly hooked (flipper, jaw or mouth), or deeply-hooked (ingested) and how much gear was left on the animal when it was released.

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 53). 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 95th percentile of the 1-year, 2-year, and 3-year predictions) to estimate future mortality of each species.

Table 80. 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 leatherback sea turtles captured in the HI SSLL fishery (see Table 49 and Table 50). Using the mortality coefficients of Ryder et al. (2006), of the 105 captured animals an estimated 21 leatherback sea turtles died over 14 years in the HI SSLL fishery (Table 81). The corresponding mortality rate for leatherback sea turtle interactions in this fishery is 20% (95% CI: 0.135, 0.287). The remaining 84 leatherback sea turtles that were released alive likely suffered from effects ranging from high stress immediately following post release to more severe effects such as injuries that may have impacted their feeding, migration, or even breeding if their injuries did not result in mortality.

Table 81. 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). The 95% confidence interval for the fishery mortality rate was calculated using the Wilson Score Method without continuity correction (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 51). We calculated the mortality for the mean and 95th percentile for the 1-year, 2-year, and 3-year predictions for leatherback sea turtles (Table 82). NMFS predicts that the HI SSLL fishery would capture a mean of 10 and up to 21 leatherback sea turtles each year. Our model accounts for annual variability and predicts that over a three-year time period the HI SSLL fishery would capture a mean of 30 and up to 48 leatherback sea turtles (see Table 51). As a result, the upper estimated number of leatherback sea turtles that would likely die from their capture would be up to three in any given year (mean =1), or 9 over a 3-year period (Table 82).

We would also expect sublethal responses from leatherback sea turtles that are hooked or entangled and released alive. The responses of animals may range from those that are temporary in nature such as elevated stress levels to more significant injuries that may affect feeding, movement, or even breeding success. These effects may decline over time as the surviving turtles heal from their injuries, although the extent to which such sublethal injuries occur or persist are not known.

Table 82. Anticipated mortalities of leatherback sea turtles based on predicted exposure values for the mean and 95th percentile for 1-3 years (see Table 51)

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 52). We estimated loggerhead sea turtle mortalities 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 (were recorded as dead on the vessel), for a combined total of 28 loggerhead sea turtle mortalities out of 177 (Table 83). The corresponding mortality rate for loggerhead sea turtle interactions in this fishery is 16% (95% CI: 0.11, 0.22). The remaining 149 loggerhead sea turtles that were released alive likely suffered responses ranging from high stress immediately following post release, to more severe injuries that may have impacted their feeding, migration, or even breeding success.

Table 83. 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. The 95% confidence interval for the fishery mortality rate was calculated using the Wilson Score Method without continuity correction (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 56) and added the at-vessel mortality coefficient to derive the future mortality estimates. Table 84 provides the mortality estimates for loggerhead sea turtles for the mean and 95th percentile for 1, 2, and 3-years. NMFS predicts that the HI SSLL fishery would capture a mean of 16 and up to 36 loggerhead sea turtles each year. Our model accounts for annual variability and consequently predicts that the HI SSLL fishery would capture a mean of 47 and up to 81 loggerhead sea turtles over a three-year time period (see Table 56). As a result, the number of loggerhead sea turtles that would die from their capture would be up to 6 per year (mean = 2), or up to 13 over a 3-year period.

We would also expect sublethal responses from loggerhead sea turtles that are hooked or entangled and released alive. The responses of animals may range from those that are temporary in nature such as elevated stress levels to more significant injuries that may affect feeding, movement, or even breeding success. These effects may decline over time as the surviving turtles heal from their injuries, although the extent to which such sublethal injuries occur or persist are not known.

Table 84. Anticipated mortalities of loggerhead sea turtles based on predicted exposure values for the mean and 95th percentile for 1-3 years plus the coefficient for at-vessel mortality.

Period of ATL	Mortality estimate for the mean	Upper mortality estimate
annual	2	6
2	5	10
3	8	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 57). We calculated olive ridley sea turtle mortality using the injury mortality coefficients of Ryder et al. (2006). Between 2004 and 2018, there were likely no olive ridley sea turtles that died from injuries sustained during these 10 interactions; while our sample size is notably low, all animals have been released with all gear removed increasing their chances of survival (Table 85). The confidence intervals for a mortality rate of zero olive ridley sea turtles in this fishery is 95% CI: 0.0, 0.28. Because we rounded each interaction category to the nearest whole turtle, and used 0.5 (or half) as our determining value for rounding our estimate, this resulted in a prediction of zero mortalities. Because our sample size is small, there is inherent uncertainty in our mortality coefficients, and had we summed the values prior to rounding or chosen a different critical value to round to whole numbers, with providence we expect that there may have been 1 latent mortality in the ten interactions in 14 years. We expect that the remaining 9 olive ridley sea turtles released alive likely suffered from effects ranging from high stress immediately following post release to more severe injuries that may affect feeding, movement, or even breeding success. These effects may decline over time as the surviving turtles heal from their injuries, although the extent to which such sublethal injuries occur or persist are not known.

Importantly, during the course of this consultation, in 2019, for the first time in 15 years the HI SSLL fishery recorded an at vessel mortality for an olive ridley sea turtle. We added this mortality into our evaluation and calculated a coefficient for at vessel mortality similar to that of the loggerhead sea turtles.

Table 85. 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). The 95% confidence interval for the fishery mortality rate was calculated using the Wilson Score Method without continuity correction (Newcombe 1998).

*There may have been 1 latent mortality in the ten interactions in 14 years (see text for more explanation) and in 2019 there was 1 at vessel mortality.

	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	-	-
Insertion Visible	0	0	0	0
Jaw	0	-	0	0
Grand Total	0	0	0-1*	0
At vessel mortality				1
Grand Total				2
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 and added the at-vessel mortality coefficient to derive the future mortality estimates. Table 86 provides the mortality estimates for olive ridley sea turtles for the mean and 95th percentile for 1, 2, and 3-years. NMFS predicts that the HI SSLL fishery would capture a mean of 1 and up to four olive ridley sea turtles each year (see Table 58). Our model accounts for annual variability and consequently predicts that the HI SSLL fishery would capture a mean of 4 and up to 11 olive ridley sea turtles over a three-year time period (see Table 58). As a result, the upper estimated number of olive ridley sea turtles that would likely die from their capture would be 1 per year, or up to two over a three-year period (Table 86). 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.

We would expect most of the olive ridley sea turtles released alive to experience some sublethal effects from being captured in the fishery. The responses of animals may range from those that are temporary in nature such as elevated stress levels to more significant injuries that may affect feeding, movement, or even breeding success. These effects may decline over time as the surviving turtles heal from their injuries, although the extent to which such sublethal injuries occur or persist are not known.

Table 86. Anticipated mortalities of olive ridley sea turtles based on predicted exposure values for the mean and 95th percentile for 1-3 years.

Period of ATL	Mortality estimate for the mean	Upper mortality estimate
annual	0	1
2	1	2
3	1	2

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 59). We calculated green sea turtle mortality using the injury mortality coefficients of Ryder et al. (2006). The resulting estimate suggests that between 2004 and 2018, there were likely no green sea turtles that died from injuries sustained during these 10 interactions. While our sample size is notably low because all animals were released with all gear removed, we would expect this to increase their chances of survival (Table 87). Because we rounded each interaction category to the nearest whole turtle, and used 0.5 (or half) as our determining value for rounding our estimate, this resulted in a prediction of zero mortalities. Because our sample size is small, there is inherent uncertainty in our mortality coefficients, and had we summed the values prior to rounding or chosen a different critical value to round to whole numbers, with providence we expect that there may have been 1 latent mortality in the ten interactions in 14 years. The confidence intervals for a mortality rate of zero green sea turtles in this fishery is 95% CI: 0.0, 0.28.

We would also expect sublethal responses from the 9 green sea turtles that are hooked or entangled and released alive. The responses of animals may range from those that are temporary in nature such as elevated stress levels to more significant injuries that may affect feeding, movement, or even breeding success. These effects may decline over time as the surviving turtles heal from their injuries, although the extent to which such sublethal injuries occur or persist are not known.

Table 87. 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). The 95% confidence interval for the fishery mortality rate was calculated using the Wilson Score Method without continuity correction (Newcombe 1998).

	Number of green sea turtle mortalities			
Injury Category	**Line < 1/2 SCL**	**Line ≥ 1/2 SCL**	**No Gear**	**Grand Total**
Entangled	-	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	0-1*	0-1*
Fishery mortality rate (95% CI)			0.0 (0.0, 0.28)	

*There may have been 1 latent mortality in the ten interactions in 14 years (see text for more explanation).

We applied the same injury category mortality coefficients to the anticipated exposure numbers for green sea turtles to derive the future mortality estimates. Table 88 provides the mortality estimates for green sea turtles for the mean and 95th percentile for 1, 2, and 3-years. NMFS predicts that the HI SSLL fishery would capture a mean of 1 and up to 5 green sea turtles each year (see Table 61). Our model accounts for annual variability and consequently predicts that the HI SSLL fishery would capture a mean of 4 and up to 10 green sea turtles over a three-year time period (see Table 61). As a result, the number of green sea turtles that would likely die from their capture would be up to 1 over a 3-year period (Table 88). 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 mortality prediction. Our sample size of green sea turtles is very small and small changes in the number and type of injuries (i.e., hook location and depth) could significantly affect the predicted number of mortalities.

We would also expect sublethal responses from green sea turtles that are hooked or entangled and released alive. The responses of animals may range from those that are temporary in nature such as elevated stress levels to more significant injuries that may affect feeding, movement, or even breeding success. These effects may decline over time as the surviving turtles heal from their injuries, although the extent to which such sublethal injuries occur or persist are not known.

Table 88. Anticipated mortalities of green sea turtles based on predicted exposure values for the mean and 95th percentile for 1-3 years.

Period of ATL	Mean Mortality estimate	Upper for mortality estimate
Annual	0	0
2	0	1
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 62). 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 2012), and the recent status review conducted in 2016 (Young et al. 2017). 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% (Moyes et al. 2006; Campana et al. 2009, 2016; Musyl et al. 2011), resulting in overall mortality estimates between 23 and 59% for those particular species (i.e. shortfin mako, blue shark, porbeagle; Musyl et al. (2011) also assessed oceanic whitetip, bigeye thresher, and silky sharks but did not determine post release mortality values for these species).

Post release survival rates, however, are species-specific with much of the research focusing on underlying physiology of how sharks handle stress. Because these are studies of sharks in other locations of the world, and some species exhibit physiologic differences that affect their survivability these data should be interpreted cautiously. Even so, they are a reasonable (best available) surrogate from published studies and fisheries that operate similarly to HI SSLL fishery. 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 2012; WCPFC 2018a). This stock assessment may not be a good surrogate or comparison 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 affects mortality risk. Sharks that swallow hooks, compared to those that are hooked in the jaw, are more likely to die from their hooking (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 (6 and 19%) resulting in an overall mortality estimate of 18 to 31%. We then applied these rates to the number of oceanic white tip sharks that we expect will be captured by the HI SSLL fishery (Table 65) to estimate future mortalities (Table 89).

Harley et al. (2015) conducted Monte Carlo simulation modelling to predict impacts to oceanic whitetip sharks in the WCPFC operational area. Predicted mortality estimates ranged from approximately 10 to 40% under various management scenarios (see Harley et al. 2015). We compared Harley et al.'s (2015) model results to provide context to our estimated mortality metrics within this fishery. Although the study by Harley et al. (2015) examined fisheries that were significantly different than the HI SSLL fishery, which included different operational areas and gear configurations, had low levels of observer coverage from flag states, and close to zero observer coverage for several of the key flag states within the Pacific (i.e. China, Chinese Taipei, Korea, and Japan; Harley et al. 2015) we found that the metrics produced by our analysis of estimated mortality within the HI SSLL fishery were similar to those produced by Harley et al.'s (2015) model.

Table 89. Estimated oceanic whitetip shark mortalities based on predicted exposure values for the mean and the 95th percentile (see also Table 65).

Number of years	Mean	Middle range mortality estimate	Upper exposure estimate	Upper range mortality estimate
1	32	6 to 10	102	19 to 32
2	65	12 to 20	166	30 to 51
3	97	18 to 30	299	54 to 92

NMFS predicts that based on a mean of 32 interactions 6 to 10 individual oceanic whitetip sharks may die from their capture. Using the upper estimated of predicted interactions of 102, NMFS predicts that up to 32 oceanic whitetip sharks could die in any year. Over a three-year time period the HI SSLL fishery would capture a mean of 97 and an upper estimate of 299 oceanic whitetip sharks, which leads to an estimated mortality of up to 92 animals during the same time frame.

We would expect animals that are released alive and survive may exhibit sublethal responses from their capture. The responses of oceanic whitetip sharks that survive may range from those that are temporary such as elevated stress levels, to injuries that are more significant injuries such as those that may affect feeding, movement, or even breeding success. These effects may decline over time if the surviving sharks heal from their injuries, although the extent to which such sublethal injuries occur or persist are not known.

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 southwestern 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 U.S. pelagic longline fishery were released alive.

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 often 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, which sometimes cause 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; WCPFC 2017b).

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, observers do not record the hooking location when giant manta rays are captured in the HI SSLL fishery. 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 one of the largest reptiles in the world. Further, both animals propel through the ocean environment using similar movements; the leatherback sea turtle employs its long front flippers while the giant manta ray relies on its large pectoral fins as wings to "fly" through the water. 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 proxy for determining post-hooking survival of giant manta rays (see Table 53).

Since giant manta rays 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 72). 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 rays over 14 years in the HI SSLL fishery (Table 90).

Table 90. 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). The 95% confidence interval for the fishery mortality rate was calculated using the Wilson Score Method without continuity correction (Newcombe 1998).

Injury Category	**Line < 1/2 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 the adjusted mean and upper future capture estimates for giant manta rays (see Table 72). NMFS predicts that the HI SSLL fishery would capture a mean of 7 and up to 13 giant manta rays each year. NMFS predicts that the HI SSLL fishery would capture a mean of 12 and up to 22 giant manta rays over a three-year time period (see Table 72). As a result, the number of giant manta rays that would likely die from their capture would be up to 4 in any year (mean n=2) or up to 9 over a 3-year period.

Table 91. Anticipated mortalities of giant manta rays based on predicted exposure values for the mean and adjusted upper estimate for 1-3 years (see Table 72).

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.; J. Carretta pers. comm. 2018). Those with a SI determination 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 takes 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.

To estimate the total number of Guadalupe fur seal mortalities, we used the Wilson score method for the appropriation of unidentified animals to the listed species to estimate the number of listed fur seals that would be expected to interact with the HI SSLL 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 92).

Table 92. 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 for both the mean and upper exposure estimates.

Number of years	Adjusted Exposure estimate (mean)	Mortality estimate for the mean	Adjusted Upper Exposure Estimate	Upper Mortality Estimate
1	6	5	11	9
2	8	6	14	12
3	10	7	17	14

Thus, mortality ranges exist for the projected number of takes for up to the projected 3 years. For instance for one year, the mean is projected at 6 interactions (exposures), whereas the upper mortality estimate is 11 Guadalupe fur seal interactions. Applying the 80% overall mortality rate results in 9 individuals for the upper mortality estimate and 5 for the mean. 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 using the upper mortality estimate.

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 turtles, leatherback sea turtles, green sea turtles, or olive ridley sea turtles considered herein. The basic premise of the argument is that the HI SSLL fishery benefits sea turtles by its existence because this domestic fishery interacts with and kills fewer turtles than non-domestic fisheries that would otherwise fish in its place. The BE suggests that the number of turtles captured and killed—globally or, at least, Pacific wide—in commercial fisheries would be higher but for the continued operation of the HI SSLL fishery at current effort because: (1) reduced landings in the HI SSLL fishery—resulting from closing or reducing fishing effort—would create market demands ("market transfer" or "spillover effect") that would be satisfied by non-domestic fisheries; (2) to meet the consumer demand in the United States that would no longer be filled by the domestic fisheries, specifically the HI SSLL fishery, non-domestic fisheries would increase their fishing effort; (3) increasing effort in non-domestic fisheries would increase the number of interactions between those fisheries and threatened and endangered sea turtles; and (4) more of those sea turtles would be killed because non-domestic fleets do not abide by the same turtle-friendly protective measures as the HI SSLL fishery.

The proposition seemed plausible so we evaluated whether the evidence of this "transfer effect" was strong enough to treat it as an "indirect effect" of the HI SSLL fishery. The section 7 regulations define indirect effects as effects that are "*caused by the proposed action* and are later in time, but still are *reasonably certain to occur* (50 CFR 402.02).” We examined the available evidence to determine if changes in the HI SSLL fishery were reasonably certain to cause an increase in the number of threatened and endangered sea turtles captured and killed in non-domestic fisheries. We also examined the best available scientific and commercial data to assess the potential magnitude or significance of these transfer effects, if they occurred, 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 (Sarmiento 2006; Rausser et al. 2009; Bartram 2010; Chan and Pan 2012, 2016; Komoroske and Lewison 2015; Mukherjee 2015; Helvey et al. 2017; Scorse et al. 2017; Rausser and Kovach unpublished 2018). However, our review did not lead us to conclude that a market effect is caused by the HI SSLL fishery and is reasonably certain to occur. Most studies have focused on market transfer effects that might have occurred during the 2001–2004 closure of the HI SSLL fishery, which does not necessarily establish that such an effect occurs today.

Fifteen years have passed since the 2000-2004 closure and no additional empirical evidence is available to evaluate the strength of the models and correlations drawn by several authors about the closure period, nor are new data evaluated to ascertain whether spillover effects are occurring today. The papers we found through our literature review considered the potential effect of a market transfer effect resulting from the 2000 – 2004 closure of the HI SSLL fishery. During the course of our review, we asked “is the global market for swordfish such that reducing effort in the domestic longline fishery would be offset by increased effort in non-domestic fisheries for swordfish?” Perhaps. Rausser et al. (2009) and Chan and Pan (2012, 2016) only establishes that such a relationship is feasible. It does not establish that a transfer effect occurred during the 2001-2004 closure. Because Rausser et al (2009) and Chan and Pan (2012, 2016) based their analyses on a suite of assumptions that may or may not be valid, their conclusions represent the “world as it may have existed” if their assumptions were correct. We do not know how a change in one or more of their assumptions would have changed their results.

Several of these studies relied on econometric and other models to evaluate the market transfer effect in the HI SSLL fishery; while these models might establish that transfer effects were plausible, they do not present any empirical evidence to establish that there are or were indirect effects caused by the HI SSLL fishery to sea turtles. Most available studies relied on changes in import data to conclude that a market transfer effect co-occurred with the 2001–2004 closure of the HI SSLL fishery. However, those studies did not establish that market changes caused changes to turtle interactions or mortality.

Given management and market changes that have occurred since 2004, the arguments the papers advance may no longer be true. Since 2008, the WCPFC (2008b, 2018b) has adopted CMMs (CMM 2008-03, 2018-04) that require, among other measures, that members require longline vessels fishing for swordfish in the shallow set manner to employ at least one of the following measures: large circle hooks, use only whole finfish bait, or use additional measures to conserve sea turtles (see FAO 2010 for additional description of the 2008 measure). These recent improvements in non-domestic 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. However, data on swordfish shallow-set fishing by foreign countries are extremely limited (ABNJ 2017) and consequently it is difficult to make any meaningful comparisons as to the interaction rates of foreign fisheries with sea turtles. Low observer coverage on foreign longline fleets, coupled with the different operational characteristics and spatio-temporal aspects of the foreign fisheries makes the analysis particularly challenging. Swordfish shallow-set effort in the WCPO represents only 1% of the longline effort (ABNJ 2017).

The Handbook (1998) notes that indirect effects be reasonably certain to occur, "as evidenced by appropriations, work plans, permits issued, or budgeting; they follow a pattern of activity undertaken by the agency in the action area; or they are a logical extension of the proposed action." Indirect effects are more uncertain than direct effects, are more numerous than direct effects, often have weak causal linkages, and delayed effects. The greater the relative distance from an event to the endpoint the greater the opportunity that the causes are mediated by factors not in the direct causal path (Efroymson et al. 2016; EPA 2019). Because causality is difficult to demonstrate, we examined the causal pathway and the strength of evidence necessary to conclude that indirect changes in sea turtle interactions and mortality would be caused by the proposed action and would be reasonably certain to occur.

Figure 69 depicts the causal argument as a simple chain diagram. Box A represents the HI SSLL fishery, which was closed for about three years (2000-2004). A review of the literature suggests that non-domestic swordfish imports to the United States supplanted domestic swordfish supplies during this period, as illustrated by Box B. This is the strongest indirect effect argument that can be made about the fishery because the best data available comes from measurable changes in the contribution of swordfish from suppliers (domestic and non-domestic sources) before, during and for some time after the closure. However, this step in the causal chain is not sufficient to suggest there was an indirect effect on sea turtles without further expansion of the causal pathway.

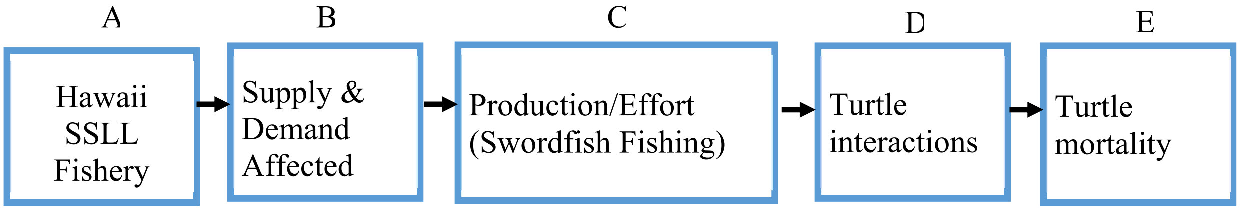

Figure 69. Simple conceptual model of the indirect causal pathway from swordfish catch to sea turtle mortalities.

As depicted in Figure 69 the causal chain is lengthy, with several steps between the source, the HI SSLL fishery, and the endpoint, sea turtle mortality. Each subsequent step is inherently weaker than the former and requires additional evidence to demonstrate that the source is causes the endpoint, and is reasonably certain to occur.

What must follow next to support the causal chain for spillover effects is that swordfish fishing must have increased in response to the closure (depicted in our conceptual diagram as the move from supply and demand in Box B to a corresponding increase in foreign fisheries as a result). Here the relationship is more tenuous because many more variables could influence how and why foreign sources could fill the gap in the domestic market. As noted earlier, the best evidence we found in the studies we examined in our systematic review was provided in economic models, not empirical data. Importantly, because we do not know the suite of assumptions used by that authors to build their models, we cannot critically evaluate the strength, relevance or credibility of those assumptions or how a change in one or more of their assumptions would have changed their results.

Box C, changes in non-domestic fishery effort and catch, presupposes that we can determine what drives changes in non-domestic fisheries. This step in the causal chain may be among the most difficult to demonstrate because it gets at motivation and production, and is going to be affected by many factors. Because causal relationships are inherently difficult to demonstrate analytical frameworks for causal analyses tend to focus attention on eliminating other candidate causes, which may be particularly important to evaluating long causal chains and ascertaining what is reasonably certain to occur. This is because confidence in a causal argument increases when many types of evidence consistently support or weaken it (EPA 2019). After Box B the asserted consequences seem plausible but we could not find evidence to conclude that they necessarily caused the next and as such would be considered *reasonably certain* to occur. Some of the papers in our review, discussed other candidate causes for market shifts during the 2000-2004 closure period but none were particularly comprehensive or contained weight of evidence approaches to eliminating candidate causes.

From Box C, we move to Box D. This move along the causal pathway requires the assumption of the (a) number of sea turtle interactions (b) with a particular turtle species that would have occurred, had the fishery *not* been closed and the (c) number of interactions with a particular species (e) with a particular non-domestic fishery to derive an estimate of the number of turtles, and of which species that would have been interacted with. This means assumption must be built on top of assumption as to exposure of the species, the number of individuals of each, and their likely responses because both species and fishery would affect the outcome of their exposure. We do not have evidence to begin to answer most of these questions. The final box in our causal pathway, Box E, requires that we make a risk statement based on the previous set of assumptions for which we lack current and reliable data. Finally, we acknowledge that whatever assumptions we make (or made) about the 2000-2004 closure and what occurred in the swordfish market 15 years ago, may not be valid today. Even if the "but for" test of causality was met by evidence presented by the Sarmiento (2006), Rausser et al. (2009), and Chan and Pan (2012, 2016), 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 that the mere existence of the HI SSLL fishery benefits sea turtles, either generally or the specific sea turtle species that are adversely affected by the HI SSLL fishery. The causal chain from source, HI SSLL fishery, to endpoint, sea turtle catch and mortality, is extremely complex and evidence to understand if it is indeed causal is weak at best, and importantly it does not appear to be reasonably certain to occur.

Based on our analyses, the evidence available do not suggest that the continued operation of the HI SSLL fishery is reasonably certain to cause a change in the number of sea turtles captured and killed in foreign fisheries. As a result, we do not treat the number of sea turtles captured and killed in foreign longline fleets as an "indirect effect" of the proposed action. Instead, we evaluate the effects of other fisheries, including non-domestic/foreign fisheries in the action area, on threatened and endangered species in the *Environmental Baseline* of this biological opinion. Specifically, we treat foreign fisheries that occur in the Action Area for the HI SSLL fishery 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. NMFS searched for information on future State, tribal, local, or private actions that were reasonably certain to occur in the action area. Most of the action area is outside of territorial waters of the United States of America, which would preclude the possibility of future state, tribal, or local action that would not require some form of federal funding or authorization. NMFS conducted electronic searches of business journals, trade journals, and newspapers using Google, *WorldCat*, and other electronic search engines. Those searches produced no evidence of future private action in the action area that would not require federal authorization or funding and is reasonably certain to occur. As a result, NMFS is not aware of any actions of this kind that are likely to occur in the action area during the foreseeable future.

7 INTEGRATION AND SYNTHESIS OF EFFECTS

The *Status of the Listed Resources*, *Environmental Baseline*, and *Cumulative Effects* described the pre-existing condition of the listed species globally and within the action area given the effects of activities such as commercial fisheries, direct harvests and modification or degradation of habitat caused by marine debris and climate change. The pre-existing condition of these species serves as the point of reference for our conclusions. The *Effects of the Action* section of this biological opinion describes the 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.

This section of this biological opinion recapitulates, integrates, and synthesizes the information that has been presented thus far to evaluate the risks that continuing the HI SSLL fishery poses endangered leatherback sea turtles, endangered North Pacific loggerhead sea turtles, threatened Eastern Pacific green sea turtles, threatened Central North Pacific green sea turtles, threatened East Indian-West Pacific green sea turtles, endangered Central West Pacific green sea turtles, threatened Southwest Pacific green sea turtles, endangered Central South Pacific green sea turtles, endangered Mexico breeding population of olive ridley sea turtles and threatened (other) populations of olive ridley sea turtles, threatened oceanic whitetip sharks, threatened giant manta rays and threatened Guadalupe fur seals in the Pacific Ocean. The "risks" this section of the opinion considers are (1) increases in the extinction probability of particular populations and of the species as they have been listed; and (2) reductions in their probability of being conserved (that is, of reaching the point where they no longer warrant the protections of the ESA). These two probabilities correspond to the species' likelihood of surviving in the wild (that is, avoiding extinction) and their likelihood of recovering in the wild (that is, being conserved). Our analyses give equal consideration to both probabilities; however, to satisfy the explicit purposes of the ESA and NMFS' obligation to use its programs to further those purposes (16 US.C. 1536(a)(1)), a species' probability of being conserved has greater influence on our conclusions and jeopardy determinations. As part of these analyses, we consider the action's effects on the reproduction, numbers, and distribution of each species.

In the *Approach to the Assessment* section of this biological opinion, we stated that focus our assessment on three variables in the jeopardy definition that determine a species likelihood of survival and recovery in the wild: reductions in the species' reproduction, numbers or distribution. We measure risks to individuals of endangered or threatened species using changes in the individuals' "fitness" or the individuals' growth, survival, annual reproductive success, and lifetime reproductive success. When we do not expect listed individuals exposed to an action's effects to experience reductions in fitness, we would not expect the action to have adverse consequences on the viability of the populations that those individuals represent or the species those populations comprise (Stearns 1977, 1992; Mills and Beatty 1979; Anderson 2000). As a result, if we conclude that listed animals are *not* likely to experience reductions in their fitness, we would conclude our assessment. If we conclude that one or more population is likely to experience an increase in its' extinction probability (measured using probability of demographic, ecological, or genetic extinction in 10, 25, 50, or 100 years) we assess the effect of that increase on the probable reproduction, numbers, or distribution of the species. For these analyses, we combine our knowledge of the patterns that accompanied the decline, collapse, or extinction of populations and species that have experienced these phenomena in the past supplemented by indicators of probable demographic performance produced by population viability analyses.

We recognize that an appreciable reduction in a species' likelihood of surviving and recovering in the wild is not necessarily a quantitative calculation. Rather, since we rarely have information on the abundance and trend, growth rate, spatial structure or other indicators of population and species diversity to place numeric values on a species chances of persistence we use a combination of quantitative and qualitative analytical methods for our risk evaluation. It was our intent to build an analytical framework that appropriately considers the information available to each step of this analysis and uses the best method to evaluate and incorporate that information into our assessment. At the same time it was important to appropriately insure that the federal action is not likely to jeopardize the continued existence of the endangered and threatened species, and to do so in a way that minimizes uncertainty about the potential impacts listed species can be expected to experience. Underestimating the number of listed individuals that might be exposed to the adverse effects of an action and the number of listed species expected to be "taken" as a result of that exposure poses the greatest risk for endangered and threatened species. The explicit mandate of section 7(a)(2) of the ESA and congressional direction to provide the benefit of doubt to endangered and threatened species should make the choice of predictive metrics relatively simple: select the metric and interval that provides the greatest confidence of being correct and the lowest risk of being incorrect. The latter would (1) help federal agencies insure that their actions are not likely to jeopardize the continued existence of listed species (for example, by minimizing the chances of underestimating the number of listed species adversely affected by an action) and (2) provide endangered and threatened species with the benefit of the doubt in the face of uncertainty (by minimizing the probability of reaching an incorrect conclusion). This approach recognizes the inherent asymmetrical risk that may be borne by threatened and endangered species because extinction is irreversible and makes appropriate use of many types of best available scientific and commercial data, and is not limited or constrained by numeric information or models. We use such tools where they can best support the information and the analysis that are available, and use other qualitative methods like systematic reviews to compile and evaluate qualitative information into our assessment. As such, this assessment better incorporates available information that is otherwise poorly studied or integrated by quantitative methods.

In this assessment, we examined several scenarios to estimate the risk the proposed action poses listed species. We considered that threatened and endangered species may be exposed to the proposed action for at least another 25 years, as well as the effect of the action on future population sizes of 50%, 25% and 12.5% the current estimate. This means we examined the effect of the action based on a future period that varies by species. For most species, this means we examined a future scenario that is well beyond 25 years in the future, based on our best estimates. For instance, for leatherback sea turtles we predict that the West Pacific Ocean population could reach 25% of its current size in about 37 years (range 7 to 91 years), whereas North Pacific loggerhead sea turtles could reach 25% of its current size in about 56 years (range 3 to 99 years) based on our population viability assessments for these two species. During this consultation, we were reminded that our jeopardy analyses needed to address the concept of "tipping points." The concept of a tipping point and the need for the Services to consider it has arisen in a series of courts' opinions (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, emphasis added). The biological opinion implicated in that case argued (in part) that a jeopardy determination would only be appropriate when an action would both: (1) reduce appreciably the likelihood of survival; and (2) reduce appreciably the likelihood of recovery. The 9th Circuit rejected this argument because "a species can often cling to survival even when recovery is far out of reach."

We understand the idea the Court articulated and the Court's concern. However, it is technically impossible to know, in advance, where such a "tipping point" might lie for free-ranging plants and animals (and even animals in captivity). We know that when a species has been reduced to a single individual, that effectively precludes its recovery (Scheffer 2010). At the same time, several populations and species have recovered after having been reduced to a populations consisting of handfuls of individuals while retaining the ability to recover. The New Zealand black robin (*Petroica traversi*) had been reduced to seven individuals, but has since increased to a population that has been estimated to number in the hundreds. California condors, Whooping cranes, Vancouver Island marmot, and Cayman Island blue iguanas had all declined to populations in the low 20s and Black-footed ferret had declined to a population of 18, but those small population sizes did not preclude their recovery.

At the same time, we are also aware of the long list of species that have become extinct after their populations experienced declines similar to those listed in the preceding paragraph. The extinction of the species *Achatinella apexfulva* this January was a reminder of the number of species that have become extinct in the Hawaiian Islands. Several investigators have studied demographic tipping points beyond which imperiled populations cannot be expected to recover, (these are formally called "transcritical" or "catastrophic" bifurcations; see Fagan and Holmes 2005; Drake and Griffen 2010; Scheffer 2010). Laboratory experiments on zooplankton (*Daphnia spp.*) suggest it might be possible to identify early warning signs of an impending bifurcation (Drake and Griffen 2010), but those experiments are still works in progress. Right now, we cannot identify the point below which recovery would be precluded for a particular population or species.

Although the information available does not allow us to identify specific "tipping points" for the species this biological opinion considers, the discipline of conservation biology provides a few rules of thumb. First, when a species consists of multiple populations, the loss of one or more of those populations places the species at greater risk of extinction and lowers its likelihood of recovery. Therefore, our analyses first consider an action's effects on the viability and resilience of the populations the action affects. Second, when populations decline to about 50 individuals (1) it is not likely to increase in abundance before it becomes extinct; (2) any increases in abundance will be very short-lived; and (3) the persistence of the population depends on the reproductive success of a small proportion of individuals (Fagan and Holmes 2001). That same study suggests that these three phenomena can affect populations numbering 300 individuals. Third, our analyses carefully consider the spatial structure of populations, patterns of immigration and emigration that connect populations, and the existence of population refugia because these will determine a species' resilience (Thomas and Jones 1993; Thomas and Kunin 1999; O'Grady et al. 2004). Finally, we rely on our knowledge of species that have declined, collapsed, and are now extinct and well as those that avoided extinction and those that recovered from endangerment. That knowledge allows us to consider general patterns that have resulted in extinction, avoided it, or that have allowed species to recover from endangerment.

Our analyses find that the proposed action—while it results in the death of individual endangered leatherback sea turtles, endangered North Pacific loggerhead sea turtles, threatened Eastern Pacific green sea turtles, threatened Central North Pacific green sea turtles, threatened East Indian-West Pacific green sea turtles, endangered Central West Pacific green sea turtles, threatened Southwest Pacific green sea turtles, endangered Central South Pacific green sea turtles, endangered Mexico breeding population of olive ridley sea turtles, and threatened (other) populations of olive ridley sea turtles, threatened oceanic whitetip sharks, threatened giant manta rays and threatened Guadalupe fur seals—it has very small effects on the dynamics of the populations those individuals represent or the species those populations comprise. As a result, we believe it does not appreciably reduce these species' likelihood of survival and recovery in the wild.

We explain the basis for this conclusion for each species in the following sections. These 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 93 for a snapshot of this history as it relates to leatherback sea turtles. In 2001, NMFS issued a jeopardy biological opinion on the operation of this fishery (at the time, NMFS consultation was on the mixed fisheries) for leatherback sea turtles, and a few other species. In 2001, the fishery was estimated to interact with 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. With the reopening of the fishery in 2004 the fishery changed its operational practices, which led to a significant reduction in interactions for all sea turtles. In 2004, that reduction led NMFS to conclude that the action was not likely to jeopardize the leatherback sea turtle. At the time NMFS expected that the fishery would interact annually 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. Three 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.

In the past fourteen years (2004 through 2018) since the substantial operational changes occurred in the HI SSLL fishery, this fishery has interacted with 105 adult and subadult leatherback sea turtles. As discussed in earlier sections of this opinion, these 105 interactions have 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. Accordingly this suggests that over the past 24 years, the HI SSLL fishery has interacted with more than 650 leatherback sea turtles, most of which have been adult and subadult animals.

Because operational changes in the fishery in 2004 significantly changed the past interaction rate for this species and other turtles, we only rely upon data from the past 14 years to estimate the effect of continuing the proposed action into the future. Even so, not all 14 years represent complete years due to years of fishery closures. In closure years, it is possible that the fishery may have interacted with higher numbers of turtles if there was no hard cap, and we considered this in our analyses (this is described in more detail in McCracken 2018). While we have a complete census for the 14 years of the fishing effort and interactions with leatherbacks, we highlight that there is still a great deal of uncertainty in the predictions we rely on in this opinion for several reasons. One of the main reasons for this uncertainty is that interactions with leatherbacks are rare events, and 14 years of data is a relatively short period when analyzing rare events. In addition, there is also uncertainty surrounding the post-hooking mortality rate, which is used to estimate the future number of anticipated mortalities.

Table 93. 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

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. (2006), 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. As discussed in the *Approach to the Assessment*, we purposefully used the 95th percentile (the upper estimate) from our analyses of the number predicted interactions in a 1, 2, and 3-year period. 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 (i.e. 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 population is doing particularly poorly, and our PVA is built on incomplete an uncertain data, we weighed the effect of 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 conservative 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 also important to maintaining diversity and resiliency of the population and the species. When a species exhibits multiple life history strategies it is important that NMFS consider the effect of an action on that life history strategy, particularly as is the case for this action, which is expected to disproportionately affect one particular life history type. This is because life history strategies are often indicative of demographic diversity, and can have impacts on a species' chances of survival and recovery. Diversity protects species against short-term spatial and temporal changes in the environment and can provide the raw genetic material for surviving long-term ecological changes. "Any actions that affect basic demographic and evolutionary processes... have the potential to alter a species' diversity" (McElhaney et al. 2000). 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.

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 1,851 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. 2013b, 2013c). 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 3 years (mean 26 years, range 3-86 years,). PVA modeled estimates suggest the modeled population presently consists of about 1,180 adult female leatherback 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 similar to that of the IUCN (1,851 compared to about 1,770 mature leatherback sea turtles—our number is adjusted to account for the fact the PVA estimate in the status section of this opinion represents about 85% of the nesting females and the population has a 3:1 sex ratio females to males).

Based on Jones et al. (2012) there were about 2,600 (1,800-3,400) mature leatherback sea turtles in the West Pacific Ocean population in 2004. Current estimates suggest the adult portion of the population is about 1,851 (1,488-2,320; (based on Jones et al. 2018). 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 adjusted as noted above results in an estimated adult population size of 1,488 (1,116 females and 372 males). As noted in the *Status of Listed Resources,* this suggests that the total West Pacific Ocean population is comprised of about 175,000 individual leatherback sea turtles, ranging between 68,000 and 360,000.

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 U.S. 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 U.S. 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 of extinction based NMFS' PVA analyses by Jones et al. (Jones 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). Based on Tapilatu et al. (2013) we would expect about 60% of the data to represent summer nesters.

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 each year, and this includes the HI SSLL fishery (Peatman et al. 2018b; BDEP data tables). If this is correct, then some leatherback sea turtles are likely exposed to more than one fishery interaction each year. 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.

Climate change may be affecting the species already but will likely increase in the future. Global average sea levels are expected to continue to rise by at least several inches in the next 15 years, and by one to four feet by 2100 (Wuebbles et al. 2017). This could affect migrations and feeding patterns by changing ocean circulation. Increased sand temperatures can also cause decrease egg survival and an increase in female turtles skewing sex ratios and affecting the reproductive capacity of the population.

Leatherback sea turtles represent 35% of the turtles captured in the HI SSLL fishery, most of which are subadults or adults. A higher percentage of leatherback sea turtles are released with trailing gear than other turtle species, 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 eight leatherback sea turtles, which means the HI SSLL fishery exceeded eight interactions half of the time with the highest number of interactions on record as 16 leatherback sea turtles. In 2011, the HI SSLL fishery reached 16 leatherback sea turtle interactions and the fishery closed, so we do not know if more interactions would have occurred that year had the fishery not closed. In 2014, when the HI SSLL fishery again had 16 interactions with leatherback sea turtles, the fishery did not close because the cap for annual interactions was higher. Our model takes into account the partial year, 2011 when the fishery closed because it met its leatherback sea turtle interaction cap; the results is a future upper estimate of up to 21 leatherback sea turtles that may be hooked and entangled in the HI SSLL fishery in any given year. Although the mean (10) yields a better prediction of the cumulative effect of this fishery over many years, the upper estimate (21) is a better estimate for the number of interactions that could happen in a single year. The hooking or entanglement of up to 21 (mean=10) leatherback sea turtles per year is expected to result in the mortality of up to 3 leatherback sea turtles in any given year from their exposure as described in detail in the *Exposure and Response* section of this biological opinion.

Assuming the West Pacific Ocean population includes 68,000 leatherback sea turtles of all age classes, which represents our low estimate for the population, if the HI SSLL fishery interacted with up to 21 leatherback sea turtles in a year, this would represent 0.03% of the total leatherback sea turtle population. In year 1, a 5% reduction of the total population of 68,000 population represents about 3,000 leatherback sea turtle that would be expected to die from the combination of natural sources of mortality and the cumulative effect of exposure to stressors on their natal beaches and along migratory routes to and from feeding areas and nesting areas. In a year of high mortalities (upper estimate 3), the contribution of the HI SSLL fishery to this total mortality is only 0.08%.

Assuming there are currently 1,488 adult leatherback sea turtles in the West Pacific Ocean population, that the HI SSLL fishery could be expected to interact with up to 21 leatherback sea turtles in any given year, and we would expect those interactions would be with adults and sub-adult leatherback sea turtles. The HI SSLL fishery may interact with 1.4% of the adult population in any given year. This rate of interaction, however would not be expected in every year. As indicated by our modeled estimate, predicted interaction numbers decline with the longer period considered in our model runs as depicted in Table 51. In other words, across 3 years we would expect that the HI SSLL fishery would interact with up to 48 individual leatherback sea turtles in a 3-year period. The best estimate from our modeled interactions is 10 and 30 for 1 and 3 years respectively, as noted in Table 51. While the best estimate is the mean over longer time periods, which is 10, we used the 95th percentile for our evaluation in large part because in any given year we would expect that the number of interactions between the HI SSLL fishery and leatherbacks would not likely exceed 21 but would exceed 10 in some years.

The corresponding mortality estimate with an interaction of up to 21 leatherback sea turtles is three. The loss of three individual leatherback sea turtles from an adult population comprised of 1,488 animals represents 0.20% of the population, and the loss of three individuals from a total population of 68,000 represents 0.004% of the total population. If we assume that all three are represent animals of the summer life strategy, which appears to be a reasonable assumption, and that 60% of the total adult animals are summer animals then the loss of three leatherback sea turtles from the summer life history component would represent about 0.3% of adults of the summer life history strategy. While our assessment estimates that up to three leatherback sea turtles could die from interactions with the HI SSLL fishery, up to 18 would be released with varying degrees of injuries and some with trailing gear. The exact number of animals affected by non-lethal injuries and their fate is not known; however, we acknowledge that other responses can occur from these interactions, and can range from short term behavioral changes to longer lasting effects. These types of effects are not easy to monitor. We anticipate that most of the turtles that do not die as a result of their injuries may experience differing levels of harassment or other sublethal effects from the exposure to hooking and entanglement, which range from being temporary in nature such as elevated stress levels to more significant injuries that may heal over time but may affect the individual's lifetime fitness. Of the 18 animals that live through their capture or entanglement, we expect that a small portion of the turtles that survive may experience fitness level impacts.

As noted in the *Approach to the Assessment* we are concerned about the viability of the species, and the populations that comprise the species. In the case of the West Pacific 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).

Our best estimate is that the HI SSLL fishery will hook and entangle *up to* 21 leatherback sea turtles each year (mean = 10), and from these interactions *up to* 3 will die. Assuming this fishery continues for at least another 25 years, based on the mean estimated number of interactions we would expect that the HI SSLL fishery would hook or entangle about 250 more leatherback sea turtles and about 25 of these animals would die from their interactions over a 25-year time frame. The upper estimate of 21 leatherback sea turtles would produce *up to* 525 interactions in 25 years, with *up to* 75 mortalities across those 25 years.

Based on the PVA for leatherback sea turtles, as described in the *Status of Listed Resources* section of this biological opinion, we would expect that the West Pacific Ocean population of leatherback sea turtles would reach 50% of its current abundance in about 26-28 years, and 25% of its current abundance in about 31-37 years (see Table 8 and Table 9). If the total West Pacific Ocean leatherback sea turtle population is reduced to 50%, 25% and 12.5% of its current estimate of 68,000, and the fishery interactions remain the same, in each scenario the fishery would interact with less than 0.2% of the total population. Total mortality would be less than 0.03% of the total population when the population reached 12.5% of its current size. Adjusting for summer nesters, and looking at a future population of about 12.5% of its current size, we still conclude that effect is inconsequential—the fishery would interact with less than 0.5% of total summer nester population, and mortalities would represent up to 0.066% of the summer nester population). Our analysis examines the effect of the HI SSLL fishery on leatherback sea turtles, under several lines of reasoning. Under each scenario examined, we conclude that the impact of the HI SSLL fishery on leatherback sea turtles is sufficiently small that we would not expect it to appreciably reduce the species' chances of survival and recovery in the wild. Not one scenario even reaches a measurable effect of 1% relative to the metric measured, and our estimates are all less than 0.5% even when the upper estimates of mortality are weighed against adjusted estimates of adults of the summer life history strategy only.

Baseline conditions that continue to act on the species, like the cumulative effect of foreign fisheries puts this population at high risk of extinction, and as a result 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 up to 21 leatherback sea turtles and killing up to three leatherback sea turtles in any given year, would have inconsequential impacts on the species' overall reproduction, numbers and distribution in the wild.

We supplemented these analyses by conducting an evaluation using the de-lifing procedure developed by Coulson et al. (1999), which examines the contribution individuals make to population growth by surviving and producing young. Populations grow based on the fraction of individuals that produce more than two offspring over their reproductive lifespan. We simulated two populations of X size, declining at 6% per year. We then used a Poisson distribution to simulate the number of individuals in that population that would be expected to produce offspring. We then used the Coulson et al. (1999) de-lifing procedure to consider what would happen if the turtles killed in the fishery were in the highest performing group (producing more offspring over their reproductive lifespan) and were killed before they would reproduce. This model allowed for the consideration of the original argument in the context of a population that still produces young and has recruitment, even if it does not replace adult losses.

Regardless of whether the population size was the smallest population size reported since 1999 (X=570), the median (X=1970), or the maximum (X=3,850), the effect of "de-lifing" 3, 5, 7, or 10 adults had no measurable effect on the population trajectory or reproductive output. The differences were greatest when the population size was smallest (X=570), but even then the effect on λ (i.e. population growth rate) was 1.3×10^{-4}, which is substantially smaller than the year-to-year variation in abundance. At greater abundances, the effect is even smaller.

Despite the evidence that suggests leatherback sea turtle populations in the Pacific are either extinct or face high risk of extinction, the number of leatherback turtles expected to die as a result of the HI SSLL fishery would not be expected to appreciably increase the extinction risk of the West Pacific Ocean population, reduce its probability of recovering from endangerment, or impede that recovery. Although declining, the population size is too large for the death of these individuals to have measurable effect on its status, trend, or dynamics.

If the population numbered in the 10s or 100s and showed signs of depensatory compensation (a feedback loop where a decrease in the breeding population leads to reduction in the survival of eggs or offspring that may cause a population to decline or collapse) the conclusion would be different, but that is not the case here. With a breeding population numbering in the 100s and 1000s, depensatory compensation is highly improbable (for example, see Fagan et al. 2001 and Fagan and Holmes 2006).

Based on our analysis, we expect no 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 upon which they depend, 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 yet 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 it. Given our concern for the West Pacific Ocean leatherback sea turtle's current status, we find that additional steps to further mitigate the effect of the HI SSLL fishery on leatherback sea turtles are appropriate.

The NMFS and USFWS (1998) Pacific population leatherback sea turtle recovery plan contains a number of goals and criteria that should be met to achieve recovery. These include all regional stocks that use U.S. waters have been identified to source beaches based on reasonable geographic parameters; each stock must average 5,000 (or a biologically reasonable estimate based on the goal of maintaining a stable population in perpetuity) females estimated to nest annually over six years; nesting populations at "source beaches" are either stable or increasing over a 25-year monitoring period; existing foraging areas are maintained as healthy environments; foraging populations are exhibiting statistically significant increases at several key foraging grounds within each stock region; all Priority #1 tasks have been implemented; a management plan designed to maintain sustained populations of turtles is in place.

We do not believe that the proposed action will impede progress on carrying out any aspect of the recovery plan or achieving the overall recovery strategy. The majority of the recovery criteria and priority #1 tasks will not be affected by the proposed action. Those that could potentially be affected and are most relevant to the analysis of the proposed action on recovery are: (1) each stock must average 5,000 (or a biologically reasonable estimate based on the goal of maintaining a stable population in perpetuity) females estimated to nest annually over six years; (2) nesting populations at "source beaches" are either stable or increasing over a 25-year monitoring period; (3) foraging populations are exhibiting statistically significant increases at several key foraging grounds within each stock region; and (4) reduce incidental mortality in commercial, recreational fisheries.

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. 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

Loggerheads from the North Pacific 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 approximately 341,071 North Pacific loggerhead sea turtles (T. Jones pers. comm. 2019). 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 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 ($\lambda > 1$), decrease geometrically ($\lambda < 1$), or remain the same ($\lambda = 1$). 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 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 subpopulations have not had consistent monitoring and/or the data were not available to do a PVA, therefore the trends are not known at this time.

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 mortality 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 from the North Pacific; 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 (2007) 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 species 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).

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 affecting the species already but will likely increase in the future. Global average sea levels are expected to continue to rise by at least several inches in the next 15 years, and by one to four feet by 2100 (Wuebbles et al. 2017). This could affect migrations and feeding patterns by changing ocean circulation but will also reduce the beach access for nesting due to significant beach armoring in Japan. Increased sand temperatures can also cause decrease egg survival and an increase in female turtles skewing sex ratios and affecting the reproductive capacity of the population.

Based on the 177 loggerhead 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 36 loggerhead sea turtles per year, (or up to 81 over a three-year period). From these we expect that up to six loggerhead sea turtles could die in any one year or up to 13 over a 3-year period). Assuming the North Pacific loggerhead includes about 340,000 individuals of all age classes, if the HI SSLL fishery interacted with up to 36 individual loggerhead turtles in a year, this would represent an interaction with 0.01% of the total number of North Pacific loggerhead sea turtles.

This rate of interaction, however would not be expected in every year. As indicated by our modeled estimate, predicted interaction numbers decline with the longer time frame considered in our model runs as depicted in **Table** 51 In other words, across 3 years we would expect that the HI SSLL fishery would interact with up to 81 individual loggerhead sea turtles in a 3-year period. The best estimate from our modeled interactions is 16 and 47 for 1 and 3 years, as noted in Table 56. While the best estimate is the mean over longer time periods, which is 16, we used the 95th percentile for our evaluation in large part because in any given year we would expect that the number of interactions between the HI SSLL fishery and loggerhead sea turtles would not likely exceed 36 but is likely to exceed 16 in some years. The corresponding mortality estimate with an interaction of up to 36 loggerhead sea turtles is six. Assuming there are currently about 14,000 adult North Pacific loggerhead sea turtles, and we applied all 36 loggerhead sea turtles to the adult portion of the population, then the HI SSLL fishery may interact with up to 0.2% of the adult population. The loss of six individual loggerhead sea turtles from an adult population comprised of about 14,000 animals represents 0.04% of the adult population, and the loss of six individuals from a total population of about 340,000 loggerhead sea turtles represents 0.0018% of the total population.

While our assessment estimates that up to six loggerhead sea turtles could die from interactions with the HI SSLL fishery, up to 30 would be released with varying degrees of injuries. The exact number of animals affected by non-lethal injuries and their fate is not known; however, we acknowledge that other responses can occur from these interactions, and can range from short term behavioral changes to longer lasting effects. These types of effects are not easy to monitor. We anticipate that most of the turtles that do not die as a result of their injuries may experience differing levels of harassment or other sublethal effects from the exposure to hooking and entanglement, which range from being temporary in nature such as elevated stress levels to more significant injuries that may heal over time but may affect the individual's lifetime fitness. Of the 30 animals that live through their capture or entanglement, we expect that a small portion of the turtles that survive may experience fitness level impacts.

If we assume that the six individuals that may die in any year and all of their potential offspring are from the smaller declining subpopulation, the risk to the species may be higher. We calculated the potential impacts to each subpopulation 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 0.001 % 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 0.002 % 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 0.003 % of the subpopulation.

Finally, similar to leatherback sea turtles, we examined the effect of maintaining the same number of interactions with the species as its numbers declined. If the total North Pacific loggerhead sea turtle is reduced to 50%, 25% and 12.5% of its current estimated abundance of 340,000, and the fishery interactions remain the same, in each scenario the fishery would interact with less than 0.09% of the total population. Total mortality would be less than 0.02% of the total population when the population reached 12.5% of its current size. Our analysis examines the effect of the HI SSLL fishery on loggerhead sea turtles, under several lines of reasoning. Under each scenario examined, we conclude that the impact of the HI SSLL fishery on loggerhead sea turtles is sufficiently small that we would not expect it to appreciably reduce the species' chances of survival and recovery in the wild.

The NMFS and FWS (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.

Moreover, we do not believe that the proposed action will impede progress on carrying out any aspect of the recovery plan or achieving the overall recovery strategy. The majority of the recovery criteria and priority # 1 tasks will not be affected by the proposed action. Those that could potentially be affected and are most relevant to the analysis of the proposed action on recovery are: (1) To the best extent possible, reducing take in international waters, (2) Ensuring all females estimated to nest annually (FENA) at "source beaches" are either stable or increasing for over 25 years; (3) 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"; (4) Ensuring foraging populations are exhibiting statistically significant increases at several key foraging grounds within each stock region; and (5) Reducing incidental mortality in commercial, recreational fisheries.

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 species 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

As described in the *Status of Listed Species*, the green sea turtle Central North Pacific 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 2004b). 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 and Chaloupka 2004b; 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 (Whittow and Balazs 1982; Balazs et al. 1996; Parker and Balazs 2010). 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, and an adult nester population of 4,000 (Chaloupka and Balazs 2007; IUCN 2012; Seminoff et al. 2015). The historic population size is estimated to be approximately 320,000 individuals (IUCN 2012).

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 species. Survival of this species 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 species highly vulnerable. As discussed in the *Status of Listed Species*, in October of 2018, East Island was demolished by Hurricane Walaka and impacts to the population are unknown at this time. No green sea turtles were using 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 was 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 seven 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.

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). Sharks and other marine predators also kill adult turtles (Seminoff et al. 2015). Fibropapillomatosis affects this species of green turtles (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 population 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 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 (Balazs et al. 1987; NMFS 2012b). Some Central North Pacific green turtles also interact with recreational hook-and-line fisheries (Francke et al. 2013). These activities are reasonably likely to continue, and may increase over time due to the effects of increased human population and increased human consumption of fish products.

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

Other climatic aspects, such as extreme weather events, precipitation, ocean acidification and sea level rise also have potential to affect marine turtle populations. Changes in global climatic patterns will likely have profound effects on the coastlines of every continent, thus directly affecting sea turtle nesting habitat (Wilkinson and Souter 2008). Lastly, studies examining the 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 and the Central North Pacific (P. Dutton pers. comm. 17 December 2018). NMFS estimates (Table 94) that between 20-64% of the turtles caught in the shallow-set longline fishery could be from the Central North Pacific (P. Dutton pers. comm. November 20, 2017).

Table 94. Genetic composition of green turtles sampled from the Hawaii shallow-set longline fishery (P. 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.

Green Sea Turtle Species (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 five green sea turtles per year. This rate of interaction, however would not be expected in every year. As indicated by our modeled estimate, predicted interaction numbers decline with the longer period considered in our model runs as depicted in Table 61**Table** 51 In other words, across 3 years we would expect that the HI SSLL fishery would interact with up to 10 individual green sea turtles in a 3-year period. The best estimate from our modeled interactions is 1.4 and 4.1 for 1 and 3 years, as noted in Table 61. While the mean is 1.4, we used the 95th percentile for our evaluation because in any given year we would expect that the number of interactions between the HI SSLL fishery and green sea turtles would not likely exceed 5 but is likely to exceed 1.4 in some years. The corresponding mortality estimate with an interaction of up to five green sea turtles is one. Assuming there are currently, about 8,000 adult Central North Pacific green sea turtles and we applied all five green sea turtles to the adult portion of the population, then the HI SSLL fishery may interact with up to 0.06% of the adult population. The loss of one individual Central North Pacific green sea turtle from an adult population comprised of about 8,000 animals represents 0.01% of the adult population, and the loss of one individuals from a total population of about 265,600 Central North Pacific green sea turtles represents 0.0004% of the total population.

While our assessment estimates that up to, one green sea turtle could die from interactions with the HI SSLL fishery, up to four would be released with varying degrees of injuries. The exact number of animals affected by non-lethal injuries and their fate is not known; however, we acknowledge that other responses can occur from these interactions, and can range from short term behavioral changes to longer lasting effects. These types of effects are not easy to monitor. We anticipate that most of the turtles that do not die as a result of their injuries may experience differing levels of harassment or other sublethal effects from the exposure to hooking and entanglement, which range from being temporary in nature such as elevated stress levels to more significant injuries that may heal over time but may affect the individual’s lifetime fitness. Of the four animals that live through their capture or entanglement, we expect that a small portion of the turtles that survive may experience fitness level impacts.

Finally, like we did for leatherback and loggerhead sea turtles, we examined the effect of maintaining the same number of interactions with the species as if its numbers declined. If the total Central North Pacific green sea turtle is reduced to 50%, 25% and 12.5% of its current estimated abundance of 265,600, and the fishery interactions remain the same, in each scenario the fishery would interact with less than 0.02% of the total population. Total mortality would be less than 0.003% of the total population when the population reached 12.5% of its current size.

Our analysis examines the effect of the HI SSLL fishery on Central North Pacific green sea turtles, under several lines of reasoning. Under each scenario examined, we conclude that the impact of the HI SSLL fishery on Central North Pacific green sea turtles is sufficiently small that we would not expect it to appreciably reduce the species' chances of survival and recovery in the wild.

Based on the evidence available, we conclude that the incidental take and resulting mortality of green sea turtles from the Central North Pacific 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 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 Pacific green sea turtles' ability to meet their lifecycle requirements and to retain the potential for recovery.

7.3.2 East Pacific

As described in the *Status of Listed Resources*, the East Pacific green sea turtle is listed as threatened and nesting has been steadily increasing at the primary nesting sites in 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 (NMFS and FWS 2007a; Wallace et al. 2010a). 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 threatened East Pacific green sea turtle, comprising nearly 58% of the total adult female population. The total for the entire Eastern Pacific green sea turtle is estimated at 20,112 nesting females (Seminoff et al. 2015). There are currently no total abundance estimates for this species, however, we estimate that the total population of the Eastern Pacific green sea turtle is greater than 265,600, the estimated total population of the Central North Pacific green sea turtle, since the number of nesting females is greater than the number of nesting females in the Central North Pacific (4,000).

Eastern Pacific green sea turtles are exposed to a variety of natural threats both at their nesting beaches and in the open ocean. The beaches where turtles nest and the nest themselves are threatened by hurricanes, tropical storms, storm surges, sand accretion, and rainfall associated with storms. The largest threat on nesting beaches to the East Pacific green sea turtle is reduced availability of habitat due to heavy armament and subsequent erosion. 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 mahi mahi fisheries operating out of Peru (Kelez et al. 2003; Alfaro-Shigueto et al. 2006) and, to a lesser extent, Chile (Donoso and Dutton 2010).

As described in the *Environmental Baseline*, past and present fisheries interactions have been, and continue to be, a threat to Eastern Pacific green turtles within the *Action Area*. Bycatch of green sea turtles in this species 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 that may result from 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 (Newson et al. 2009; Patricio et al. 2017; Jensen et al. 2018).

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 and the Central North Pacific (P. Dutton pers. comm. 17 December 2018). NMFS estimates (Table 95) that between 32-77% of the turtles caught in the shallow-set longline fishery could be from the Eastern Pacific (P. Dutton pers. comm. November 20, 2017).

Table 95. Genetic composition of green sea turtles sampled from the Hawaii shallow-set longline fishery (P. 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.

Green Sea Turtle Species (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 five green sea turtles per year. This number of interactions would not be expected in every year. As indicated by our modeled estimate, predicted interaction numbers decline with the longer period considered in our model runs as depicted in Table 61**Table** 51 In other words, across 3 years we would expect that the HI SSLL fishery would interact with up to 10 individual green sea turtles in a 3-year period. The best estimate from our modeled interactions is 1.4 and 4.1 for 1 and 3 years, as noted in Table 61. While the best estimate is the mean over longer time periods, which is 1.4, we used the 95th percentile for our evaluation in large part because in any given year we would expect that the number of interactions between the HI SSLL fishery and green sea turtles would not likely exceed 5 but is likely to exceed 1.4 in some years. The corresponding mortality estimate with an interaction of up to five green sea turtles is one. Assuming there are currently about 40,224 adult East Pacific green sea turtles and we applied all five green sea turtles to the adult portion of the population, then the HI SSLL fishery may interact with up to 0.01% of the adult population. The loss of one individual East Pacific green sea turtle from an adult population comprised of about 40,224 animals represents 0.003% of the adult population, and the loss of one individual from a total population of greater than 265,600 East Pacific green sea turtles represents 0.0004% of the total population.

While our assessment estimates that up to one green sea turtle could die from interactions with the HI SSLL fishery, up to four would be released with varying degrees of injuries. The exact number of animals affected by non-lethal injuries and their fate is not known; however, we acknowledge that other responses can occur from these interactions, and can range from short term behavioral changes to longer lasting effects. These types of effects are not easy to monitor. We anticipate that most of the turtles that do not die as a result of their injuries may experience differing levels of harassment or other sublethal effects from the exposure to hooking and entanglement, which range from being temporary in nature such as elevated stress levels to more significant injuries that may heal over time but may affect the individual's lifetime fitness. Of the four animals that live through their capture or entanglement, we expect that a small portion of the turtles that survive may experience fitness level impacts.

Finally, like we did for leatherback and loggerhead sea turtles, we examined the effect of maintaining the same number of interactions with the species as its numbers declined. If the total East Pacific green sea turtle is reduced to 50%, 25% and 12.5% of its current estimated abundance of 265,600, and the fishery interactions remain the same, in each scenario the fishery would interact with less than 0.02% of the total population. Total mortality would be less than 0.003% of the total population when the population reached 12.5% of its current size. Our analysis examines the effect of the HI SSLL fishery on East Pacific green sea turtles, under several lines of reasoning. Under each scenario examined, we conclude that the impact of the HI SSLL fishery on East Pacific green sea turtles is sufficiently small that we would not expect it to appreciably reduce the species' chances of survival and recovery in the wild.

Based on the evidence available, we conclude that the incidental take and resulting mortality of the East Pacific green sea turtle 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 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 turtles' ability to meet their lifecycle requirements and to retain the potential for recovery.

7.3.3 Central South Pacific

As described in the *Status of Listed Resources*, the Central South Pacific green sea turtle is listed as endangered and population trends are poorly understood. Based on available data, we estimate there are nearly 3,000 nesters. However, the largest nesting site, Scilly Atoll, which comprises roughly one third of the entire nesting abundance, was last monitored in the early 1990s (Balazs et al. 1995b) and has reportedly significantly declined in the past 30 years as a result of commercial exploitation (Conservation International Pacific Islands Program 2013). No sites have long-term monitoring programs, and no single site has had standardized surveys for even five continuous years. There is currently no total population estimates for this species, however, we estimate that the total population of the Central South Pacific green sea turtle is 184,870 sea turtles, which is 70% less than the estimated total population of the Central North Pacific green sea turtle, since the number of nesting females is 70% less than the number of nesting females for the Central North Pacific green sea turtle (4,000). 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 population likely has fewer than 3,600 nesters. Green turtle temporal population trends in the Central South Pacific green sea turtle 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 population, 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 population, presents significant challenges to any trend analyses (Seminoff et al. 2015).

Central South Pacific 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 green sea turtle. Depredation may have been a factor that contributed to the historical decline of this population. The best available data suggest that current nest and hatchling predation on several Central South Pacific green sea turtle nesting beaches and in water habitats is a potential threat to this population. 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 (Balazs 1983; Tuato'o-Bartley et al. 1993; Weaver 1996; NMFS and FWS 1998a; Obura and Stone 2002; Alefaio et al. 2006). Commercial harvest (a major threat), as well as subsistence and ceremonial harvest, are all contributing factors. Other anthropogenic threats include incidental capture in artisanal and commercial fisheries.

As described in the *Environmental Baseline*, past and present fisheries interactions may have been, and may continue to be, a threat to Central South Pacific green sea turtles within the *Action Area*. Bycatch of green sea turtles occurs in many fisheries throughout the geographic oceanic range of this species. While mitigation and minimization measures have reduced fisheries sea turtle bycatch in the U.S. in recent years, large numbers of Central South Pacific 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 (Newson et al. 2009; Patricio et al. 2017; Jensen et al. 2018).

Other climatic aspects, such as extreme weather events, precipitation, ocean acidification and sea level rise also have potential to affect marine turtle populations. Changes in global climatic patterns will likely have profound effects on the coastlines of every continent, thus directly affecting sea turtle nesting habitat (Wilkinson and Souter 2008). Lastly, studies examining the 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 and the Central North Pacific (P. 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, Central North Pacific, Southwest Pacific, and the Central South Pacific. To date, no animals from these additional four populations have been captured. However, the potential exists based on the movement of juvenile green sea turtles among these populations within the Pacific (see Seminoff et al. 2015 for a discussion of genetics and tagging studies evaluated in determining discreteness). NMFS estimates (Table 96) that 0-5% of the turtles caught in the shallow-set longline fishery could be from the Central South Pacific (P. Dutton pers. comm. November 20, 2017).

Table 96. Genetic composition of green turtles sampled from the Hawaii shallow-set longline fishery (P. 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.

Green Sea Turtle Species (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 population with great certainty as evidenced from the large CI, we will use the upper 95% CI to analyze the potential impacts to each population. 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 five green sea turtles per year. This rate of interaction, however would not be expected in every year. As indicated by our modeled estimate, predicted interaction numbers decline with the longer period considered in our model runs as depicted in Table 61**Table** 51 In other words, across 3 years we would expect that the HI SSLL fishery would interact with up to 10 individual green sea turtles in a 3-year period. The best estimate from our modeled interactions is 1.4 and 4.1 for 1 and 3 years, as noted in Table 61. While the best estimate is the mean over longer time periods, which is 1.4, we used the 95th percentile for our evaluation in large part because in any given year we would expect that the number of interactions between the HI SSLL fishery and green sea turtles would not likely exceed 5 but is likely to exceed 1.4 in some years. The corresponding mortality estimate with an interaction of up to five green sea turtles is one. Assuming there are currently about 6,000 adult Central South Pacific green sea turtles, and we applied all five green sea turtles to the adult portion of the population, then the HI SSLL fishery may interact with up to 0.08% of the adult population. The loss of one individual Central South Pacific green sea turtle from an adult population comprised of about 6,000 animals represents 0.02% of the adult population, and the loss of one individual from a total population of greater than 184,870 Central South Pacific green sea turtles represents 0.0005% of the total population.

While our assessment estimates that up to one green sea turtle could die from interactions with the HI SSLL fishery, up to four would be released with varying degrees of injuries. The exact number of animals affected by non-lethal injuries and their fate is not known; however, we acknowledge that other responses can occur from these interactions, and can range from short term behavioral changes to longer lasting effects. These types of effects are not easy to monitor. We anticipate that most of the turtles that do not die as a result of their injuries may experience differing levels of harassment or other sublethal effects from the exposure to hooking and entanglement, which range from being temporary in nature such as elevated stress levels to more significant injuries that may heal over time but may affect the individual's lifetime fitness. Of the four animals that live through their capture or entanglement, we expect that a small portion of the turtles that survive may experience fitness level impacts.

While we focus much of our analysis on the number of mortalities that are anticipated to occur from these interactions, we acknowledge that other forms of take occur from these interactions, which range from harassment to other sublethal effects. These types of effects are not easy to quantify but we anticipate that most of the turtles that do not die as a result of their injuries may experience differing levels of harassment or other sublethal effects from the exposure to hooking and entanglement, which range from being temporary in nature such as elevated stress to more significant injuries that may heal over time but may affect the individual's fitness. From the remaining nine interactions that may occur over the 3 years, we expect that a small portion of the turtles that survive may experience fitness level impacts.

Finally, similarly to our analyses for leatherback and loggerhead sea turtles, we examined the effect of maintaining the same number of interactions with the species as its numbers declined. If the total Central South Pacific green sea turtle is reduced to 50%, 25% and 12.5% of its current estimated abundance of 184,870, and the fishery interactions remain the same, in each scenario the fishery would interact with less than 0.02% of the total population. Total mortality would be less than 0.004% of the total population when the population reached 12.5% of its current size. Our analysis examines the effect of the HI SSLL fishery on Central South Pacific green sea turtles, under several lines of reasoning. Under each scenario examined, we conclude that the impact of the HI SSLL fishery on Central South Pacific green sea turtles is sufficiently small that we would not expect it to appreciably reduce the species' chances of survival and recovery in the wild.

Based on the evidence available, we conclude that the incidental take and resulting mortality from the Central South Pacific green sea turtle 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 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 turtles' ability to meet their lifecycle requirements and to retain the potential for recovery.

7.3.4 Central West Pacific

As described in the *Status of Listed Resources*, the Central West Pacific green sea turtle is listed as endangered and there is insufficient long-term and standardized monitoring information to adequately describe abundance and the population trend. The limited available information suggests a nesting population decrease in some areas like the Marshall Islands, or unknown trends in other areas such as Palau, Papua New Guinea, the Marianas, Solomon Islands, or the FSM (Maison et al. 2010). Currently, there are approximately 51 nesting sites and 6,518 nesting females in the Central West Pacific. There are no total population estimates for this species, however, we estimate that the total population of the Central West Pacific green sea turtle is greater than 265,600, the estimated total population of the Central North Pacific green sea turtle, since the number of nesting females is greater than the number of nesting females in the Central North Pacific (4,000). There are a number of unquantified nesting sites, possibly with small numbers, although specifics regarding these sites are unknown. The largest nesting site is in the FSM, and that particular site hosts approximately 22 percent of the total annual nesting females (Seminoff et al. 2015).

There is only one site for which 15 or more years of recent data are available for annual nester abundance (one of the standards for representing PVAs in Seminoff et al. 2015). This is at Chichijima, Japan, one of the major green turtle nesting concentrations in Japan (Horikoshi et al. 1994). The PVA has limitations, and does not fully incorporate other key elements critical to the decision making process such as spatial structure or threats. It assumes all environmental and anthropogenic pressures will remain constant in the forecast period and it relies on nesting data alone. The PVA suggests the probability that this population will fall below the trend reference or absolute abundance reference in 100 years approaches zero. The population has increased from a mean of approximately 100 females/year in the late 1970s/early 1980s to a mean of approximately 500 per year since 2000. Chaloupka et al. (2008a) reports an estimated annual population growth rate of 6.8% per year for the Chichijima nesting site.

Central West Pacific green sea turtles are exposed to a variety of natural threats both at their nesting beaches and in the open ocean. Predators such as ghost crabs, monitor lizards (*Varanus* sp.), wild pigs, rats, megapode birds, and iguanas have all been documented to consume large numbers of eggs in many of the nesting beaches (Seminoff et al. 2015) 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; Hitipeuw and Maturbongs 2002; Philip 2002; Humber et al. 2014). In addition to the collection of eggs from nesting beaches, the killing of nesting females continues to threaten the stability of green turtle populations. Other anthropogenic threats include incidental capture in artisanal and commercial fisheries.

As described in the *Environmental Baseline*, past and present fisheries interactions may have been, and may continue to be, a threat to Central West Pacific green sea turtles within the *Action Area*. Bycatch of green sea turtles from this population occurs in many fisheries throughout the geographic oceanic range of this 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 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 (Newson et al. 2009; Patrício et al. 2017; Jensen et al. 2018).

Other climatic aspects, such as extreme weather events, precipitation, ocean acidification and sea level rise also have potential to affect marine turtle populations. Changes in global climatic patterns will likely have profound effects on the coastlines of every continent, thus directly affecting sea turtle nesting habitat (Wilkinson and Souter 2008). Lastly, studies examining the 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 and the Central North Pacific (P. 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, Central West Pacific, Southwest Pacific, and the Central South Pacific. To date, no animals from these additional four populations have been captured. However, the potential exists based on the movement of juvenile green sea turtles among these populations within the Pacific (see Seminoff et al. 2015 for a discussion of genetics and tagging studies evaluated in determining discreteness). NMFS estimates (Table 97) that between 0 and8% of the turtles caught in the shallow-set longline fishery could be from the Central West Pacific (P. Dutton pers. comm. November 20, 2017).

Table 97. Genetic composition of green turtles sampled from the Hawaii shallow-set longline fishery (P. 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.

Green Sea Turtle Species (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 population with great certainty as evidenced from the large CI, we will use the upper 95% CI to analyze the potential impacts to each population. 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 five green sea turtles per year. This rate of interaction, however would not be expected in every year. As indicated by our modeled estimate, predicted interaction numbers decline with the longer period considered in our model runs as depicted in Table 61**Table** 51 In other words, across 3 years we would expect that the HI SSLL fishery would interact with up to 10 individual green sea turtles in a 3-year period. The best estimate from our modeled interactions is 1.4 and 4.1 for 1 and 3 years, as noted in Table 61. While the best estimate is the mean over longer time periods, which is 1.4, we used the 95th percentile for our evaluation in large part because in any given year we would expect that the number of interactions between the HI SSLL fishery and green sea turtles would not likely exceed 5 but is likely to exceed 1.4 in some years. The corresponding mortality estimate with an interaction of up to five green sea turtles is one. Assuming there are currently about 13,036 adult Central West Pacific green sea turtles and we applied all five green sea turtles to the adult portion of the population, then the HI SSLL fishery may interact with up to 0.08% of the adult population. The loss of one individual Central West Pacific green sea turtle from an adult population comprised of about 13,036 animals represents 0.008% of the adult population, and the loss of one individual from a total population of greater than 265,600 Central West Pacific green sea turtles represents 0.0004% of the total population. While our assessment estimates that up to one green sea turtle could die from interactions with the HI SSLL fishery, up to four would be released with varying degrees of injuries. The exact number of animals affected by non-lethal injuries and their fate is not known; however, we acknowledge that other responses can occur from these interactions, and can range from short term behavioral changes to longer lasting effects. These types of effects are not easy to monitor. We anticipate that most of the turtles that do not die as a result of their injuries may experience differing levels of harassment or other sublethal effects from the exposure to hooking and entanglement, which range from being temporary in nature such as elevated stress levels to more significant injuries that may heal over time but may affect the individual's lifetime fitness. Of the four animals that live through their capture or entanglement, we expect that a small portion of the turtles that survive may experience fitness level impacts.

Finally, similarly to our analyses for leatherback and loggerhead sea turtles, we examined the effect of maintaining the same number of interactions with the species as its numbers declined. If the total Central West Pacific green sea turtle is reduced to 50%, 25% and 12.5% of its current estimated abundance of 265,600, and the fishery interactions remain the same, in each scenario the fishery would interact with less than 0.02% of the total population. Total mortality would be less than 0.003% of the total population when the population reached 12.5% of its current size.

Our analysis examines the effect of the HI SSLL fishery on Central West Pacific green sea turtles, under several lines of reasoning. Under each scenario examined, we conclude that the impact of the HI SSLL fishery on Central West Pacific green sea turtles is sufficiently small that we would not expect it to appreciably reduce the species' chances of survival and recovery in the wild.

Based on the evidence available, we conclude that the incidental take and resulting mortality of green sea turtles from the Central West Pacific green sea turtle 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 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 turtles' ability to meet their lifecycle requirements and to retain the potential for recovery.

7.3.5 Southwest Pacific

As described in the *Status of Listed Resources*, the Southwest Pacific green sea turtle is listed as threatened and nesting occurs in many islands throughout their range , 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 turtles foraging (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 as cited in Seminoff et al. 2015). 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 as cited in Seminoff et al. 2015). The mean annual nester abundance varied between 26 and 1,801 during 1999–2004 (Limpus 2009). There are currently no total population estimates for this species, however, we estimate that the total population of the Southwest Pacific green sea turtle is greater than 265,600, the estimated total population of the Central North Pacific green sea turtle, since the number of nesting females is greater than the number of nesting females in the Central North Pacific (4,000).

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 green sea turtle species, also exist for the Southwest Pacific green turtle. 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 green turtles are vulnerable to harvest throughout Australia and neighboring countries such as New Caledonia, Fiji, Vanuatu, Papua New Guinea, and Indonesia (Limpus 2009). Nesting turtles are also vulnerable to the Queensland East Coast Trawl Fisheries and the Torres Strait Prawn Fishery, and to the extent they forage west of Torres Strait, they are also vulnerable to the northern prawn fishery. Total mortality of green turtles in fisheries bycatch is not known because there is not reliable reporting of threatened species bycatch in Australian commercial fisheries (Limpus 2009).

As described in the *Environmental Baseline*, past and present fisheries interactions may have been, and may continue to be, a threat to Southwest Pacific green sea turtles within the *Action Area*. Bycatch of green sea turtles occurs in many fisheries throughout the geographic oceanic range of this species. While mitigation and minimization measures have reduced fisheries sea turtle bycatch in the U.S. in recent years, large numbers of Southwest Pacific 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 (Newson et al. 2009; Patricio et al. 2017; Jensen et al. 2018).

Other climatic aspects, such as extreme weather events, precipitation, ocean acidification and sea level rise also have potential to affect marine turtle populations. Changes in global climatic patterns will likely have profound effects on the coastlines of every continent, thus directly affecting sea turtle nesting habitat (Wilkinson and Souter 2008). Lastly, studies examining the 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 and the Central North Pacific (P. 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, Central West Pacific, Southwest Pacific, and the Central South Pacific. To date, no animals from these additional four populations have been captured. However, the potential exists based on the movement of juvenile green sea turtles among these populations within the Pacific (see Seminoff et al. 2015 for a discussion of genetics and tagging studies evaluated in determining discreteness). NMFS estimates (Table 98) that between 0 and 8% of the turtles caught in the shallow-set longline fishery could be Southwest Pacific green sea turtles (P. Dutton pers. comm. November 20, 2017).

Table 98. Genetic composition of green turtles sampled from the Hawaii shallow-set longline fishery (P. 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.

Green Sea Turtle Species (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 population with great certainty as evidenced from the large CI, we will use the upper 95% CI to analyze the potential impacts to each population. 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 five green sea turtles per year. This rate of interaction, however would not be expected in every year. As indicated by our modeled estimate, predicted interaction numbers decline with the longer period considered in our model runs as depicted in Table 61**Table** 51 In other words, across 3 years we would expect that the HI SSLL fishery would interact with up to 10 individual green sea turtles in a 3-year period. The best estimate from our modeled interactions is 1.4 and 4.1 for 1 and 3 years, as noted in Table 61. While the best estimate is the mean over longer time periods, which is 1.4, we used the 95th percentile for our evaluation in large part because in any given year we would expect that the number of interactions between the HI SSLL fishery and green sea turtles would not likely exceed 5 but is likely to exceed 1.4 in some years. The corresponding mortality estimate with an interaction of up to five green sea turtles is one. Assuming there are currently about 166,116 adult Southwest Pacific green sea turtles and we applied all five green sea turtles to the adult portion of the population, then the HI SSLL fishery may interact with up to 0.002% of the adult population. The loss of one individual Southwest Pacific green sea turtle from an adult population comprised of about 166,116 animals represents 0.0006% of the adult population, and the loss of one individual from a total population of greater than 265,600 Southwest Pacific green sea turtles represents 0.0004% of the total population.

While our assessment estimates that up to one green sea turtle could die from interactions with the HI SSLL fishery, up to four would be released with varying degrees of injuries. The exact number of animals affected by non-lethal injuries and their fate is not known; however, we acknowledge that other responses can occur from these interactions, and can range from short term behavioral changes to longer lasting effects. These types of effects are not easy to monitor. We anticipate that most of the turtles that do not die as a result of their injuries may experience differing levels of harassment or other sublethal effects from the exposure to hooking and entanglement, which range from being temporary in nature such as elevated stress levels to more significant injuries that may heal over time but may affect the individual’s lifetime fitness. Of the four animals that live through their capture or entanglement, we expect that a small portion of the turtles that survive may experience fitness level impacts.

Finally, like we did for leatherback and loggerhead sea turtles, we examined the effect of maintaining the same number of interactions with the species as its numbers declined. If the total Southwest Pacific green sea turtle is reduced to 50%, 25% and 12.5% of its current estimated abundance of 265,600, and the fishery interactions remain the same, in each scenario the fishery would interact with less than 0.02% of the total population. Total mortality would be less than 0.003% of the total population when the population reached 12.5% of its current size. Our analysis examines the effect of the HI SSLL fishery on Southwest Pacific green sea turtles, under several lines of reasoning. Under each scenario examined, we conclude that the impact of the HI SSLL fishery on Southwest Pacific green sea turtles is sufficiently small that we would not expect it to appreciably reduce the species’ chances of survival and recovery in the wild.

Based on the evidence available, we conclude that the incidental take and resulting mortality of the Southwest Pacific green sea 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 threatened Southwest Pacific green sea turtles' 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 turtles'' ability to meet their lifecycle requirements and to retain the potential for recovery.

7.3.6 East Indian/West Pacific

As described in the *Status of Listed Species*, the East Indian/West Pacific green sea turtle is listed as threatened and there are four sites for which 15 or more years of recent data are available for annual nester abundance: Sabah Turtle Islands in Malaysia; Royal Navy Center in Khram Island, Thailand; Redang in Terrengganu, Myanmar; and Thameela Islands, Myanmar. Only Sabah Turtle Islands represent a sizable nesting population, estimated at 7,011 in 2011 with a sex ratio of 1M: 4F (Pilcher 2010b). The PVA indicates that the nesters from Sabah Turtle Islands in Malaysia, with an estimated 7,000 nesters, will likely continue to increase, while the nesters from the Royal Navy Center in Khram Island, Thailand (estimated 297 nesters), Redang in Terrengganu, Myanmar (estimated 278 nesters), and Thameela Islands, Myanmar (estimated 109 nesters) will likely continue to decline (Seminoff et al. 2015). The total abundance for this population is estimated at 77,009 nesters (Seminoff et al. 2015). There are currently no total population estimates for this species, however, we estimate that the total population of the East Indian/West Pacific green sea turtle is greater than 265,600, the estimated total population of the Central North Pacific green sea turtle, since the number of nesting females is greater than the number of nesting females in the Central North Pacific (4,000).

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 green sea turtles are exposed to a variety of natural threats both at their nesting beaches and in the open ocean. The best available data suggest that current nest and hatchling predation on the East Indian-West Pacific green sea turtle is prevalent. Depredation of nests by feral animals is also widespread in many South Asian areas (Sunderraj et al. 2001 as cited in Seminoff et al. 2015; 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 1996). 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. Sharks may be congregating 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 as cited in Seminoff et al. 2015), Western Australia (Raidal and Prince 1996; Aguirre and Lutz 2004), and on PhuQuoc in Vietnam (Ministry of Fisheries 2003). Epidemiological studies indicate rising incidence of this disease (George 1997); thus, the above list will likely grow in the future.

As described in the *Environmental Baseline*, past and present fisheries interactions may have been, and may continue to be, a threat to East Indian-West Pacific green sea turtle within the *Action Area*. Bycatch of green sea turtles occurs in many fisheries throughout the geographic oceanic range of this species. While mitigation and minimization measures have reduced fisheries sea turtle bycatch in the U.S. in recent years, large numbers of East Indian-West Pacific 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 (Newson et al. 2009; Patrício et al. 2017; Jensen et al. 2018).

Other climatic aspects, such as extreme weather events, precipitation, ocean acidification and sea level rise also have potential to affect marine turtle populations. Changes in global climatic patterns will likely have profound effects on the coastlines of every continent, thus directly affecting sea turtle nesting habitat (Wilkinson and Souter 2008). Lastly, studies examining the 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 and the Central North Pacific (P. 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, Central West Pacific, Southwest Pacific, and the Central South Pacific. To date, no animals from these additional four populations have been captured. However, the potential exists based on the movement of juvenile green sea turtles among these populations within the Pacific (see Seminoff et al. 2015 for a discussion of genetics and tagging studies evaluated in determining discreteness). NMFS estimates (Table 99) that between 0 and 10% of the turtles caught in the shallow-set longline fishery could be from East Indian-West Pacific green sea turtles (P. Dutton pers. comm. November 20, 2017).

Table 99. Genetic composition of green turtles sampled from the Hawaii shallow-set longline fishery (P. 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.

Green Sea Turtle Species (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 population with great certainty as evidenced from the large CI, we will use the upper 95% CI to analyze the potential impacts to each population. 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 five green sea turtles per year. This rate of interaction, however would not be expected in every year. As indicated by our modeled estimate, predicted interaction numbers decline with the longer period considered in our model runs as depicted in Table 61**Table** 51 In other words, across 3 years we would expect that the HI SSLL fishery would interact with up to 10 individual green sea turtles in a 3-year period. The best estimate from our modeled interactions is 1.4 and 4.1 for 1 and 3 years, as noted in Table 61. While the best estimate is the mean over longer time periods, which is 1.4, we used the 95th percentile for our evaluation in large part because in any given year we would expect that the number of interactions between the HI SSLL fishery and green sea turtles would not likely exceed 5 but is likely to exceed 1.4 in some years. The corresponding mortality estimate with an interaction of up to five green sea turtles is one. Assuming there are currently about 96,261 adult East Indian-West Pacific green sea turtles, and we applied all five green sea turtles to the adult portion of the population, then the HI SSLL fishery may interact with up to 0.005% of the adult population. The loss of one individual East Indian-West Pacific green sea turtle from an adult population comprised of about 96,261 animals represents 0.001% of the adult population, and the loss of one individual from a total population of greater than 265,600 East Indian-West Pacific green sea turtles represents 0.0004% of the total population.

While our assessment estimates that up to one green sea turtle could die from interactions with the HI SSLL fishery, up to four would be released with varying degrees of injuries. The exact number of animals affected by non-lethal injuries and their fate is not known; however, we acknowledge that other responses can occur from these interactions, and can range from short term behavioral changes to longer lasting effects. These types of effects are not easy to monitor. We anticipate that most of the turtles that do not die as a result of their injuries may experience differing levels of harassment or other sublethal effects from the exposure to hooking and entanglement, which range from being temporary in nature such as elevated stress levels to more significant injuries that may heal over time but may affect the individual's lifetime fitness. Of the four animals that live through their capture or entanglement, we expect that a small portion of the turtles that survive may experience fitness level impacts.

Finally, like we did for leatherback and loggerhead sea turtles, we examined the effect of maintaining the same number of interactions with the species as its numbers declined. If the total East Indian-West Pacific green sea turtle is reduced to 50%, 25% and 12.5% of its current estimated abundance of 265,600, and the fishery interactions remain the same, in each scenario the fishery would interact with less than 0.02% of the total population. Total mortality would be less than 0.003% of the total population when the population reached 12.5% of its current size. Our analysis examines the effect of the HI SSLL fishery on East Indian-West Pacific green sea turtles, under several lines of reasoning. Under each scenario examined, we conclude that the impact of the HI SSLL fishery on East Indian-West Pacific green sea turtles is sufficiently small that we would not expect it to appreciably reduce the species' chances of survival and recovery in the wild.

Based on the evidence available, we conclude that the incidental take and resulting mortality of the East Indian-West Pacific green sea 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 threatened East Indian-West Pacific green sea turtles' 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 turtles' 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 2007d, 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 (Marquez-M. and Van Dissel 1982; Cornelius and Robinson 1985; Gates et al. 1996; Valverde and Gates 1999; Bézy and Valverde 2012) 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 10 observed interactions that have resulted in a hooking or entanglement. All 10 interactions resulted in all gear being removed. Based on the 10 olive ridley sea turtles that were caught by the SSLL fishery, NMFS predicts that the HI SSLL fishery will interact with up to five olive ridley sea turtles per year, (or up to 11 over a three-year period). From these we expect that up to one olive ridley sea turtle could die in any one year or up to two over a 3-year period).

This rate of interaction, however would not be expected in every year. As indicated by our modeled estimate, predicted interaction numbers decline with the longer period considered in our model runs as depicted in Table 58. In other words, across 3 years we would expect that the HI SSLL fishery would interact with up to 11 individual olive ridley sea turtles in a 3-year period. The best estimate from our modeled interactions is one and four for 1 and 3 years, as noted in Table 58. While the best estimate is the mean over longer time periods, which is 1, we used the 95th percentile for our evaluation in large part because in any given year we would expect that the number of interactions between the HI SSLL fishery and olive ridley sea turtles would not likely exceed 5 but is likely to exceed 1 in some years. The corresponding mortality estimate with an interaction of up to five olive ridley sea turtles is one. Assuming total abundance for olive ridley sea turtles is more than one million individuals, and we assume that five individuals from this species could be hooked or entangled, then the HI SSLL fishery may interact with up to 0.0005% of the total population. The loss of one individual olive ridley sea turtle assuming total abundance equals more than one million animals, represents 0.0001% of the total population.

Although our assessment estimates that up to one olive ridley sea turtle could die from interactions with the HI SSLL fishery, up to four would be released with varying degrees of injuries. While we focus much of our analysis on the number of mortalities that are anticipated to occur from these interactions, we acknowledge that other forms of take occur from these interactions, which range from harassment to other sublethal effects. These types of affects are not easy to quantify but we anticipate that most of the turtles that do not die as a result of their injuries may experience differing levels of harassment or other sublethal effects from the exposure to hooking and entanglement which range from being temporary in nature such as elevated stress to more significant injuries that may heal over time but may affect the individual's fitness. Projected interactions were 5, 8, and 11 olive ridleys over one, two and 3-years respectively. In our *Response* section, we concluded two mortalities across those 3 years could occur. 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. From the remaining nine interactions over the 3 years, we expect a small portion of the turtles that survive may experience fitness level impacts.

Finally, as we did for leatherback sea turtles, we examined the effect of maintaining the same number of interactions with the species as its numbers declined. If the total population of olive ridley sea turtles were reduced to 50%, 25% and 12.5% of its current estimated abundance of over one million, and the fishery interactions remain the same, in each scenario the fishery would interact with less than 0.03% of the total population. Total mortality would be less than 0.005% of the total population when the population reached 12.5% of its current size. Our analysis examines the effect of the HI SSLL fishery on olive ridley sea turtles, under several lines of reasoning. Under each scenario examined, we conclude that the impact of the HI SSLL fishery on olive ridley sea turtles is sufficiently small that we would not expect it to appreciably reduce the species' chances of survival and recovery in the wild.

The low number of interactions expected from the HI 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 HI SSLL fishery poses to individual olive ridley sea turtles to have 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. (2017) believe 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. 2011a; Rice and Harley 2012; Young et al. 2017), with purse seine fisheries being secondary sources of mortality. In addition to bycatch-related mortality, the oceanic whitetip shark is a preferred species for retention because its large fins obtain a high price in the Asian fin market, and comprises approximately 2% of the global fin trade (Clarke et al. 2006a). This high value and demand for oceanic whitetip fins incentivizes the retention and subsequent finning of oceanic whitetip sharks when caught, and thus represents the main 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. 2017). 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. (2017) 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 six 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). This rate of interaction, however would not be expected in every year. As indicated by our modeled estimate, predicted interaction numbers decline with the longer period considered in our model runs as depicted in Table 89**Table** 51 In other words, across 3 years we would expect that the HI SSLL fishery would interact with up to 299 individual oceanic whitetip sharks in a 3-year period. The best estimate from our modeled interactions is 19 to 32 and 54 to 92 for 1 and 3 years respectively, as noted in Table 89. While the best estimate is the mean over longer time periods, which is 6 to 10 individuals (1 year), we used the 95th percentile for our evaluation in large part because in any given year we would expect that the number of interactions between the HI SSLL fishery and oceanic whitetip sharks would not likely exceed 102 but is likely to exceed 32 individuals in some years. The corresponding mortality estimate with an interaction for the 1 year is up to 32 sharks. Assuming there are currently about 96,971 adult oceanic whitetip sharks in the western Pacific, and we applied all 102 interactions to the adult portion of the population, then the HI SSLL fishery may interact with up to 0.1052% of the adult population. The loss of 32 individuals from an adult population represents 0.0330% of the adult population, and the loss of 102 individual from a total population of greater than 200,000 oceanic whitetip sharks in the western Pacific represents 0.0160% of the total population.

Finally, like we did for other species under consideration in this opinion, we examined the effect of maintaining the same number of interactions with the species as its numbers declined. If the total oceanic whitetip shark population is reduced to 50%, 25% and 12.5% of its current estimated abundance of 200,000, and the fishery interactions remain the same, in each scenario the fishery would interact with less than 0.102% of the total population. Total mortality would be 0.128% of the total population when the population reached 12.5% of its current estimated abundance. Our analysis examines the effect of the HI SSLL fishery on oceanic whitetip shark, under several lines of reasoning. Under each scenario examined, we conclude that the impact of the HI SSLL fishery on oceanic whitetip shark is sufficiently small that we would not expect it to appreciably reduce the species' chances of survival and recovery in the wild.

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 HI 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 shark. 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 three 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.

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 are 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.

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. Although the mean (7) yields a better prediction of the cumulative effect of this fishery over many years, the upper estimate (13) is a better estimate for interactions in any given single year. The hooking or entanglement of up to 13 (mean=7) giant manta rays per year is expected to result in the mortality of up to 4 giant manta rays in any given year from their exposure as described in detail in the *Exposure and Response* section of this biological opinion.

Although our assessment estimates that up to four giant manta rays could die from interactions with the HI SSLL fishery, up to nine would be released with varying degrees of injuries. While we focus much of our analysis on the number of mortalities that are anticipated to occur from these interactions, we acknowledge that other forms of take occur from these interactions, which range from harassment to other sublethal effects. These types of effects are not easy to quantify but we anticipate that most of the giant manta rays that do not die as a result of their injuries may experience differing levels of harassment or other sublethal effects from the exposure to hooking and entanglement, which range from being temporary in nature such as elevated stress to more significant injuries that may heal over time but may affect the individual's fitness.

Assuming the West Pacific Ocean population includes a total of 2,200 giant manta rays of all age classes, which represents our low estimate for the population, if the HI SSLL fishery interacted with up to 13 giant manta rays in a year, this would represent 0.06% of the total giant manta ray population. This rate of interaction, however would not be expected in every year. As indicated by our modeled estimate, predicted interaction numbers decline with the longer period considered in our model runs as depicted in Table 72. In other words, across 3 years we would expect that the HI SSLL fishery would interact with up to 22 individual giant manta rays in a 3-year period. The best estimate from our modeled interactions is 7 and 12 for 1 and 3 years respectively, as noted in Table 51. While the best estimate is the mean over longer time periods, which is 7, we used the 95th percentile for our evaluation in large part because in any given year we would expect that the number of interactions between the HI SSLL fishery and giant manta rays would not likely exceed 13 but would exceed 7 in some years. The corresponding mortality estimate with an interaction of up to 13 giant manta rays is four. The loss of four individual giant manta rays from a total population comprised of 2,200 animals represents 0.18% of the total population.

Our best estimate is that the HI SSLL fishery will hook and entangle *up to* 13 giant manta ray each year (mean = 7), and from these interactions *up to* 2 will die. Assuming this fishery continues for at least another 25 years, based on the mean estimated number of interactions we would expect that the HI SSLL fishery would hook or entangle about 175 more giant manta rays and about 50 of these animals would die from their interactions over a 25-year time frame. The upper estimate of 13 giant manta rays would produce *up to* 325 interactions in 25 years, with *up to* 100 mortalities across those 25 years.

Finally, like we did for leatherback sea turtles, we examined the effect of maintaining the same number of interactions with the species as its numbers declined. If the total West Pacific Ocean population of giant manta ray is reduced to 50%, 25% and 12.5% of its current estimated abundance of 2,200 and the fishery interactions remain the same, in each scenario the fishery would interact with less than 5% of the total population. Total mortality would be less than 2% of the total population when the population reached 12.5% of its current size. Our analysis examines the effect of the HI SSLL fishery on giant manta rays, under several lines of reasoning. Under each scenario examined, we conclude that the impact of the HI SSLL fishery on giant manta rays is sufficiently small that we would not expect it to appreciably reduce the species' chances of survival and recovery in the wild.

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 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 estimate 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. 2017b).

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 et al. 2016; NMFS 2017c; Juarez-Ruiz et al. 2018). Additionally, 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). 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 fishery.

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 to 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, nine of those animals will die. The second year is expected to result in 12 combined mortalities, and 14 mortalities are expected over 3 years. This rate of interaction, however would not be expected in every year. As indicated by our modeled estimate, predicted interaction numbers decline with the longer period considered in our model runs as depicted in Table 92**Table** 51 In other words, across 3 years we would expect that the HI SSLL fishery would interact with up to 17 individual Guadalupe fur seals in a 3-year period. The best mortality estimate from our modeled interactions is 9 and 14 individuals for 1 and 3 years respectively, as noted in Table 92.

While the best estimate is the mean over longer time periods, which is 5 individuals (1 year), we used the 95th percentile for our evaluation in large part because in any given year we would expect that the number of interactions between the HI SSLL fishery and Guadalupe fur seals would not likely exceed 9 but is likely to exceed 5 individuals in some years. The corresponding mortality estimate with an interaction for 1 year is up to nine individuals. Assuming there are currently about 20,000 Guadalupe fur seals and we applied all 11 interactions to the population, then the HI SSLL fishery may interact with up to 0.055% of the total population. The loss of nine individuals from a population represents 0.0450% of the population. At this time, the abundance for the adult population is unknown; therefore, effects to the adult population cannot be specifically addressed.

Finally, like we did for other species under consideration in this opinion, we examined the effect of maintaining the same number of interactions with the species as its numbers declined. If the total Guadalupe fur seal population is reduced to 50%, 25% and 12.5% of its current estimated abundance of 20,000, and the fishery interactions remain the same, in each scenario the fishery would interact with less than 0.11% of the total population. Total mortality would be 0.36% of the total population when the population reached 12.5% of its current estimated abundance. Our analysis examines the effect of the HI SSLL fishery on Guadalupe fur seals under several lines of reasoning. Under each scenario examined, we conclude that the impact of the HI SSLL fishery on the species is sufficiently small that we would not expect it to appreciably reduce the species' chances of survival and recovery in the wild.

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 3 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= 0.085% of the species over 3 years; Mortalities: 14/20,000=0.0007*100= 0.07% of the species over 3 years). We also note that the estimated population growth is estimated to be approximately 10% (Carretta et al. 2017b) and interactions have been occurring with the fishery since 2014. Therefore, the population abundance has been increasing and continues to increase during a period 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 North Pacific loggerhead sea turtles, threatened green sea turtles in the Eastern Pacific, Central North Pacific, East Indian-West Pacific, Southwest Pacific, endangered green sea turtles in the Central West Pacific and Central South Pacific, threatened olive ridley sea turtles and olive ridley sea turtles

from the endangered Mexico breeding population, 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 loggerhead sea turtles, Eastern Pacific green sea turtles, Central North Pacific green sea turtles, East Indian-West Pacific green sea turtles, Central West Pacific green sea turtles, Southwest Pacific green sea turtles, Central South Pacific green sea turtles, endangered Mexico breeding population of olive ridley sea turtles, and threatened (other) populations of olive ridley sea turtles, threatened oceanic whitetip sharks, threatened giant manta rays and threatened 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 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 3 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 individual threatened and endangered species, or their designated critical habitat, would likely be taken by harassing, harming, pursuing, hunting, shooting, wounding, killing, trapping, capturing, or collecting or attempting to engage in any such conduct. If take is anticipated to occur, then the Services must describe the amount or extent of such anticipated take and the reasonable and prudent measures, and terms and conditions necessary to minimize the impacts of incidental take (FWS and NMFS 1998). If, during the course of the action, this level of incidental take is exceeded for any of the species as listed, NMFS SFD must immediately reinitiate formal consultation with NMFS PRD pursuant to the section 7 regulations (50 CFR 402.16). 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 predicted to result from the implementation of the proposed action are shown in Table 100 below.

Table 100. 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	1
*Green sea turtle (all species)	5	1
**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 turtles. The anticipated number killed for green turtles is 0-1 annually, which we rounded to one.

**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

Reasonable and prudent measures are those actions necessary or appropriate to minimize the impacts, i.e., amount or extent, of incidental take (50 CFR 402.02). The associated terms and conditions set out the specific methods by which the reasonable and prudent measures are to be accomplished, e.g., who is to be educated, when/what/how; the actions necessary to reduce predation; how to avoid the species; or the protocol for monitoring. Reasonable and prudent measures along with the terms and conditions that implement them cannot alter the basic design, location, scope, duration, or timing of the action, and may involve only minor changes. Terms and conditions of an incidental take statement must include reporting and monitoring requirements that assure adequate action agency oversight of any incidental take [50 CFR 402.14(i)(1)(iv) and (i)(3)]. Compliance with the terms and conditions specified in the incidental take statement exempts the Federal agency and any permit or license applicant involved from the taking prohibitions of the ESA up to the level specified in the incidental take statement.

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. NMFS shall evaluate and develop a minimization measure, or a suite of minimization measures designed to reduce the incidental capture and mortality of leatherback and loggerhead sea turtles in the HI SSLL fishery.
2. 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.
3. 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.
4. 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.
5. 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.
6. 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 9.3 above. These terms and conditions are non-discretionary, and if NMFS SFD fails to adhere to these terms and conditions, or fails to implement measures requiring the 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, upon receiving a signed biological opinion, set an annual interaction limit for the fishery of 16 leatherback sea turtles, which represents an approximate 25% reduction in the number of turtles from our predicted interaction numbers in this biological opinion. If the fishery reaches this limit then

NMFS shall close the HI SSLL fishery for the remainder of the calendar year. NMFS may modify this requirement as appropriate upon implementation of minimization measures identified in Term and Condition 1c.

b. NMFS SFD shall set a trip limit not to exceed 2 leatherback sea turtles or 5 loggerhead sea turtles per vessel trip. Any vessel that reaches the established trip limit must immediately stop fishing and return to port. These vessels will not engage in shallow set longline fishing for 5 days while NMFS evaluates vessel and turtle interactions to identify any problems and determine if guidance can be provided to the vessel to reduce the interactions.

   Vessels that reach the per trip limit for either leatherback or loggerhead sea turtles twice in a calendar year shall be prohibited from shallow-set longline fishing for the remainder of the calendar year. NMFS shall require any vessel that reaches a trip limit for either species twice in one calendar year to have an annual vessel limit of 2 leatherbacks or 5 loggerheads for the following year.

c. NMFS SFD shall evaluate and develop minimization measures to reduce incidental catch and mortality of leatherback and loggerhead sea turtles. At a minimum, NMFS must consider the following 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), or otherwise implement TurtleWatch under a regulatory framework, to develop meaningful measures to minimize incidental catch of leatherback sea turtles and loggerhead sea turtles. NMFS may replace or modify the requirements of Term and Condition 1a, if after this evaluation NMFS develops alternative minimization measures for both species that are commensurate with the goal of a 25% reduction of the incidental capture and mortality of leatherback and loggerhead sea turtles.

d. NMFS SFD shall conduct an analysis of fleet-wide interactions with leatherback and loggerhead sea turtles to evaluate patterns of interactions between these species and vessel owners and operators. 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 the fleet to reduce impact of individual vessels on leatherback and loggerhead sea turtles and provide fleet-wide guidance on how to avoid reaching limits established in Terms and Conditions a and b.

   NMFS SFD may develop and implement other measures to reduce the impact of the shallow-set longline fishery provided that they are as effective leatherback and loggerhead sea turtle trip limits and individual vessel limits described in Term and Condition 1b. By addressing individual behavior(s), NMFS may establish it has met (in part) the requirements of RPM 1, and may modify or remove the requirements in Term and Condition 1b.

e. NMFS SFD shall conduct an interactive workshop that includes fishermen, observers, relevant 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. Prior to the workshops, 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 workshops.

    WPRFMC estimated that mortalities for leatherback turtles could have been reduced by approximately 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 sea turtles.

    f. 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.
    g. On at least an annual basis, NMFS SFD shall evaluate the effectiveness of adopted measures and report findings to NMFS PRD.
    h. If Terms and Conditions 1a and 1 b have not been implemented by regulation by January 1, 2020, the HI SSLL fishery may reopen under an annual interaction limit of 16 leatherback and 17 loggerhead sea turtles until such regulations are in place. If the fishery reaches either limit, then NMFS shall close the HI SSLL fishery for the remainder of the calendar year or until regulations implementing RPM 1 allow for reopening.

2. The following terms and conditions implement Reasonable and Prudent Measure No. 2:
    a. NMFS SFD shall maintain observer coverage at rates that are at least as high or above those 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 longline 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.

3. The following terms and conditions implement Reasonable and Prudent Measure No. 3:

   a. NMFS SFD shall continue to require and conduct protected species workshops for owners and operators of registered vessels for use with Hawaii limited entry longline fishing permits vessels, and should include crewmembers on 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 in order to be able 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 WCPFC 2017b, 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.

4. The following term and condition implements Reasonable and Prudent Measure No. 4:
   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.

5. The following term and condition implements Reasonable and Prudent Measure No. 5:
    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.
6. The following terms and conditions implement Reasonable and Prudent Measure No. 6:
    a. NMFS SFD shall evaluate and develop minimization measure(s), as appropriate 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; and
    e. Studies on ecology, habitat use, genetics, and post interaction survivability of leatherback and loggerhead sea turtles and other ESA-listed marine species.

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.

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