Endangered Species Act – Section 7 Consultation

Biological Opinion

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

Activity: Continued operation of the Hawaii-based deep-set pelagic longline fishery.

Consulting Agency: National Marine Fisheries Service, Pacific Islands Region, Protected Resources Division

Approved By: ______________________

Michael D. Tosatto
Regional Administrator, Pacific Islands Region

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Table of Contents

ACRONYMS.....4

1 INTRODUCTION.....6

2 CONSULTATION HISTORY.....11

3 DESCRIPTION OF THE ACTION.....14

4 ACTION AREA.....16

5 STATUS OF LISTED SPECIES.....18

5.1 SPECIES AND CRITICAL HABITAT NOT LIKELY TO BE ADVERSELY AFFECTED.....19
5.2 SPECIES LIKELY TO BE ADVERSELY AFFECTED.....24

6 ENVIRONMENTAL BASELINE.....97

6.1 HUMPBACK WHALES.....97
6.2 SPERM WHALES.....98
6.3 MAIN HAWAIIAN ISLANDS INSULAR FALSE KILLER WHALES.....99
6.4 LOGGERHEAD TURTLES.....99
6.5 LEATHERBACK TURTLES.....102
6.6 OLIVE RIDLEY TURTLES.....104
6.7 GREEN TURTLES.....104
6.8 SCALLOPED HAMMERHEAD SHARKS.....106
6.9 ALL SPECIES: IMPACTS ASSOCIATED WITH CLIMATE CHANGE.....106

7 EFFECTS OF THE ACTION.....108

7.1 HUMPBACK WHALES.....112
7.2 SPERM WHALES.....114
7.3 MAIN HAWAIIAN ISLANDS INSULAR FALSE KILLER WHALE.....116
7.4 LOGGERHEAD TURTLES.....120
7.5 LEATHERBACK TURTLES.....124
7.6 OLIVE RIDLEY TURTLES.....131
7.7 GREEN TURTLES.....133
7.8 SCALLOPED HAMMERHEAD SHARKS.....136

8 CUMULATIVE EFFECTS.....138

9 INTEGRATION AND SYNTHESIS OF EFFECTS.....140

9.1 HUMPBACK WHALES.....140
9.2 SPERM WHALES.....144
9.3 MHI INSULAR FALSE KILLER WHALES.....147
9.4 LOGGERHEAD TURTLES.....150
9.5 LEATHERBACK TURTLES.....153
9.6 OLIVE RIDLEY TURTLES.....157
9.7 GREEN TURTLES.....160
9.8 INDO-WEST PACIFIC SCALLOPED HAMMERHEAD SHARK DPS.....164

10 CONCLUSION.....165

11 CONSERVATION RECOMMENDATIONS.....166

12 REINITIATION NOTICE.....167

13 INCIDENTAL TAKE STATEMENT.....167

14 LITERATURE CITED.....175

Figures and Tables

Table 1. ESA-listed marine species found in the action area..18
Table 2. Genetics results for turtle bycatch, shallow-set vs. deep-set, 1995-2014 26
Table 3. Genetics results for turtle bycatch in deep-set fishery from 2004 -2014 26
Table 4. Minimum population estimates and trends for sperm whales..33
Table 5. Annual olive ridley population estimates at major arribada nesting sites..80
Table 6. Estimates of current effective population size of Indo Pacific scalloped hammerhead sharks......................93
Table 7. Number of observed and estimated total interactions with listed species...111
Table 8. List of interactions and injury determination with humpback whales. 112
Table 9. Observed interactions with sperm whales..114
Table 10. List of interactions and injury determinations with false killer whales and unidentified blackfish...........116
Table 11. Observed interactions with unidentified blackfish and false killer whales...117
Table 12. Observed interactions with loggerheads...122
Table 13. Observed interactions with leatherbacks 126
Table 14. Annual Adult female leatherback mortality from the proposed action..128
Table 15. Observed interactions with olive ridley turtles 132
Table 16. Observed interactions with green turtles..134
Table 17. Observed interactions of the Indo-west Pacific scalloped hammerhead DPS..137
Table 18. Incidental Take Statement 170

Figure 1. Spatial distribution of fishing effort by the Hawaii longline deep-set fishery…………………………….17
Figure 2. Map of Scalloped hammerhead DPS boundaries…………………………………………………………. 22
Figure 3. Stock boundaries of MHI insular, NWHI insular, and pelagic false killer whales……………………….. 38
Figure 4. Observed interactions with false killer whales and unidentified blackfish in the overlap zone……………45
Figure 5. Map of Loggerhead Sea Turtle DPS boundaries………………………………………………………….. 49
Figure 6. Nesting population abundance for the East Pacific leatherback sea turtle……………………………….. 67
Figure 7. Satellite-tracked movements of 135 leatherback sea turtles in the Pacific Ocean………………………… 69
Figure 8. Distribution of longline effort of distant-water fleets in the Action area………………………………… 100

Acronyms

AFM	Adult female mortalities
AFR	Age at First Reproduction
ANE	Adult nester equivalent
BE	Biological Evaluation
CFR	Code of Federal Regulations
CITES	Convention on International Trade in Endangered Species
CMM	Conservation and Management Measure
DPS	Distinct population segment
EEZ	Exclusive Economic Zone
ESA	Endangered Species Act
FAO	Food and Agriculture Organization of the United Nations
FEIS	Final Environmental Impact Statement
FENA	Females estimated to nest annually
FEP	Fishery Ecosystem Plan
FMP	Fishery Management Plan
FR	Federal Register
HLA	Hawaii Longline Association
IAC	Inter-American Convention for the Protection and Conservation of Sea Turtles
IFKW	Insular False Killer Whale
ITS	Incidental Take Statement
MHI	Main Hawaiian Islands
MMPA	Marine Mammal Protection Act
M&SI	Mortality and Serious Injury
NMFS	National Marine Fisheries Service (also NOAA Fisheries)
NOAA	National Oceanic and Atmospheric Administration
PBR	Potential Biological Removal
PDO	Pacific Decadal Oscillation
PIFSC	Pacific Islands Fisheries Science Center
PIR	Pacific Islands Region
PIRO	Pacific Islands Regional Office
PIROP	Pacific Islands Region Observer Program
PNG	Papua New Guinea
PRD	Protected Resources Division, NMFS Pacific Islands Regional Office
PSW	Protected Species Workshop
PVA	Population Viability Assessment
QET	Quasi-extinction threshold
SCL	Straight carapace length
SEIS	Supplement Environmental Impact Statement
SFD	Sustainable Fisheries Division, NMFS Pacific Islands Regional Office
SQE	Susceptibility to Quasi-Extinction
SSC	Scientific and Statistical Committee of the WPFMC
SSLL	Shallow-set longline
STAJ	Sea Turtle Association of Japan
TEWG	Turtle Expert Working Group

TRP	Take Reduction Plan
USFWS	U.S. Fish and Wildlife Service
WCP	Western Central Pacific
WCPFC	Western and Central Pacific Fisheries Commission
WPFMC	Western Pacific Fishery Management Council

1 Introduction

Section 7(a) (2) of the Endangered Species Act (ESA) of 1973, as amended (ESA; 16 U.S.C. 1536(a) (2)) requires each federal agency to ensure that any action it authorizes, funds, or carries 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" an ESA-listed species, that agency is required to consult formally with the National Marine Fisheries Service (for marine species or their designated critical habitat) or the U.S. Fish and Wildlife Service (for terrestrial and freshwater species or their designated critical habitat). Federal agencies are exempt from this formal consultation requirement if they have concluded that an action "may affect, but is not likely to adversely affect" ESA-listed species or their designated critical habitat, and the National Marine Fisheries Service (NMFS, or NOAA Fisheries) or the U.S. Fish and Wildlife Service (USFWS) concur with that conclusion (see ESA Section 7 Implementing Regulations; 50 CFR 402). The purpose of this Biological Opinion (BiOp) is for the purpose of consulting on ESA-listed species under NMFS jurisdiction.

The proposed action is the continued operation of the Hawaii deep-set pelagic longline fishery (deep-set fishery) as currently managed under the existing regulatory framework. NMFS anticipates the deep-set fishery to continue to operate largely unchanged in terms of fishing location, the number of vessels, catch rates of target, non-target, and bycatch species, depth of hooks, or deployment techniques in setting longline gear. From 2004-2008 the number of hooks has gradually increased from just over 30 million hooks to over 40 million hooks set. Based on an analysis of fishery effort trends NMFS estimates 128 vessels to make approximately 1,305 trips, 18,592 sets, and deploy 46,117,532 hooks annually (NMFS 2013c).

The Hawaii-based deep-set longline fishery is one of two Hawaii-based longline fisheries, which also includes a shallow-set longline fishery. An overview of the Hawaii-based longline fisheries is given below to provide context for the deep-set fishery.

1.1 The Hawaii-based Longline Fisheries

Domestic longline fishing around Hawaii consists of two separately managed fisheries: a deep-set fishery that targets primarily bigeye tuna, and a shallow-set fishery that targets swordfish. No regulatory distinction was made between the two fisheries in 1999, when a Court Order entered a limited injunction that, among other things, closed an area north of Hawaii to longlining while NMFS prepared an environmental impact statement on the fishery. The Court subsequently expanded its order while NMFS completed its analysis of the fishery's interactions with sea turtles. NMFS regulations promulgated in 2002 and the related biological opinion prohibiting longline fishing of swordfish because of its impacst on sea turtles were vacated and remanded to the agency.

In 2004, NMFS issued new regulations based on a 2004 biological opinion reopening the swordfish fishery. The new regulations contained measures to reduce the number and severity of turtle bycatch interactions[1], while restricting the fishery to considerably less fishing effort than pre-2001 effort levels (NMFS 2004b). The deep-set fishery became an increasingly larger proportion of total Hawaii-based longline fishing effort until there was only deep-set fishing during the swordfish-targeting prohibition from 2001-2004. Since 2004, the deep-set fishery has made up the majority of total domestic longline fishing around Hawaii (NMFS 2013d). The regulatory history of Hawaii-based longline fisheries is described in the 2001 Pelagics Final Environmental Impact Statement (NMFS 2001), the 2004 Biological Opinion (BiOp) (NMFS 2004a), the 2004 Pelagics FSEIS (NMFS 2004b), the 2005 BiOp (NMFS 2005a), 2008 BiOp (NMFS 2008a), and the 2012 BiOp (NMFS 2012a). For a comprehensive regulatory overview see the 2009 Pelagics Fishery Ecosystem Plan (FEP) (WPFMC 2009a).

Longline fishing, as defined in regulations 50 CFR 665.800, utilizes a type of fishing gear consisting of a mainline that exceeds one nautical mile (6,076 ft) in length that is suspended horizontally in the water column, from which branchlines with hooks are attached (NMFS 2008a). The term “Hawaii-based” is used to specify those longline vessels primarily operating out of Hawaii, in order to distinguish them from other longline vessels operating in the same waters, but based in other states or nations. The Hawaii-based longline fleet grew to 141 vessels in 1991 when the Western Pacific Fishery Management Council (the Council) established a limited entry program to control the fishery’s growth. The limited entry program allows a ceiling of 164 vessels, and vessel size is limited to a maximum of 101 feet in length (NMFS 2001, WPFMC 2006, and WPFMC 2009b). Approximately 124-128 vessels are typically active during any given year.

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. Longline fishing involves setting a mainline horizontally at a preferred depth in the water column using floats spaced at regular intervals. Three to five radio buoys are usually attached at fairly regular intervals along the mainline so the line may be easily located both for initial retrieval and in case the mainline breaks during fishing operations. Branchlines are clipped to the mainline at regular intervals during gear deployment, and each branchline has a single baited hook. Mainline lengths can be 30 to 100 km (18 to 60 nm) long depending on the fishery, individual vessel, or other factors, and after the mainline is completely deployed, the gear is allowed to “soak” for several hours before being retrieved (“hauled”). In longlining, a “set” is a discrete unbroken section of line floats and branchlines. Usually, only one set is fished per day. Depending on the fishery, trips around Hawaii are typically three to five weeks long (NMFS 2001, NMFS 2005a, WPFMC 2006, Beverly and Chapman 2007, WPFMC 2009b).

Longline fishing for swordfish is known as shallow-set longline fishing because the bait is set at relatively shallower depths of 30 to 90 m. The portion of the mainline with branchlines attached is suspended between floats at about 20 to 75 m of depth, and the branchlines hang off the mainline another 10 to 15 m. Only four to six branchlines are clipped 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 branchlines and hauled during the day.

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[1] An interaction occurs when a protected species is hooked or entangled by fishing gear, thus encompassing all hookings, entanglements, captures, and mortalities, whether the animal is brought on board the vessel or not.

Formerly, J-hooks and squid bait were used, but since 2004, 18/0 or larger circle hooks and mackerel-type bait have been required. These gear restrictions were implemented to reduce turtle bycatch. The most productive swordfishing areas for Hawaii-based longliners are north of Hawaii outside the U.S. Exclusive Economic Zone (EEZ) on the high seas.

Tunas, primarily bigeye and yellowfin, are targeted in the deep-set fishery, which sets bait at 40 to 350 m depth (depending on the target species). To target a range of depths, a hydraulic mainline deployment device ("line shooter") is used on deep sets to deploy the mainline faster than the speed of the vessel, so that the mainline sags as the gear sinks to the desired depth. Deep-set longline gear is typically set in the morning and hauled at night. In contrast to shallow-set longline fishing, a minimum of 15, but typically 20 to 30, branchlines are clipped to the mainline at regular intervals between floats. A typical deep-set consists of 2,000 to 2,500 hooks. Lightsticks are not attached to the branchlines, as they are prohibited onboard Hawaii-based deep-set longline fishing vessels. The most productive tuna fishing areas vary seasonally around the Hawaiian Islands and there is some overlap with the shallow-set fishery. A comparison of shallow-set and deep-set longline fishing methods is provided in Bartram and Kaneko (2004).

Hawaii-based longline fisheries are managed by federal regulations pertaining to the Pelagics FEP, as well as other federal fisheries regulations that apply to the western Pacific. For the complete set of these federal regulations, see 50 CFR Part 665, Title 50 of the Code of Federal Regulations (CFR), Parts 229, 300, 404, 600, and 665 and for a summary see Summary of Hawaii Longline Regulations (NMFS 2013e).

1.2 The Regulatory framework for the Hawaii-based Longline Fisheries

The Pelagics FEP establishes the framework for the Council and NMFS to manage U.S. pelagic fisheries in the Western Pacific Region (WPFMC 2009a). It contains a description of the pelagic and benthic environment, pelagic species managed, and the fisheries managed by the Council and NMFS, including the deep-set longline fishery. It also includes information regarding the management program, identification, and description of essential fish habitat and Federal, State, and local laws and Acts that apply to pelagic fisheries under the Council's jurisdiction. The Biological evaluation (BE) prepared by the NMFS PIRO Sustainable Fisheries Division in June 2013 to support reinitiation of consultation (see Section 2 below) also provides details of the fishery's management (NMFS 2013c).

Actions taken to amend the plan or modify regulations that govern the deep-set fishery, including those mentioned in the section below, are undertaken in accordance with the Magnuson-Stevens Fishery Conservation and Management Act (Magnuson-Stevens Act or MSA), National Environmental Policy Act (NEPA), ESA, Marine Mammal Protection Act (MMPA), and other applicable Federal statutes.

The NMFS regulation summaries and compliance guides for the Hawaii longline fisheries provide an overview of the current regulations and requirements for permit holders, vessel owners, and operators (http://www.fpir.noaa.gov/SFD/SFD_regs_2.html).

Longline fishermen need a Hawaii longline limited access permit (Hawaii longline permit) and certificates to fish in the U.S. EEZ (3 to 200 nm from shore) around Hawaii. NMFS requires other permits and endorsements to longline fish on the high seas (offshore of 200 nm), as described below. The Hawaii longline permit is also required to land longline-caught pelagic fish in Hawaii, regardless of where the fish were caught. Permitted vessels are limited to 101 feet in length. The requirements include Federal fishing and transshipment logbooks, a vessel monitoring system, vessel and gear identification, and requirements for notifying NMFS before leaving on a fishing trip so the agency can place an observer, if needed. Permit holders or designated agents must notify NMFS of the type of fishing that will be conducted (deep- or shallow-set) prior to leaving port and may not switch types after the trip is underway. NMFS deploys observers on 20 percent of all deep-set fishing trips to document protected species interactions and collect other fishery information.

NMFS prohibits longline fishing in certain areas (typically within 50-75 nm of shore) around the Northwestern Hawaiian Islands (NWHI) and in the Main Hawaiian Islands (MHI) Longline Fishing Prohibited Area to protect marine mammals, reduce gear conflicts among pelagic fisheries, and reduce localized overfishing of pelagic fish. The MHI Longline Fishing Prohibited Area, implemented in 1992, was revised by the False Killer Whale Take Reduction Plan (FKWTRP) regulations in 2012 to eliminate a seasonal contraction of the boundary and to prohibit longline fishing year-round within the larger boundary (see 50 CFR 229.37). In 2006 and 2009, Presidential Proclamations 8031 and 8336 established marine national monuments around the NWHI and Pacific Remote Island Areas (PRIA) of Johnston Atoll, Kingman Reef and Palmyra Atoll, Howland and Baker Islands, Jarvis Island, and Wake Island where, except for Wake Island, the fishery occasionally operated in the past. The NWHI and PRIA monuments extend to approximately 50 nm from shore inside of which NMFS prohibits commercial fishing. In addition, the False Killer Whale Take Reduction Plan requires NMFS to prohibit deep-set longline fishing within a Southern Exclusion Zone (SEZ), an area south of the MHI and inside the EEZ boundary, if a specified number of interactions with the Hawaii pelagic stock of false killer whales occurs with the deep-set fishery while fishing inside the EEZ around Hawaii (50 CFR 229.37). The SEZ is triggered to mitigate impacts of the fishery on false killer whales.

Each year, the owner and operator (captain) of a longline vessel must complete a protected species workshop (PSW) conducted by NMFS Pacific Islands Regional Office (PIRO). Currently NMFS offers workshops in person and online. The workshops instruct fishermen in mitigation, handling, and release techniques for sea turtles, seabirds, and marine mammals. Fishermen must carry and use specific equipment, and follow certain procedures for handling and releasing sea turtles, seabirds, and marine mammals that may be caught incidentally while fishing.

The proposed action does not include new regulations and retains all existing ones. The current requirements governing the Hawaii deep-set longline fishery are grouped into the following categories, and each category is summarized below.

Fishing Permits and Certificates on board the vessel:

- Hawaii Longline Limited Access Permit
- Marine Mammal Authorization Program Certificate
- High Seas Fishing Compliance Act Permit for fishing on the high seas (outside 200 nm)
- Western and Central Pacific Fisheries Commission (WCPFC) Convention Area Endorsement for fishing on the high seas in the convention area

- Protected Species Workshop (PSW) Certificate
- Western Pacific Receiving Vessel Permit, if applicable
- 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 unit
- 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)
- Carry a fishery observer on board if requested by NMFS; NMFS places observers on board at least 20 percent of deep-set longline trips
- Follow fisheries observer guidelines

Recent and Proposed Deep-set Fishery Management Measures

Management of the deep-set fishery is dynamic as a result of the cross-jurisdictional and pelagic nature of the fishery, as well as the participation by the U.S. in international fisheries management organizations. Recent fishery management actions are described in this section to contextualize the regulatory framework within which the fishery operates. Recent and proposed management measures that pertain to the deep-set longline fishery are listed below. Changes in the operation of the fishery that resulted from these measures are also discussed in this section.

1. Revised swordfish trip limits in the deep-set fishery to reduce regulatory discards (77 FR 43721, July 26, 2012).

2. A bigeye tuna catch limit of 3,763 mt for vessels in the U.S. pelagic longline fisheries in the western and central Pacific Ocean, consistent with the requirements of the Convention on the Conservation and Management of Highly Migratory Fish Stocks in the Western and Central Pacific Ocean (77 FR 51709, August 27, 2012).

3. Final rule to implement the final False Killer Whale Take Reduction Take Reduction Plan for the Hawaii-based longline fisheries (77 FR 71260, November 29, 2012).

4. Final requirements for U.S. fishing vessels used for commercial fishing that offload or receive transshipments of highly migratory species, provide bunkering or other supports services to fishing vessels, as well as reporting and purse seine discard requirements (77 FR 71501, December 3, 2012).

5. Consolidated and Further Continuing Appropriations Act, 2012 (CFCAA 2012) – Section 113, authorizing the use, assignment, allocation, and management of catch limits of highly migratory fish stocks assigned by the Commission to U.S. Participating Territories (Pub. L. 112-55, November 18, 2011), as extended by CFCAA, 2013 (Pub. L.113-6). The

provisions of this section expired in December of 2013. On March 28, 2014, NMFS approved Amendment 7 to the Fishery Ecosystem Plan for Pacific Pelagic Fisheries of the Western Pacific Region (Pelagics FEP), which established a framework for approving highly migratory species catch or fishing effort limits and accountability measures for U.S. Participating Territories (American Samoa, Guam, and the CNMI), as well as authority to transfer a portion of such limits to eligible vessels issued permits under the Pelagics FEP. For 2014, NMFS would specify for each U.S. Participating Territory an annual bigeye tuna catch limit of 2,000 mt, and further specify that each PT may allocate up to 1,000 mt of that limit to a U.S. longline vessel or vessels identified in a fishing agreement. If implemented by final rule, catch and effort under Amendment 7 is expected to be consistent with the catch and effort observed under CFCAA 2012.

6 Under Amendment 7, NMFS proposed an annual limit of 2,000 metric tons (mt) of longline-caught bigeye tuna for each U. S. Participating territory, using the framework described in the proposed rule. NMFS would allow a territory to allocate up to 1,000 mt of the 2,000 mt each year to a U.S. longline fishing vessel or vessels in a specified fishing agreement that meets the established criteria. NMFS would monitor, attribute, and restrict catches of longline-caught bigeye tuna, including catches made under a specified fishing agreement, using the procedures and accountability measures described in the proposed rule. The longline bigeye tuna catch limit specifications would be effective in 2014. The effort under Amendment 7 is expected to approximate the effort under CFCAA 2012, discussed *supra*.

2 Consultation History

NMFS issued a final biological opinion (BiOp) on proposed regulatory amendments to the Fishery Management Plan for the Pelagic Fisheries of the Western Pacific Region (Pelagics FMP) on February 23, 2004 (2004 BiOp) (NMFS 2004a). That BiOp considered effects of authorizing the Pelagics FMP as modified by proposed sea turtle protection measures, on threatened and endangered species under NMFS' jurisdiction. That BiOp determined that authorization of the Pelagics FMP as modified by proposed sea turtle protection measures, was not likely to jeopardize the continued existence of threatened and endangered green turtles (*Chelonia mydas*), endangered leatherback turtles (*Demochelys coriacea*), threatened loggerhead turtles (*Caretta caretta*), or threatened and endangered olive ridley turtles (*Lepidochelys olivacea*). That BiOp also determined that the proposed action was not likely to adversely affect humpback or sperm whales.

The 2004 BiOp contained an Incidental Take Statement (ITS) specifying take levels of threatened and endangered sea turtles anticipated to occur incidental to the proposed action. The ITS differentiated anticipated interactions in the various components of the fishery and specified separate take levels for the Hawaii-based shallow-set longline fishery which targets swordfish; the Hawaii-based deep-set longline fishery which targets tuna; and the handline, troll, and pole and line fisheries managed under the Pelagics FMP as well as the longline fisheries based out of America Samoa. The ITS stipulated that formal consultation be reinitiated upon exceeding specified take levels, which is one of the criteria for reiniation. NMFS promulgated a final rule on April 2, 2004 which implemented sea turtle protection measures analyzed in the 2004 BiOp and the March 2004 Final Supplemental Environmental Impact Statement (FSEIS) (NMFS 2004b).

In the 2004 BiOp, NMFS set incidental take limits for individual fishery components, such that exceedence of take in one fishery component would not require reinitiation of formal consultation in components of the fishery in which take levels had not been exceeded. In 2004, the deep-set component of the Hawaii-based pelagic longline fishery was estimated to have exceeded the take of olive ridley sea turtles authorized in the 2004 ITS. Formal consultation on the deep-set component of the Hawaii-based pelagic longline fishery was reinitiated on February 17, 2005.

On October 5, 2005 NMFS issued a BiOp for the deep-set longline fishery that authorized incidental take of green, leatherback, loggerhead, and olive ridley sea turtles (NMFS 2005a). That BiOp also determined that the proposed action was not likely to jeopardize the continued existence of humpback whales; however, incidental take was not authorized in the ITS at that time. One of the reiniation criteria states that NMFS must re-initiate formal consultation if, in a single fishing year, the fishery incidentally captures or kills 50 percent or more than the total take level specified/anticipated for multiple years for any species. NMFS must also re-initiate consultation if the incidental capture or mortality exceeds the level specified over a consecutive three-year period. NMFS deploys observers on at least 20 percent of all deep-set trips to, among other things, monitor fishery interactions with sea turtles. Take estimates are then derived by the expanding the observed interactions to a fleet-wide total. The estimates are then compared to the ITS (Table 18).

The no-jeopardy 2005 BiOp did not include an ITS for humpback whales because the necessary findings under MMPA section 101(a)(5)(E) had not been completed and an MMPA permit had not been issued to the fishery. On May 28, 2010, NMFS issued a permit, pursuant to MMPA section 101(a)(5)(E), for a period of three years to authorize the incidental, but not intentional, taking of four humpback whales from the central North Pacific (CNP) stock by the Hawaii deep- and shallow-set longline fisheries (75 FR 29984). On June 1, 2010, NMFS amended the 2005 BiOp ITS to include CNP humpback whales.

The sequence of events related to this formal consultation and leading up to the development of this BiOp are provided below.

On June 5, 2013, the Sustainable Fisheries Division (SFD) of NMFS PIR sent a memorandum to the Protected Resources Division (PRD) of NMFS PIR, requesting reinitiation of formal consultation on effects of the ongoing operation of the Hawaii-based pelagics, deep-set longline fishery on ESA-listed sea turtles and marine mammals. Reinitiation of consultation is required 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.

The reinitiation triggers for this consultation are as follows: NMFS listed the Main Hawaiian Islands insular false killer whale (MHI IFKW) as an endangered distinct population segment (DPS) in December 2012, which under the ESA constitutes a new “species”. In addition, in 2011, the deep-set fishery entangled and/or hooked (specifics unknown) a sperm whale. Prior to this incident, there had not been an observed sperm whale interaction in the deep-set fishery since observer coverage began in 1994. The 2005 BiOp did not authorize take for this species. In February of 2014, while the consultation was ongoing, the deep-set fishery exceeded the ITS for leatherbacks in the 2005 BiOp.

In the June 5, 2013 BE (NMFS 2013c) accompanying the request for reinitiation, SFD concluded the proposed action may affect, but is not likely to adversely affect Hawaiian monk seals, hawksbill sea turtles, and blue, fin, sei, and North Pacific right whales. SFD concluded the proposed action may affect, and is likely to adversely affect, MHI insular false killer whales, humpback whales, and sperm whales. Further, SFD concluded the proposed action may affect, and is likely to adversely affect green, leatherback, olive ridley sea turtles, and the North Pacific loggerhead sea turtle DPS. SFD also analyzed the proposed action’s effects on two proposed DPSs of scalloped hammerhead sharks, which are now listed as threatened and endangered in the Indo-West Pacific and Eastern Pacific, respectively (79 FR 38214, July 3, 2014). The action area overlaps with the Pacific DPSs, and SFD determined that the deep-set fishery may affect these populations.

On June 5, 2013, in a record to the file, the Regional Administrator (RA) of NMFS PIR issued findings and decisions pursuant to section 7(a)(2), 7(d), and 7(a)(4) of the ESA, 16 U.S.C. § 1536(d). The RA concluded that the continuing operation of the deep-set pelagic tuna longline fishery during the consultation period would not violate the prohibition in ESA section 7(d) against making irreversible or irretrievable commitment of resources that preclude the formulation or implementation of reasonable and prudent alternatives to avoid the likelihood of jeopardy to listed species or adverse modification of critical habitat. In addition NMFS determined that during the period of consultation the continued operation of the deep-set fishery may affect, but is not likely to adversely affect, MHI insular false killer whales and sperm whales, thus satisfying ESA section 7(a)(2).

In February of 2014, while the consultation was ongoing the deep-set fishery, exceeded the 2005 ITS for leatherbacks. This exceedence was identified after approval of Pelagics FEP Amendment 7, but prior to approval of the final rule implementing Amendment 7, and triggered reinitiation of consultation for leatherback sea turtles. Accordingly, on June 16, 2014, NMFS issued findings and decisions pursuant to ESA sections 7(a)(2) and 7(d) authorizing the continuation of the deep-set fishery, while withholding action on the Amendment 7 final rule pending completion of reinitiated consultation. NMFS concluded that the likelihood of an interaction with a sperm whale or MHI insular false killer whale prior to the completion of the current consultation remains extremely low. Accordingly, the determination that the longline fishery may affect, but is not likely to adversely affect, these two whale species during the period of reinitiated consultation, as documented in the June 5, 2013 memorandum, remained valid and consistent with ESA section 7(a)(2). In addition, NMFS estimated that the fishery could potentially interact with up to 63 leatherback turtles, including 22.7 mortalities, prior to the completion of reinitiated consultation. This represents up to an additional 0.5 adult nesting female mortality per year prior to the completion of consultation, which is unlikely to have a detectable influence on population trends and is not expected to impact the survival or recovery of the species. Accordingly, NMFS determined that continuation of the Hawaii deep-set fishery during the reinitiation period would not violate ESA section 7(a)(2). NMFS determined that the 2005 BiOp remains valid for this species during the period of reinitiated consultation. Finally, NMFS determined that continuation of the deep-set fishery during the period of consultation does not constitute an irreversible or irretrievable commitment of resources under ESA section 7(d).

PIRO/PRD provided a draft biological opinion to PIRO/SFD, with a request for comments, on July 22, 2014. Comments were received from PIRO/SFD on August 4, 2014. On August 19, 2014, the draft biological opinion was provided to the Applicant, HLA, for the proposed action. On August 29, 2014 a teleconference was conducted with the applicant to receive their comments on the draft BiOp. These comments were reflected in written comments that were received from HLA on September 3, 2014.

3 Description of the Action

The proposed action is the continued operation of the deep-set fishery, as currently managed under the Pelagics FEP, and the existing regulatory regime as described in section1. NMFS anticipates the deep-set fishery to continue to operate largely unchanged from what has occurred in the last several years under CFCAA 2012 in terms of fishing location, the number of vessels that deep-set, catch rates of target, non-target, and bycatch species, depth of hooks, or deployment techniques in setting longline gear. NMFS estimates 128 vessels to make approximately 1,305 trips, with 18,592 sets, and 46,117,532 hooks annually. NMFS anticipates the deep-set fishery will continue to operate throughout the year, fish sustainably, and utilize proven bycatch mitigation measures to manage impacts to ESA-listed marine mammals and sea turtles as required under regulations in 50 CFR parts 229 and 665.

The Pelagics FEP and its implementing regulations at 50 CFR Part 665 contain requirements to prevent and mitigate the effects of the deep-set fishery on protected species – sea turtles, marine mammals, and seabirds. Regulations specific to managing impacts of the deep-set fishery on marine mammals are found at 50 CFR 229.

Prohibited Longline Fishing Areas around Hawaii:

- NWHI Protected Species Zone
- MHI Longline Fishing Prohibited Area (as modified by the False Killer Whale Take Reduction Plan)

Protected Species Workshop (PSW):

- Each year, longline vessel owners and operators must complete a PSW, and receive a certificate
- A valid PSW certificate is required to renew a Hawaii longline permit
- The operator of a longline vessel must have a valid PSW certificate on board the vessel while fishing

When Deep-set Longline Fishing[2]:

- Float lines must be at least 20 m long
- Attach at least 15 branch lines between two consecutive floats (for basket gear the requirement is at least 10 branch lines)
- No light sticks may be used or onboard a vessel
- Limits on the number of swordfish kept or landed per trip. (These limits ensure that deep-set longliners do not make shallow-sets, which, in turn, could interact with sea turtles) The limits are as follows:
    - If an observer is on board, there is no limit on the number of swordfish kept or landed
    - If there is no observer on board, and if only circle hooks are used, the limit is 25 per trip
    - If there is no observer on board, and any hooks other than circle hooks are used, the limit is 10 per trip
- Any nylon monofilament line used in a branch line or leader must have a diameter/thickness of 2.0 mm or larger. Any other line material (e.g., wire) used in a branch line or leader must have a breaking strength of 400 lb or greater
- Use circle hooks with a maximum wire diameter (thickness) of 4.5 mm. The hook shank must contain round wire that can be measured with a caliper. If the hook point is offset from the shank, it must be offset by no more than 10°

Sea Turtle and Seabird Handling and Mitigation Measures:

- Adhere to regulations for safe handling and release of sea turtles and seabirds
- Have on board the vessel all required turtle handling and dehooking gear specified in regulations
- Choose between side-setting or stern-setting the longline gear with additional regulatory specifications to reduce seabird interactions
- Certain seabird mitigation measures must be used on vessels deploying longline gear from the stern or side of the vessel when deep-set fishing north of the Equator

Marine Mammal Handling and Release:

- NMFS encourages longline vessel owners/operators to follow the marine mammal handling guidelines provided at the Protected Species Workshop
- Report injuries or mortalities of marine mammals on the Marine Mammal Authorization Program's Mortality/Injury Reporting Form
- Post NMFS-approved marine mammal handling and release, and captain notification placards on every vessel
- Captains must supervise the handling and release of marine mammals. Crew on longline vessels is encouraged to notify the captain if a marine mammal interaction occurs, so the captain can supervise the handling and release
- Captains must comply with all applicable gear, reporting, and handling requirements of the False Killer Whale Take Reduction Plan regulations

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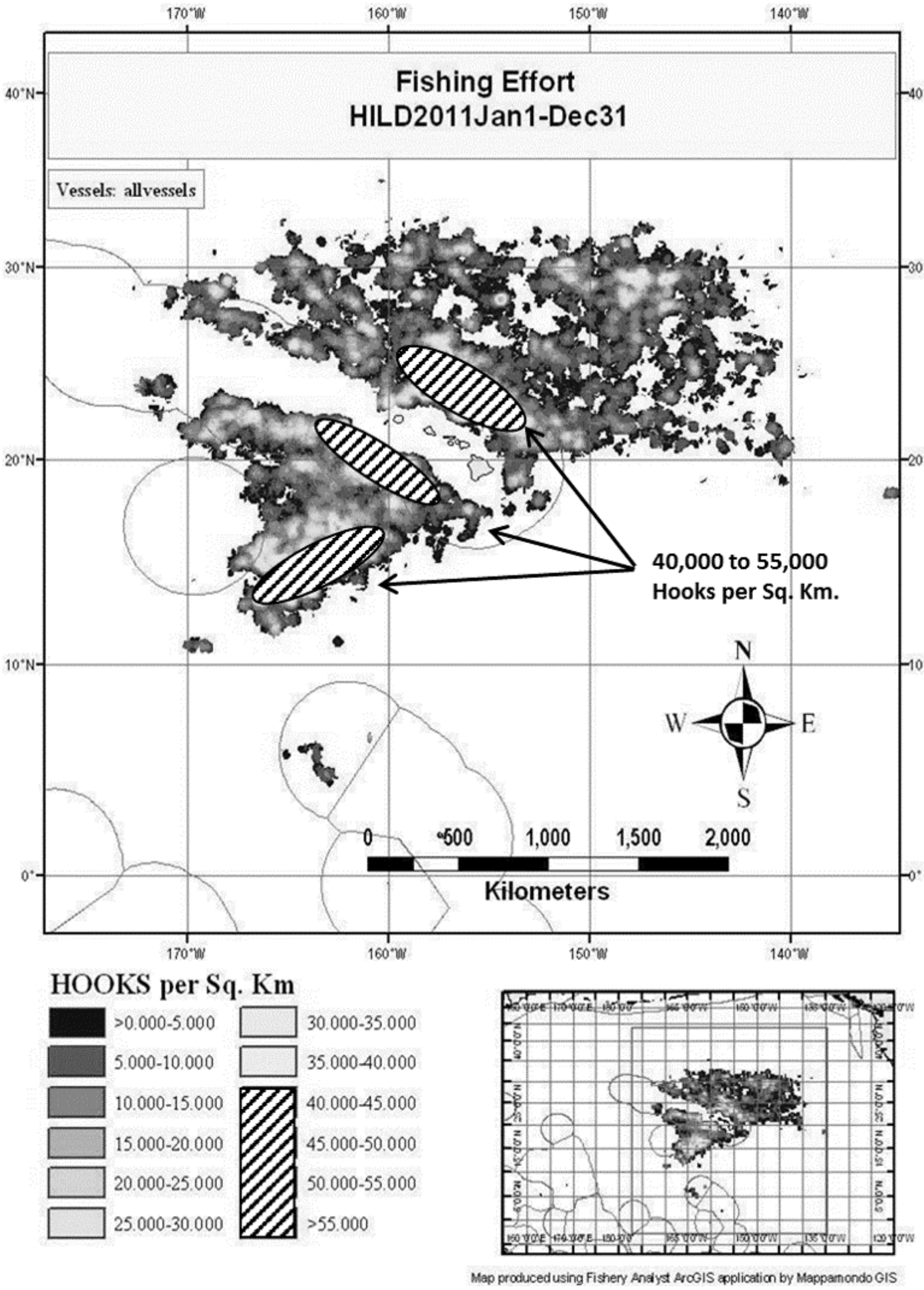
[2] The first four requirements only apply when fishing north of the equator.

4 Action Area

The action area is where the Hawaii deep-set fishing vessels operate, including transiting and fishing. This generally includes the EEZ around Hawaii and the Pacific Remote Island Areas (Baker Island, Howland Island, Jarvis Island, Johnston Atoll, Kingman Reef, Palmyra Atoll, and Wake Island) and the adjacent high seas areas (Fig. 1). The fishery operates longline gear between approximately 40-350 meters in depth. Based on historical fishing patterns from 2004-2011, the fishery may make deep-sets between 135° W and 180° W longitude and from 5° S to 40° N latitude with the majority of fishing taking place in closer proximity to the Hawaiian Archipelago. Regulations prohibit the fishery from operating in specific areas as specified in section 1.2. Fishing locations may vary seasonally based on oceanographic conditions, catch rates of target species, and management measures, among others. The deep-set fishery operates in the deep, pelagic waters around the MHI throughout the year, mostly within 300-400 nm (556-741 km) of the MHI (Fig. 1). Some fishing occurs in the U.S. EEZ around Johnston Atoll and occasionally the Pacific Remote Island Areas of Kingman Reef and Palmyra Atoll (5° N. lat.). In general, deep-set longline vessels operate out of Hawaii ports, with the vast majority based in Honolulu and a few in Hilo. Infrequently, deep-set trips originate from other ports such as Long Beach or San Francisco, California, or Pago Pago, American Samoa, and then fishermen land their catches in Hawaii. Fishermen departing from California begin fishing on the high seas, outside of the U.S. EEZ. Fishermen departing from American Samoa usually begin fishing near the Equator or farther north in the North Pacific where they expect higher catch rates of bigeye tuna.



Map produced using Fishery Analyst ArcGIS application by Mappamondo GIS

Figure 1. Spatial distribution of fishing effort by the Hawaii longline deep-set fishery, 2011. Source: NMFS PIFSC unpublished.

There is spatial overlap between the Hawaii-based shallow-set and deep-set longline fisheries. The proposed action addressed by this BiOp is the continued operation of the deep-set fishery only. The shallow-set fishery operates almost entirely north of Hawaii (in some years, depending on seawater temperature, this fishery may operate mostly north of 30° N). The deep-set fishery operates primarily to the south of Hawaii between the Equator and 35° N, but in some years there may also be considerable fishing north of Hawaii. Thus, the fisheries overlap spatially near Hawaii between 20°N and 30°N (NMFS 2012a).

5 Status of Listed Species

SFD's June 5, 2013 BE determined that the proposed action may adversely affect the eight ESA-listed marine species shown in Table 1b. SFD also requested a conference opinion on the Eastern and Indo-West Pacific scalloped hammerheads DPSs, which are now listed. The BE further determined that the seven species shown in Table 1a were not likely to be adversely affected. Critical Habitat will also be addressed in section 5.1.

Table 1. ESA-listed marine species found in the action area.

Species	Scientific Name	ESA Status	Listing Date	Federal Register Reference
Table 1a. Species not likely to be adversely affected by the proposed action.				
Hawaiian Monk Seal	*Monachus schauinslandi*	Endangered	11/23/1976	41 FR 51612
Blue Whale	*Balaenoptera musculus*	Endangered	12/02/1970	35 FR 18319
Fin Whale	*B. physalus*	Endangered	12/02/1970	35 FR 18319
Sei Whale	*B. borealis*	Endangered	12/02/1970	35 FR 18319
N. Pacific Right Whale	*Eubalaena japonica*	Endangered	12/27/2006	71 FR 77694
Hawksbill Sea Turtle	*Eretmochelys imbricata*	Endangered	7/28/1978	43 FR 32800
Scalloped Hammerhead Eastern Pacific DPS	*Sphyrna lewini*	Endangered	7/03/2014	79 FR 38214
Table 1b. Species likely to be adversely affected by the proposed action.				
Humpback Whale	*Megaptera novaeangliae*	Endangered	12/02/1970	35 FR 18319
Sperm Whale	*Physeter macrocephalus*	Endangered	12/02/1970	35 FR 18319
MHI insular false killer whale DPS	*Pseudorca crassidens*	Endangered	11/28/2012	77 FR 70915
North Pacific DPS Loggerhead Sea Turtle	*Caretta caretta*	Endangered	9/22/2011	76 FR 58868
Leatherback Sea Turtle	*Dermochelys coriacea*	Endangered	06/02/1970	35 FR 8491
Olive Ridley Sea Turtle	*Lepidochelys olivacea*			
Nesting aggregations Pacific coast of Mexico		Endangered	7/28/1978	43 FR 32800
All other Olive Ridley turtles		Threatened	7/28/1978	43 FR 32800
Green Sea Turtle	*Chelonia mydas*		7/28/1978	43 FR 32800
Nesting aggregations, Pacific coast Mexico, and Florida		Endangered	7/28/1978	43 FR 32800
All other Green turtles		Threatened	7/28/1978	43 FR 32800
Scalloped hammerhead Indo-West Pacific DPS	*Sphyrna lewini*	Threatened	7/03/2014	79 FR 38214

5.1 Species and Critical Habitat Not Likely to be Adversely Affected

In order to determine that a proposed action is not likely to adversely affect listed species, NMFS must find that the effects of the proposed action are expected to be insignificant, discountable, or beneficial as defined in the joint USFWS-NMFS Endangered Species Consultation Handbook: (1) insignificant effects relate to the size of the impact and should never reach the scale where take occurs; (2) discountable effects are those that are extremely unlikely to occur; and (3) beneficial effects are positive effects without any adverse effects (USFWS and NMFS 1998). This standard, as well as consideration of the probable duration, frequency, and severity of potential interactions, was applied during the analysis of effects of the proposed action on ESA-listed marine species to determine if and which species are "not likely to be adversely affected".

5.1.1 Hawksbill Sea turtles

Hawksbill life history is characterized by early development in the pelagic zone followed by later development in nearshore habitats. Hawksbills recruit to coastal habitats of the neritic zone at greater than 40 cm carapace length. Sub-adults and adults forage on coral reefs, primarily on sponges. Detailed information about the biology, habitat, and conservation status of the hawksbill sea turtle can be found in its recovery plan and other sources at http://www.nmfs.noaa.gov/pr/species/turtles/.

Like other sea turtles, if large enough, hawksbill sea turtles can potentially be hooked or entangled by longline gear if they come in contact with it. However, unlike other species of sea turtles, the density of hawksbills is extremely low in the action area. In Hawaii, tracking of post-nesting hawksbills from the Big Island of Hawaii and Maui suggest that primary adult foraging habitat occurs along the Hamakua coast of the Big Island (Parker et al. 2009), and adults are very rarely expected to enter waters fished by Hawaii-based longline vessels (approximately 75 nautical miles from the MHI). There has been no recorded bycatch of hawksbill sea turtles in Pacific U.S. longline fisheries. In the entire central North Pacific, only a few dozen females are thought to nest annually, thus it appears that there is low hawksbill sea turtle abundance with very few pelagic juveniles foraging in the action area. Based on very low densities of hawksbill sea turtles in the action area, and the lack of any interactions between the Hawaii-based shallow-set and deep-set fisheries since observer coverage began in 1994 we consider the probability of hawksbill interactions with gear of the proposed action extremely unlikely, and discountable.

NMFS also considered the potential for vessel collisions with hawksbill sea turtles that could be associated with the proposed action. Approximately 128 vessels participate in the fishery each year, thus the fishery makes up well under one percent of the total vessel traffic in the MHI. In addition there are very low densities of hawksbill turtles around the MHI. Thus, we consider it discountable that this species will be struck by vessels associated with the proposed action.

Based on the above finding that the effects of the proposed action are expected to be insignificant, discountable, or beneficial as defined in the joint USFWS-NMFS Endangered Species Consultation Handbook (USFWS and NMFS 1998), hawksbill sea turtles are not likely to be adversely affected by the proposed action.

5.1.2 Cetaceans

Blue, fin, sei, and North Pacific right whales can potentially be hooked or entangled by longline gear. However, their densities are extremely low in the action area. The only ESA-listed cetacean species that have been observed to be hooked or entangled by this fishery are the humpback, sperm, and MHI Insular false killer whales, which are addressed further in this document. Based on the very low densities of blue, fin, sei, and North Pacific right whales in the action area, and the lack of any interactions between the deep-set fishery and these four species since the observer program started collecting data in the 1994, we consider it discountable that these four species will be hooked or entangled by longline gear under the proposed action.

NMFS also considered the potential for vessel collisions with blue, fin, sei, and North Pacific right whales that could be associated with the proposed action. Approximately 128 vessels participate in the fishery each year, thus the fishery makes up well under one percent of the total vessel traffic in the main islands. In addition there are very low densities of these cetaceans around the main Hawaiian Islands. Thus, we consider it discountable that these four species will be struck by vessels associated with the proposed action.

Based on the above finding that the effects of the proposed action are expected to be insignificant, discountable, or beneficial as defined in the joint USFWS-NMFS Endangered Species Consultation Handbook (USFWS and NMFS 1998), blue, fin, sei and North Pacific right whales are not likely to be adversely affected by the proposed action.

5.1.3 Hawaiian Monk Seal

Some evidence in the early 1990s suggests that longline operations may have been interacting with Hawaiian monk seals, as indicated by the sighting of a few animals with hooks and other injuries. In 1991, NMFS established a permanent 50-mile protected species zone around the NWHI that is closed to longline fishing (56 FR 52214, October 14, 1991). Since 1993, no interactions with Hawaiian monk seals in the Hawaii longline fisheries have been reported (WPFMC 2009b). In addition, observers have not observed any interactions since coverage of the longline fisheries began in 1994. Although monk seals commonly travel more than 43 nm offshore, they are unlikely to encounter longline gear, because movements of this distance are more likely to be from one foraging area (island, reef, or seamount) to another, rather than into pelagic areas (NMFS 2007). Based on the very low likelihood of monk seals traveling more than 43 nm into the action area, especially where fishing operations typically occur farther offshore, we consider it discountable that this species will be hooked or entangled by longline gear under the proposed action.

There is currently no evidence that vessel collisions associated with the proposed action pose potential adverse effects to Hawaiian monk seals. In spite of the increasing abundance of monk seals in the MHI, and the large amount of vessel traffic, only two seals with possible vessel collision injuries have ever been found in the MHI (NMFS 2007). Approximately 128 vessels actively fish in the deep-set fishery each year, thus the fishery is a small fraction of the total boat traffic in the MHI. Thus, we consider it discountable that this species will be struck by vessels associated with the proposed action.

Based on the above finding that the effects of the proposed action are expected to be insignificant, discountable, or beneficial as defined in the joint USFWS-NMFS Endangered Species Consultation Handbook (USFWS and NMFS 1998), Hawaiian monk seals are not likely to be adversely affected by the proposed action.

5.1.4 Eastern Pacific Scalloped Hammerhead DPS

Scalloped hammerheads have been incidentally hooked in the Hawaii deep-set fishery at very low levels. Based on the known range and likely occurrence for the Eastern Pacific DPSs, it is unlikely that this DPS occurs in the action area of the deep-set fishery. While the DPS boundaries overlap with the action area for the deep-set fishery there has never been an observed interaction with a scalloped hammerhead shark in this area. This is likely due to the species more coastal nature and its lack of long distance migrations across deep pelagic waters. The DPS boundary was delineated in accordance with RFMO boundary lines so that any conservation measures passed by the RFMO (in this case, the IATTC) would be applicapable to the entire DPS and not the known ranges of scalloped hammerheads for the longitudinal separation of the DPSs[3] (NMFS 2014d). From 2004-2014, observers in the deep-set fishery observed three scalloped hammerhead sharks caught incidentally in the area of the threatened Indo-West Pacific DPS south of 10° N (NMFS 2014d, NMFS unpublished observer data). There were also interactions in the range of the Central Pacific DPS, which was not listed. Additional records of catch at similarly low levels exist prior to 2004 (NMFS 2014d). NMFS has no information that longline gear under the proposed action is likely to interact with scalloped hammerheads from the endangered Eastern Pacific DPS due to their behavior and biology and therefore we consider it discountable that the Eastern Pacific DPS will be hooked or entangled by longline gear under the proposed action.

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[3] From the 10° N. lat., the boundary lines were primarily a consequence of the Central Pacific DPS delineation, in order to encompass all open ocean areas (and , hence, extending to the border of the Central Pacific DPS boundary line) (NMFS 2014d).

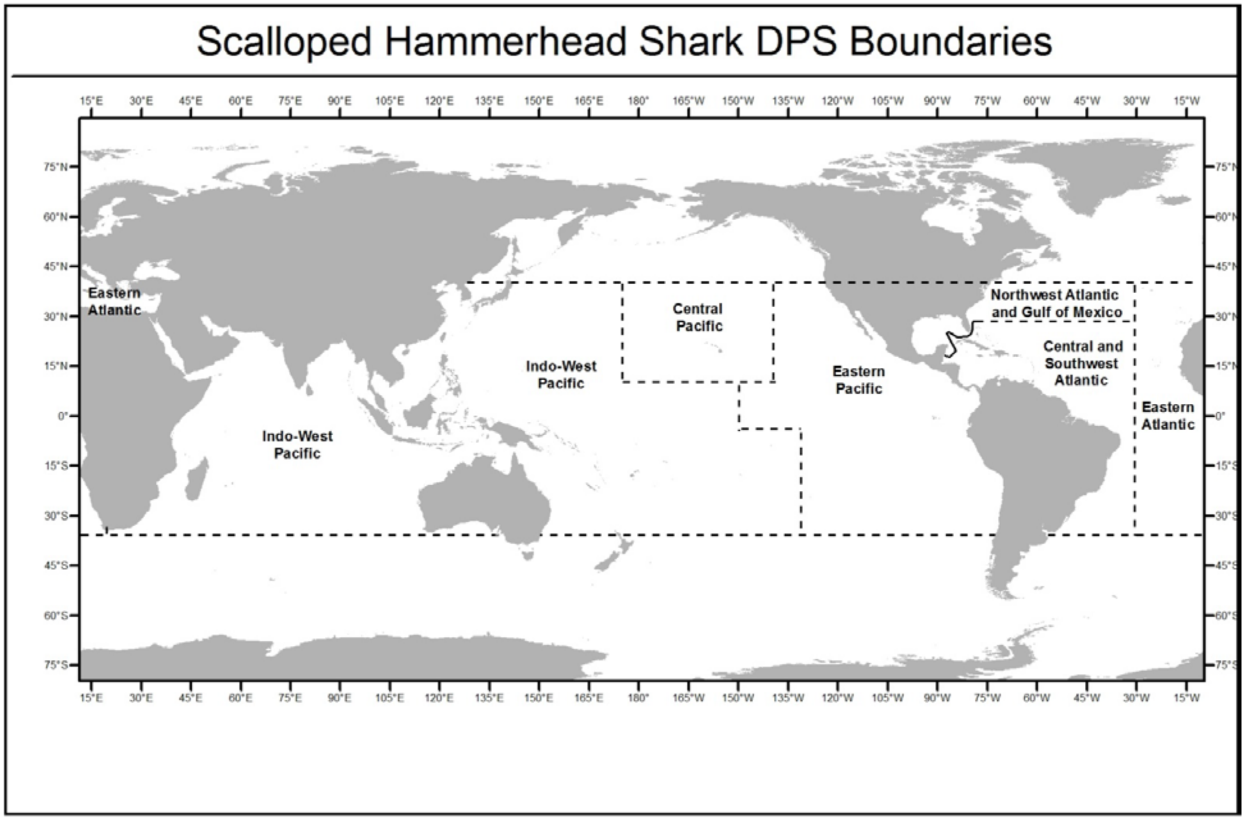

Figure 2. Map of Scalloped hammerhead DPS boundaries (79 FR 38214; July 3, 2014).

Based on the above finding that the effects of the proposed action are expected to be insignificant, discountable, or beneficial as defined in the joint USFWS-NMFS Endangered Species Consultation Handbook (USFWS and NMFS 1998), the Eastern Pacific DPS of scalloped hammerhead sharks are not likely to be adversely affected by the proposed action.

5.1.5 Critical Habitat

Federal agencies are directed under section 7(a)(2) to ensure that any activities or programs they authorize, fund, or carry out do not destroy or adversely modify critical habitat. Destruction or adverse modification of critical habitat means a direct or indirect alternation that appreciably diminishes the conservation value of critical habitat of a listed species. Such alterations include, but are not limited to, alterations adversely modifying any of those physical or biological features that were the basis for determining the habitat to be critical. Hawaiian monk seal critical habitat has been proposed in the Central Pacific portion of the action area and leatherback sea turtle critical habitat has been designated off the coast of California, where deep-set fishing vessels may transit. These are the only critical habitat designations and proposed designations that occur in the action area. The impacts to critical habitat are analyzed in the following sections.

Hawaiian Monk Seal

In 1988, NMFS expanded existing designated critical habitat for the Hawaiian monk seal to include nearly all beach areas and nearshore waters of the Northwestern Hawaiian Islands. In June 2011, NMFS proposed to revise critical habitat for monk seals to generally include terrestrial and marine habitat in the MHI from 5 m inland of the shoreline seaward to the 500 meter depth contour, and by extending the current designation in the NWHI out to the 500 m depth contour (76 FR 32026, June 2, 2011). In June 2012, NMFS announced a 6-month extension of the deadline for a final determination in order to address substantial disagreement raised during the public comment period over the data and analyses used to support the proposal (77 FR 37867, June 25, 2012). As of the date of this opinion, a final revision of critical habitat has not been released.

The action area for the proposed action does not overlap with designated critical habitat for Hawaiian monk seals but it does overlap with proposed critical habitat. The May 1988 designation includes waters from shore to a depth of 60 ft (36.6 m; 53 FR 18988, May 26, 1988). Longline vessels on deep-set trips may transit through the NWHI protected species zone, a corridor with radii of 50 nm within certain islands and reefs in the NWHI implemented to protect Hawaiian monk seals and other protected species. NMFS anticipates vessels would transit in waters deeper than the designated critical habitat. There is no area in the NWHI protected species zone in which transiting through the zone would require traveling through water 60 ft in depth or shallower.

The 2011 proposed designation identifies two physical and biological features that are essential to the conservation of the species and that may be present in the offshore environment of the proposed action. First, the proposed designation identifies areas from zero to 500 m in depth that are preferred by juvenile and adult monk seals for foraging. Second, the proposed designation identifies those marine areas with adequate prey quantity and quality to support growth and recruitment, as evidenced by the foraging habits of seals. As indicated above, although longline vessels on deep-set trips may transit through the NWHI protected species zone and MHI longline prohibited area, they are prohibited from fishing in either zone and their typical fishing grounds are generally in far deeper waters of the EEZ and high seas. Therefore, the proposed action is not likely to adversely affect designated or proposed critical habitat for the Hawaiian monk seal. Monk seal designated and proposed critical habitat is not discussed further in this biological opinion.

Leatherback Sea Turtles

On January 26, 2012, critical habitat was designated for leatherback sea turtles off the west coast of the U.S. including areas off of Washington, Oregon, and California (77 FR 4170). All of the designated area is contained within the EEZ off the U.S. west coast. Off California, where vessels on deep-set trips may occasionally transit, designated critical habitat extends for approximately 80 nm miles or less from shore from Point Arena to Point Arguello. Longline vessels would usually travel nearly directly offshore to reach the closest available fishing grounds outside of the EEZ or to transit back to Hawaii. The primary constituent element of the critical habitat is the quantity and quality of leatherback prey, which is primarily scyphomedusae of the order Semaeostomeae (e.g., Chrysaora, Aurelia, Phacellophora, and Cyanea), of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development of leatherbacks (NMFS 2011a). Activities that are listed in the designation that may impact the primary constituent element includes point source pollution, pesticide application, oil spill response, power plants, desalination plants, wave and wind energy projects, and Liquid Nitrogen Gas projects. As described previously, the deep-set fishery is prohibited from fishing in the EEZ of the U.S. west coast and therefore would not be conducting any fishing operations inside the designated critical habitat. Given the limited number of vessels that participate in the fishery and far fewer that travel to California ports adjacent to designated critical habitat, we do not anticipate that the vessels will adversely modify the primary constituent element, which is the quantity and quality of the prey. These vessels would have an insignificant effect on the level of pollution occurring in the designated area since they are required to follow Marpol (marine pollution) regulations, and the amount of oil and other pollutants carried by individual vessels is extremely small. Additionally, no leatherback prey (e.g., jellyfish) would be fished or removed from the habitat. No aspects of the proposed action would be expected to affect the condition, distribution, diversity, abundance and density of leatherback prey. Therefore we conclude that the proposed action is not likely to adversely affect the designated critical habitat.

5.2 **Species Likely to be Adversely Affected**

This section presents the biological and ecological information relevant to formulating the agency's biological opinion, including population characteristics (population structure, size, trends) for the populations affected by the proposed action, life history characteristics (especially those affecting vulnerability to the proposed action), threats to the species, major conservation efforts, and other relevant information (USFWS and NMFS 1998). Factors affecting the species within the action area are described in more detail in the Environmental Baseline section. The status of the species is first summarized below, followed by more detailed descriptions for each of the eight species addressed by this biological opinion.

The eight species addressed by this biological opinion have global distributions and all but three are listed globally at the species level (Table 1). Under the ESA, a sub-species or a DPS can also be listed (see ESA Section 7 Implementing Regulations; 50 CFR 402)[4].

However, as shown above in Figure 1, the action area is relatively small compared to the distributions of the eight species. Since the proposed action can only affect populations of these species that occur within the action area, this opinion will focus on the affected populations, then relate the effects on the affected populations to the listed species. In the case of marine mammals the affected populations will correspond to stocks as defined in the NMFS stock assessment reports (SARs). In the absence of DPSs or other formally-recognized populations for five of the eight species, affected populations must first be identified. For the purposes of this opinion, the eight species addressed by this biological opinion occur in the Pacific Ocean as the following:

1. Humpback whales: North Pacific population. NMFS has identified three overlapping stocks in the North Pacific (see humpback whale Stock Assessment Reports), and due to the complexity of these overlapping stock aggregations it is likely that there is not just one population in the North Pacific (Calambokidis et al. 2008; Fleming and Jackson 2011). The humpback whale recovery plan (NMFS 1991) states that the central south Pacific and eastern south Pacific stocks are population sub-units in the south Pacific.

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[4] Certain nesting aggregations of olive ridley and green turtles are listed as 'endangered' while each species as a whole is listed as 'threatened' (Table 1). These nesting aggregations are treated as DPSs by NMFS and the USFWS.

2. Sperm whales: North Pacific population. NMFS has identified three stocks in the North Pacific: Hawaii; California, Oregon, and Washington; and Alaska stocks (Caretta et al. 2013).
3. Main Hawaiian Islands Insular False Killer Whale DPS. The 2012 listing describes the distinct population segment (77 FR 70915; November 28, 2012).
4. Loggerhead turtles: North Pacific DPS and South Pacific DPS. The 2011 Loggerhead DPS listing describes the two distinct population segments in the Pacific.
5. Leatherback turtles: Eastern Pacific and Western Pacific populations. The leatherback 5-year status review (NMFS and USFWS 2013) describes the status of leatherback populations in geographic areas, including the eastern Pacific and western Pacific areas within the Pacific Ocean.
6. Olive Ridley turtles: Eastern Pacific and Western Pacific populations. The olive ridley 5-year status review (NMFS and USFWS 2014) describes the status of olive ridley populations in geographic areas, including the eastern Pacific and western Pacific areas within the Pacific Ocean.
7. Green turtles: Western Pacific, Central Pacific, and Eastern Pacific populations. The green turtle 5-year status review (NMFS and USFWS 2007b) describes the status of green turtle populations in geographic areas, including the western Pacific, central Pacific, and eastern Pacific areas within the Pacific Ocean.
8. Scalloped Hammerhead Shark: Indo-west Pacific DPS, and Eastern Pacific DPS. The 2014 listing (79 FR 38214, July 3, 2014; NMFS 2014d**)** describes the distinct population segments in the Pacific.

Not all Pacific populations identified above occur within the action area (Fig. 1). All humpbacks in the action area are thought to be from the central North Pacific stock (NMFS 1991, Calambokidis et al. 2008; NMFS 2010a, Fleming and Jackson 2011). All sperm whales in the action area are thought to be from the Hawaii stock (Caretta et al. 2013). Only the Indo-West Pacific DPS of the scalloped hammerhead shark has been observed to be caught by the Hawaii deep-set fishery.

For sea turtles, genetic studies have determined the source populations of individuals, based on nesting aggregations, that interact with the Hawaii-based longline fisheries (shallow-set and deep-set; Table 2). Over 200 loggerhead samples have been analyzed from the shallow-set fishery, and all were from the North Pacific DPS. The thirteen loggerhead samples from the deep-set fishery were also from the North Pacific DPS. The 50 leatherbacks sampled from the shallow-set fishery were all from the western Pacific population. One of the 24 leatherback samples from the deep-set fishery was from the eastern Pacific population. Olive ridleys are the most common turtle species hooked or entangled in the deep-set fishery, and about three-fourths are from the eastern Pacific population. Green turtle bycatch in the shallow-set is about evenly split between the central and eastern Pacific populations and for the deep-set fishery about three-fourths are from the eastern Pacific, with the remaining from the central and western pacific (Table 2).

Table 2. Genetics results for turtle bycatch, shallow-set vs. deep-set, 1995-2014 (P. Dutton, personal communication, May 2014).

Species	**Shallow-Set**		**Deep-set**	
	Samples	**Source Pop^n (%)**	**Samples**	**Source Pop^n (%)**
Loggerhead	219	219 Japan (100%)	13	13 Japan (100%)
Leatherback	50	50 W. Pacific (100%)	24	23 W. Pacific (96%) 1 E. Pacific (4%)
Olive Ridley	14	8 E. Pacific (57%) 6 W. Pacific (43%)	106	82 E. Pacific (77%) 24 W. Pacific (23%)
Green	7	4 C. Pacific (57%) 3 E. Pacific (43%)	19	13 E. Pacific (68%) 2 W. Pacific (11%) 3 C. Pacific (16%) 1 C or E. Pacific(5%)

Table 3 shows a subset of the data from Table 2, specifically, the sea turtle interactions observed since the Hawaii-based deep-set and shallow-set longline fisheries became two distinctly managed fisheries in 2004. During this period, 4 loggerhead, 23 leatherback, 72 olive ridley, and 7 green turtle interactions were observed in the deep-set fishery. The number of genetics samples taken and analyzed, and their results, are shown in Table 3 below.

Table 3. Genetics results for turtle bycatch in deep-set fishery from 2004 -2014 (P. Dutton, personal communication, May 2014).

Species	Total caught	Genetics Samples Taken	Genetics Samples Analyzed	Source Pop (%)
Loggerhead	4	4	4	4 Japan (100%)
Leatherback	23	12	11	11 W. Pacific (100%)
Olive Ridley	72	72	61	45 E. Pacific (72%) 16 W. Pacific (28%)
Green	7	7	7	4 E. Pacific (57%) 2 W. Pacific (29%) 1 C. Pacific (14%)

For the purposes of this opinion, the affected populations of the eight species addressed by this biological opinion (humpback whale, sperm whale, MHI insular false killer whale, scalloped hammerhead shark, and four sea turtle species) are defined as follows:

1. Humpback whales: Central North Pacific population
2. Sperm whales: North Pacific population
3. False Killer whales: Main Hawaiian Islands Insular DPS
4. Loggerhead turtles: North Pacific DPS
5. Leatherback turtles: Western Pacific and Eastern Pacific populations
6. Olive Ridley turtles: Eastern Pacific and Western Pacific populations
7. Green turtles: Central Pacific, Western Pacific, and Eastern Pacific populations
8. Scalloped Hammerhead sharks: Indo-west Pacific DPS

“Affected populations” of sea turtle species are defined by interactions with the Hawaii-based deep-set fishery, as determined by the genetics results summarized in Table 2 above. The affected populations of marine mammals are determined by occurrence within the action area as defined by the MMPA stock assessment reports. The affected population of scalloped hammerhead is based on where the interactions have occurred historically and the boundary lines of the final DPS designation. The focus of this opinion is on these affected populations and DPSs.

5.2.1 Humpback Whales

Information in this section is summarized from the humpback whale recovery plan (NMFS 1991), the Humpback Whale Stock Assessment Reports (e.g., Allen and Angliss 2013), the Structure of Populations, Levels of Abundance and Status of Humpback Whales in the North Pacific report (SPLASH) (Calambokidis et al. 2008), the 2012 BiOp (NMFS 2012a), the Global Review of Humpback Whales (Fleming and Jackson 2011), and other sources cited below.

5.2.1.1 Population Characteristics

Humpback whales are distributed worldwide in all ocean basins, from subtropical to subpolar waters. Humpback whales in the North Pacific carry out seasonal migrations between warmer temperate and subtropical waters in winter for reproduction, and cooler temperate and subpolar waters of high prey productivity in summer for feeding. Breeding areas in the North Pacific are more geographically separated than the feeding areas and include regions offshore of mainland Central America; mainland Baja California, and the Revillagigedos Islands, Mexico; Hawaii; and Asia, including Ogasawara and Okinawa Islands, and the Philippines. About half of the humpback whales in the North Pacific Ocean breed and calve in the U.S. territorial waters off Hawaii, and more than half feed in U.S. territorial waters (Fleming and Jackson 2011). In the North Pacific, population structure is complex with mixing between feeding grounds and breeding grounds. Stock structure of humpback whales is defined based on feeding areas.

Within the North Pacific Ocean, at least three stocks make up the North Pacific population(s): 1) the California/Oregon/Washington and Mexico stock, consisting of winter/spring populations in coastal Central America and Mexico which migrate to California and British Columbia; 2) the CNP stock that migrates between the Hawaiian Islands and northern British Columbia/Southeast Alaska, Gulf of Alaska, and the Bering Sea/Aleutian Islands; and 3) the western North Pacific stock, consisting of winter/spring populations off Asia which migrate primarily to Russia and the Bering Sea/Aleutian Islands (Allen and Angliss 2013). Until recently, the North Pacific was considered to be one population but based on complexities observed through the SPLASH study, which analyzed genetics and photographs, it appears that there is likely more than one (Fleming and Jackson 2011).

Based on whaling statistics, before 1905 the North Pacific population(s) was estimated to be 15,000 and was reduced by whaling to approximately 1,000 before it was placed under international protection in 1965 (NMFS 1991). Protection from whaling was effective, resulting in the North Pacific population rebounding to approximately 21,000 individuals by 2010 (Fleming and Jackson 2011). About half of the population(s) winters in Hawaii (the CNP stock). The annual growth rate for the North Pacific population over the last several decades is estimated at 4.9 to 6.8 percent, depending on which area and time frame are considered (Calambokidis et al. 2008).

5.2.1.2 Life History Characteristics Affecting Vulnerability to Proposed Action

Depth preference, migration routes, and diving behavior may affect vulnerability of CNP humpback whales to Hawaii-based deep-set longline fishing. In Hawaii, humpback whales have been sighted as early in the season as October and as late as June, with most mating and calving occurring from December to April. They are generally found in water less than 600 ft (182 m) deep, and cow and calf pairs appear to prefer even shallower water. However, after arriving in Hawaiian wintering habitat, most humpback whales are unlikely to interact with the Hawaii-based deep-set longline fishery because of the MHI Longline Fishing Prohibited Area, which extends to 75 nm from shore, as described in section 1.2. While migrating between feeding grounds and Hawaii, humpback whales pass through the action area where they may be exposed to deep-set longline gear and vessels.

5.2.1.3 Threats to the Species

Whaling was formerly by far the most serious threat to the species, as described in the humpback whale recovery plan (NMFS 1991), the Stock Assessment Reports (e.g., Allen and Angliss 2013), the SPLASH report (Calambokidis et al. 2008), and the Global review of Humpbacks (Fleming and Jackson 2011). From 1900-1965, nearly 30,000 whales were taken in modern whaling operations on the Pacific Ocean. Hunting humpback whales has been prohibited since 1966. Current threats include hookings and entanglement in fishing gear, ship strikes, tourism, noise, and potentially the effects of anthropogenic climate change.

Humpback whales are likely hooked or entangled by fishing gear throughout their global range, but data are scarce outside the U.S., especially in the Pacific. Reports of entangled humpback whales found swimming, floating, or stranded with fishing gear attached have increased in recent years in both Alaskan and Hawaiian waters. For example, there was a total of 112 confirmed entanglement reports in Hawaii from 2002 to 2012, with the highest number of confirmed reports during the 2008-2009, 2009-2010, and 2011-2012 field seasons in which there were a total of 55 (Lyman 2012). The 2011-12 humpback whale season had the third greatest number of large whale entanglement reports of any season since 2002. There were 31 reports, 17 of which were confirmed and involved at least 10 different animals. Many of the whales reported entangled in Hawaiian waters most likely brought the gear with them from higher latitude feeding grounds, although there have been at least 8 that were reported to have been entangled in trap gear around the MHI (Lyman 2012). While the whales are not typically at risk from drowning or immediate death, they are at increased risk of starvation, infection, physical trauma from the gear, and ship strikes as a result of entanglement. Available evidence from entangled humpback whales indicates that while it is not possible to predict whether an animal will free itself of gear, a large proportion are believed to extricate themselves based on scarring observed among apparently healthy animals. A study in southeast Alaska on the CNP stock of humpback whales estimated that the number which showed evidence of entanglement from some point in their life and survived is about 71 percent which is a much higher number than reported disentanglements (Neilson et al. 2009).

The total of all known mortality and serious injury[5] (M&SI) to the CNP stock as a result of fishing operations for the time period from 2007 through 2011 is 46.75 whales (23 in Alaska, 24 in Hawaii), resulting in an annual average M&SI of 9.35 animals (Allen and Angliss 2013, NMFS 2014e). This M&SI estimate includes observed interactions on commercial vessels with NMFS observers and opportunistic reports from commercial and recreational fisheries. In the opportunistic reports the M&SI cannot be attributed to a specific fishery (Allen and Angliss 2013). Only 0.75 M&SI is attributed to the shallow-set longline fishery in Hawaii and zero to the deep-set fishery. The majority of the remaining entanglements observed in Hawaii originated in Alaska based on the type of gear attached to the animal.

Many humpback whales are killed by ship strikes throughout the world, including along both coasts of the U.S. On the Pacific coast, one humpback whale is killed about every other year by ship strikes. Worldwide records of vessel collisions and stranding information indicate that humpback whales are one of the more common species to have ship strikes documented (Jensen and Silber 2003, Laist et al. 2001). Humpback whales, especially calves and juveniles, are highly vulnerable to ship strikes and other interactions with non-fishing vessels. Younger whales spend more time at the surface, are less visible and closer to shore, thereby making them more susceptible to collisions. Humpback whale distribution overlaps significantly with the transit routes of large commercial vessels that ply the waters off Alaska. Records of vessel collisions with large whales in Alaska indicate that strikes have involved cruise ships, recreational cruisers, whale watching catamarans, fishing vessels, and skiffs. Vessel lengths associated with these records ranged from approximately 20 feet to over 250 feet, indicating that all types and sizes of watercraft pose a threat of collision for whales. Between 2001 and 2005, reports of vessel collisions with humpback whales indicate an average of five whales struck per year in Alaska, whereas in Hawaii three to four vessel collisions with humpback whales were reported per year in 2001 through 2006. Between 2007 and 2011, reports of vessel collisions with humpback whales indicate an average of 7.2 whales struck per year in Hawaii (Bradford and Lyman 2013).

Several other threats affect humpback whales throughout their range. For example, the CNP stock is the focus of a large whale watching industry in both Hawaii and Alaska. The growth of the whale watching industry is a concern for humpback whales since harassment may occur, preferred habitats may be abandoned, and fitness or survivability may be compromised if disturbance levels are too high. Humpback whales seem to respond to moving sound sources, such as whale-watching vessels, fishing vessels, recreational vessels, and low-flying aircraft. Their responses to noise are variable and have been correlated with the size and behavior of the whales when the noise occurs. Anthropogenic sound has increased in all oceans over the last 50 years and it is thought to have doubled each decade in some areas of the ocean over the last 30 years. Low-frequency sound comprises a significant portion of this and stems from a variety of sources including shipping, hydrographic research, naval activities, and oil and gas exploration (NMFS 2006; NMFS 2008b; Fleming and Jackson 2011).

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[5] Under the MMPA serious injury (50 CFR 229.2) is defined as "any injury that will likely result in mortality", and NMFS' interpretation in the NMFS Serious Injury Polcy is "any injury that is more likely than not to result in mortality." For purposes of this consultation NMFS assumes that all serious injuries are mortalities.

Highly mobile species, such as marine mammals, can respond to effects of climate change more rapidly than their terrestrial counterparts (Harwood 2001). No significant climate change-related impacts to humpback whale populations have been reported to date (Fleming and Jackson 2011). The most likely impact of climate change on cetaceans will be changes in range related to migration, expansion, or contraction of the geographic thermal niche populations currently occupy, or changes in the distribution of prey species with particular thermal requirements. The ranges of 88 percent of cetaceans may be affected by changes in water temperature resulting from global anthropogenic climate change, however the humpback whale is a cosmopolitan species ranging throughout the world's oceans and thermal and prey limitations related to anthropogenic climate change are unlikely to impact the range of this species (MacLeod 2009). Whilst oceanic cetaceans are unlikely to be directly affected by rises in sea level, important habitats for coastal species and species that require coastal bays and lagoons for breeding, such as humpback whales, could be adversely affected in the future (Simmonds and Elliot 2009). Humpback whales that feed in Polar Regions may also encounter reduced prey. 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. Due to a lack of scientific data, specific effects climate change will have on this species in the future are not predictable or quantifiable to any degree such as would allow for more detailed analysis in this consultation.

5.2.1.4 Conservation of the Species

To minimize the possibility of collision and the potential for harassment in Hawaii and Alaska, NMFS implemented regulations that prohibit approaching humpback whales within 100 yards (90 m) when on the water or operating an aircraft within 1,000 feet (300 m) (50 CFR 224.103). The regulations also make it unlawful to disrupt the normal behavior or prior activity of whales, which may be manifested in several specific ways that include but are not limited to interruptions to breeding, nursing, or resting activities.

The Hawaiian Islands Humpback Whale National Marine Sanctuary (HIHWNMS) also protects the winter breeding, calving and nursing range of the largest Pacific population of the endangered humpback whale. The U.S. Congress designated the HIHWNMS on November 4, 1992, and the Hawaiian Islands National Marine Sanctuary Act designated the Sanctuary for the primary purpose of protecting humpback whales and their habitat within the Hawaiian Islands marine environment. It is the only National Marine Sanctuary dedicated to a species of whale and its habitat. The Sanctuary works collaboratively to conserve, enhance and protect humpback whales and their habitat by promoting and coordinating research, enhancing public awareness, and fostering traditional uses by native Hawaiians. It is jointly managed by the sanctuary manager, the state of Hawaii co-manager, and other field staff via a cooperative federal-state partnership. The Sanctuary is a series of five noncontiguous marine protected areas distributed across the MHI. The total area of the Sanctuary is 1,370 square miles. Encompassing about half of the total Sanctuary area, the largest contiguous portion is delineated around Maui, Lanai, and Molokai. The four smaller portions are located off the north shore of Kauai, off Hawaii's Kona coast, and off the north and southeast coasts of Oahu (www.hawaiihumpbackwhale.noaa.gov).

The Hawaiian Islands Disentanglement Network is a community based network that was formed in 2002 in an attempt to free endangered humpback whales and other marine animals from life threatening entanglements and at the same time gather valuable information that will help mitigate the issue of marine debris and future entanglements (www.hawaiihumpbackwhale.noaa.gov). Since 2002, the Sanctuary has received more than 211 reports of whales entangled in gear, and all but three have been humpback whales. The network and partnering agencies have mounted over 121 (on-the-water or in-the-air) responses to these reports. To date, ten humpbacks reported entangled in Hawaii have been confirmed to have gear from Alaska; seven of these represent commercial trap gear. The mean distance traveled with this gear is at least 2150 nm. The greatest known straight line distance a whale may have carried gear is 2450 nm (between North shore of Etolin Island, approx 9nm SW of Wrangell Alaska, where the gear was lost and the island of Maui where the animal was first reported) (Lyman 2012).

5.2.2 Sperm Whales

Sperm whales have been protected from commercial harvest by the IWC since 1981, although the Japanese continued to harvest sperm whales in the North Pacific until 1988 (Reeves and Whitehead 1997). They are also protected by CITES. In the U.S., sperm whales were listed as endangered when the ESA was enacted in 1973. Given their endangered status under the ESA, they are also designated as depleted and therefore considered a strategic stock under the MMPA. Critical habitat has not been designated for sperm whales anywhere in the U.S. Information in this section is summarized from the sperm whale recovery plan (NMFS 2010c), the sperm whale Stock Assessment Reports (e.g., Carretta et al. 2014), and other sources cited below.

5.2.2.1 Population Characteristics

Reaching 60 feet in length and weighing up to 45 tons, the sperm whale is the largest of the toothed whales, and is one of the most widely distributed marine mammals worldwide, between 60° N and 70° S (Leatherwood and Reeves 1983). Sperm whales are widely distributed across the entire North Pacific and into the southern Bering Sea in summer but the majority are thought to be south of 40°N in winter (Rice 1974, 1989; Gosho et al. 1984; Miyashita et al. 1995). There is no clear understanding of the global population structure of sperm whales (Dufault et al. 1999). Recent ocean-wide genetic studies indicate low, but statistically significant, genetic diversity and no clear geographic structure, but strong differentiation between social groups (Lyrholm and Gyllensten 1998; Lyrholm et al. 1996; Lyrholm et al. 1999). However, vocal dialects indicate parent-offspring transmission that indicates differentiation in populations (Rendell et al. 2011). The IWC currently recognizes four sperm whale stocks: North Atlantic, North Pacific, northern Indian Ocean, and Southern Hemisphere (Dufault et al. 1999; Reeves and Whitehead 1997).

Summer/fall surveys in the eastern tropical Pacific (Wade and Gerrodette 1993) show that although sperm whales are widely distributed in the tropics, their relative abundance tapers off markedly westward towards the middle of the tropical Pacific (near the IWC stock boundary at 150°W) and tapers off northward towards the tip of Baja California. The Hawaiian Islands marked the center of a major nineteenth century whaling ground for sperm whales (Gilmore 1959; Townsend 1935). Since 1936, at least 28 strandings have been reported from the Hawaiian Islands (Woodward 1972, Nitta 1991, Maldini et al. 2005, NMFS PIR Marine Mammal Response Network database), including 7 since 2007. Sperm whales have also been sighted throughout the Hawaiian EEZ, including nearshore waters of the main and Northwestern Hawaiian Islands (Rice 1960, Barlow 2006, Lee 1993, Mobley et al. 2000, Shallenberger 1981). In addition, the sounds of sperm whales have been recorded throughout the year off Oahu

(Thompson and Friedl 1982). Summer/fall shipboard surveys of waters within the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands resulted in 43 sperm whale sightings in 2002 and 46 in 2010 throughout the study area (Barlow 2006, Bradford et al. 2013).

The stock identity of sperm whales in the North Pacific has been inferred from historical catch records (Bannister and Mitchell 1980) and from trends in CPUE and tag-recapture data (Ohsumi and Masaki 1977). A 1997 survey designed specifically to investigate stock structure and abundance of sperm whales in the northeastern temperate Pacific revealed no apparent hiatus in distribution between the U.S. EEZ off California and areas farther west, out to Hawaii (Barlow and Taylor 2005). Recent genetic analyses revealed significant differences in mitochondrial and nuclear DNA and in single-nucleotide polymorphisms between sperm whales sampled off the coast of California, Oregon and Washington and those sampled near Hawaii and in the eastern tropical Pacific (ETP) (Mesnick et al. 2011). These results suggest demographic independence between matrilineal groups found California, Oregon, and Washington, and those found elsewhere in the central and eastern tropical Pacific. Further, assignment tests identified male sperm whales sampled in the sub-Arctic with each of the three regions, suggesting mixing of males from potentially several populations during the summer (Mesnick et al. 2011). The NMFS SARs divide sperm whales within the Pacific U.S. EEZ into three discrete, non-contiguous stocks: 1) waters around Hawaii; 2) California, Oregon, and Washington waters; and 3) Alaskan waters.

The current sperm whale abundance is estimated to be approximately 200,000-1,500,000 worldwide (NMFS 2010c). Abundance in the Pacific is approximately 152,000-226,000 using Whitehead’s 2002 methods. A spring 1997 combined visual and acoustic line-transect survey conducted in the eastern temperate North Pacific resulted in estimates of 26,300 (CV=0.81) sperm whales based on visual sightings, and 32,100 (CV=0.36) based on acoustic detections and visual group size estimates (Barlow and Taylor 2005). Sperm whales appear to be a good candidate for acoustic surveys due to the increased range of detection; however, visual estimates of group size are still required (Barlow and Taylor 2005). In the eastern tropical Pacific, the abundance of sperm whales has been estimated as 22,700 (95 percent C.I.=14,800-34,600; Wade and Gerrodette 1993). However, it is not known whether any or all of these animals routinely enter the U.S. EEZ around the Hawaiian Islands.

The Hawaii stock includes animals found both within the U.S. EEZ around the Hawaiian Islands and in adjacent high seas waters; however, because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of the Hawaii stock is evaluated based on data from U.S. EEZ waters of the Hawaiian Islands (NMFS 2005b). Therefore the abundance estimate is a minimum since it does not take into account sperm whales on the adjacent high seas. A 2002 shipboard line-transect survey of the entire Hawaiian Islands EEZ resulted in an abundance estimate of 6,919 (CV=0.81) sperm whales (Barlow 2006). The recent 2010 shipboard line-transect survey of the Hawaiian Islands EEZ resulted in an abundance estimate of 3,354 (CV = 0.34) sperm whales (Bradford et al. 2013), including a correction factor for missed diving animals. This is currently the best available abundance estimate for this stock. Following Barlow et al. (1995), the minimum population size is taken to be the log-normal 20th percentile of the 2010 abundance estimate (2,539 sperm whales) within the Hawaiian Islands EEZ. The broad and overlapping confidence intervals around the 2002 and 2010 estimates preclude assessment of trend with the available data. The PBR level for the Hawaiian stock of sperm whales is calculated as the minimum population size (2,539) within the U.S. EEZ of the Hawaiian Islands times one half the default maximum net growth rate for cetaceans (½ of 4 percent) times a recovery factor of 0.2 (for an endangered species with Nmin > 1,500 and CV Nmin > 0.50, with low vulnerability to extinction (Taylor et al. 2003), resulting in a PBR of 10.2 sperm whales per year (Carretta et al. 2014).

Barlow and Taylor (2001) estimated 1,407 (CV=0.39) sperm whales in California, Oregon, and Washington waters during summer/fall based on pooled 1993 and 1996 ship line transect surveys within 300 nmi of the coast, while Barlow and Forney (2007) estimated 2,593 (CV=0.30) sperm whales from a survey of the same area in 2001. A 2005 survey of this area resulted in an abundance estimate of 3,140 (CV=0.40) whales, which is corrected for diving animals not seen during surveys (Forney 2007). The most recent ship survey of the same area in 2008 resulted in an estimate of only 300 (CV=0.51) sperm whales (Barlow 2010). This estimate is lower than all previous estimated within this region and may be due to interannual variability of sperm whale distribution in this region. The most recent estimate of abundance for this stock is the geometric mean of the 2005 and 2008 summer/autumn ship survey estimates, or 971 (CV=0.31) sperm whales.

The minimum population estimates for sperm whale stocks within the U.S. EEZ are given below. There is no recent population estimate available for sperm whales in Alaska. The most recent data available is at least five years old, but suggests a population in the tens of thousands (Allen and Angliss 2012).

Table 4. Minimum population estimates and trends for sperm whales.

Stock	Minimum population estimate	Population trend
North Pacific[1]	Unknown	No trend information
Hawaii Stock[2]	2,539	No trend information
North Atlantic[3]	3,539	No trend information
Northern Gulf of Mexico[3]	1,409	No trend information
CA/OR/WA[2]	751	unknown

[1] Allen and Angliss 2012
[2] Carretta et al. 2014
[3] Waring et al. 2012

5.2.2.2 Life History Characteristics Affecting Vulnerability to Proposed Action

Depth preference and diving behavior may affect vulnerability of Hawaii sperm whales to Hawaii-based deep-set longline fishing. Sperm whales have been sighted throughout the Hawaiian Islands EEZ. In addition, the sounds of sperm whales have been recorded throughout the year off Oahu (Carretta et al. 2014, Thompson and Friedl 1982). Sperm whales have a strong preference for waters deeper than 1,000 m (Reeves and Whitehead 1997,), although Berzin (1971) reported that they are restricted to waters deeper than 300 m. While deep water is their typical habitat, sperm whales are rarely found in waters less than 300 m in depth (Rice 1989). They feed primarily on large and medium-sized squid, the list of documented food items is fairly long and diverse. Prey items include other cephalopods, such as octopi, and medium- and large-sized demersal fishes, such as rays, sharks, and many teleosts (Berzin 1972, , Rice 1989). The diet of large males in some areas, especially in high northern latitudes, is dominated by fish (Rice 1989). In Alaska, sperm whales are known to take sablefish off of longline gear during haulback but this has not been observed in pelagic longline fisheries to date.

5.2.2.3 Threats to the Species

Whaling was formerly by far the most serious threat to the species, as described in the sperm whale recovery plan (NMFS 2010c), and the Stock Assessment Reports (e.g., Carretta et al. 2014). From 1947-1987, nearly 258,000 whales were taken in modern whaling operations in the North Pacific Ocean. Hunting sperm whales in the North Pacific has been prohibited since 1988. Current threats include vessel disturbance, climate change, and illegal whaling or resumed legal whaling.

The vulnerability of sperm whales to incidental entanglement in fishing gear especially gillnets set in deep water for pelagic fish is well documented (Di Natale and Notarbartolo di Sciara 1994, Haase and Felix 1994, Felix et al. 1997 in NMFS 2013a). Sperm whales may become entangled in fishing gear while attempting to take fish off of the gear (most often demersal long-line gear) (Angliss and Outlaw 2007). Observational studies cannot fully evaluate the potential for entanglement because many entangled animals may die at sea and thus not be seen or reported. Although instances of stomach obstruction caused by marine debris have been documented in sperm whales it is not believed to be a major threat to the species, but the effect of entanglement and ingestion is unknown (NMFS 2010c). In U.S. waters in the Pacific Ocean, sperm whales are known to have been incidentally captured only in drift gillnet operations, which killed or seriously injured an average of 9 sperm whales per year from 1991 - 1995 (Barlow et al. 1997). Interactions between longline fisheries and sperm whales in the Gulf of Alaska have been reported over the past decade (Hill and Demaster 1998, Rice 1989). Observers aboard Alaskan sablefish and halibut longline vessels have documented sperm whales feeding on fish caught in longline gear in the Gulf of Alaska. During 1997, the first entanglement of a sperm whale in Alaska's longline fishery was recorded, although the animal was not seriously injured (Hill and Demaster 1998). The available evidence does not indicate sperm whales are being killed or seriously injured as a result of these interactions, although the nature and extent of interactions between sperm whales and long-line gear is not yet clear. An individual was caught and released from gillnetting, although injured, on Georges Bank during 1990. A second individual was freed, but injured, from gillnetting on George's Bank in 1995. In 1994, a sperm whale was disentangled from gillnet along the coast of Maine. In August 1993, a dead sperm whale, with longline gear wound tightly around the jaw, was found floating ~32 km off Maine. Ten sperm whale entanglements have been observed in U.S. fisheries since 1990 in the Pacific (Carretta and Enriquez 2012). Two additional whales have been found to die from ingestion of fishing gear (Jacobsen et al. 2010). Overall, it is estimated that 3.8 sperm whales die annually along the U.S. west coast due to fisheries interaction (Carretta et al. 2014).

Information on fishery-related mortality of sperm whales in Hawaiian waters is limited, but the gear types used in Hawaiian fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. One stranded sperm whale was found in 1990 with fishing line and netting in its stomach, though it is unclear whether the gear caused its death, and what fishery the gear came from (Bradford and Lyman 2013). In addition to the interactions reported by the Hawaii longline observer program, there has been one confirmed entanglement in Hawaiian waters between 2003 and 2012. The sperm whale was observed near Kauai in 2010 with a ball of red line and netting near the mouth region. The whale swam backwards and was able to shed the gear; the interaction was subsequently determined to not be a serious injury (Bradford and Lyman 2013). Before that interaction, the only other sperm whale in the database was the stranded sperm whale in 1990. The total of all known serious injury and mortalities to the Hawaii stock as a result of commercial fishing operations for the time period from 2007 through 2011 is three whales, resulting in an annual M&SI of 0.7 animals. The estimate is based on one observed interaction in the deep-set longline fishery. There were no interactions with sperm whales in the shallow-set fishery during this time frame or since the fishery re-opened in 2004.

Sperm whales are vulnerable to ship strikes as they raft on the surface after long dives. Quantifying the effects of ship-strikes off Hawaii is not possible at this time. There have not been any recent documented ship strikes involving sperm whales in the eastern North Pacific, although there are a few records of ship strikes in the 1990s. Two whales described as “possibly sperm whales” are known to have died in U.S. Pacific waters in 1990 after being struck by vessels (Barlow et al. 1997). There is an anecdotal record from 1997 of a fishing vessel that struck a sperm whale in southern Prince William Sound in Alaska, although the whale did not appear to be injured (Laist et al. 2001). More recently in the Pacific, two sperm whales were struck by a ship in 2005, but it is not known if these ship strikes resulted in injury or mortality (NMFS 2009). There are no reported vessel collisions with sperm whales in the database maintained by the NMFS Pacific Islands Region Marine Mammal Response Network. The lack of recent evidence should not lead to the assumption that no mortality or injury from collisions with vessels occurs as carcasses that do not drift ashore may go unreported, and those that do strand may show no obvious signs of having been struck by a ship (NMFS 2009). Worldwide, sperm whales are known to have been struck 17 times out of a total record of 292 strikes of all large whales, 13 of which resulted in mortality (Jensen and Silber 2003; Laist et al. 2001). Given the current number of reported cases of injury and mortality, it does not appear that ship strikes are a significant threat to sperm whales). Harassment from whale watch boats off Hawaii is unlikely since sperm whales are not a species that are reliably seen in a given area and therefore sightings from whale watch boats are largely opportunistic.

Sperm whales are vulnerable to marine pollution. A dramatic increase in the rate of sperm whale strandings in Western Europe since the early 1980’s has raised concern that pollution might be implicated. Although tissues were analyzed for a wide range of contaminants and detailed pathological examinations were carried out, no clear link between contamination and stranding was found (Jaques and Lambertsen 1997). However, levels of mercury, cadmium, and certain organochloines in the whale’s tissues, were high enough to cause concern about toxicity and other effects (Bouquegneau et al. 1997). Levels of contaminants in sperm whales killed off northwestern Spain indicated that the levels in females were consistently higher than those in males, a finding contrary to the usual situation in cetaceans (Aguilar 1983).

The bottom-feeding habit of sperm whales, which might involve a suction mechanism, means they often ingest marine debris (Lambertson 1997). Ingestion of marine debris can have fatal consequences even for large whales. One out of 32 sperm whales examined for pathology in Iceland had a lethal disease thought to have been caused by the complete obstruction of the gut with plastic marine debris (Lambertsen 1990). In 1989, a stranded sperm whale along the Mediterranean was found to have died from ingesting plastic that blocked its' digestive tract (Viale et al. 1992). The stomach contents of two sperm whales that stranded separately in California included extensive amounts of discarded fishing netting (NMFS 2009). A fifth individual from the Pacific was found to contain nylon netting in its stomach when it washed ashore in 2004 (NMFS 2009). In March 2012, a sperm whale stranded dead in Spain, apparently dying as a result of plastic ingestion (de Stephanis et al. 2013).

Noise may disrupt sperm whale communication, navigational ability, and social patterns. Both anthropogenic and natural sound may cause interference with these functions. The effects of noise are difficult to ascertain and research on this topic are ongoing. The severity of ship noise as a threat to the recovery of sperm whales is unknown. There have been no reported seismic-related or industry ship related deaths or injuries to sperm whales where marine mammals are present, such as the Gulf of Mexico. However, it is uncertain what threat oil and gas exploration may be to the recovery of the sperm whale population. There is currently no evidence of long-term changes in behavior or distribution as a result of occasional exposure to pulsed acoustic stimuli.

There is some evidence of disruptions of clicking and behavior from sonar (Goold 1999, Watkins and Scheville 1975, Watkins et al. 1985), pingers (Watkins and Scheville 1975), the Heard Island Feasibility Test (Bowles et al. 1994), and the Acoustic Thermometry of Ocean Climate (Costa et al. 1998). Sperm whales have been observed to frequently stop echolocating in the presence of underwater pulses made by echosounders (Watkins and Scheville 1975). Goold (1999) reported six sperm whales that were driven through a narrow channel using ship noise, echosounder, and fishfinder emissions from a flotilla of 10 vessels. Watkins and Scheville (1975) showed that sperm whales interrupted click production in response to pinger (6 to 13kHz) sounds. They also stopped vocalizing for brief periods when codas were being produced by other individuals, perhaps because they can hear better when not vocalizing themselves (Goold and Jones 1995).

Highly mobile species, such as marine mammals, can respond to effects of climate change more rapidly than their terrestrial counterparts (Harwood 2001). No significant climate change-related impacts to sperm whale populations have been reported to date (NMFS 2010c). The most likely impact of climate change on sperm whales will be changes in the distribution and abundance of prey species with particular thermal requirements. There is some evidence from Pacific equatorial waters that sperm whale feeding success and, in turn, calf production rates are negatively affected by increases in sea surface temperature. This could mean that global climate change will reduce the productivity of at least some sperm whale populations (NMFS 2010c). The feeding range of sperm whales is likely one of the greatest of any species on earth, and consequently, it is likely that the sperm whales may be more resilient to climate change, should it affect prey, than a species with a narrower range.

5.2.2.4 Conservation of the Species

The biggest conservation effort for the species has been the end of commercial whaling which is estimated to have taken at least 436,000 between 1800-1987, with approximately 33,842 taken from the North Pacific from the longitude of Hawaii to the U.S. West coast (Carretta et al. 2014).

The Hawaiian Islands Disentanglement Network is a community based network that was formed in 2002 in an attempt to free endangered humpback whales and other marine animals from life threatening entanglements and at the same time gather valuable information that will help mitigate the issue of marine debris and future entanglements (www.hawaiihumpbackwhale.noaa.gov). From 2003-2012, there was one confirmed entanglement with a sperm whale, which was able to shed the gear on its own.

5.2.3 Main Hawaiian Islands Insular False Killer Whales

Information in this section is summarized from the Status Review of Hawaiian Insular False Killer Whales under the ESA (Oleson et al. 2010), the Final Listing Rule (NMFS 2012c), the Final Take Reduction Plan (NMFS 2012d), the Hawaiian Islands False killer Whale Stock Assessment Reports (e.g., Carretta et al. 2014), the negligible impact determination (NMFS 2014e), and other sources cited below.

5.2.3.1 Population Characteristics

False killer whales are found worldwide mainly in tropical and warm temperate waters (Stacey et al. 1994). Genetic, photo-identification, and telemetry studies indicate there are three demographically-independent populations of false killer whales in Hawaiian waters. Genetic analyses indicate restricted gene flow between false killer whales sampled near the MHI, NWHI, and in pelagic waters of the Eastern (ENP) and CNP (Chivers et al. 2007, 2010; Martien et al. 2011). Chivers et al. (2010) expanded previous analyses with additional samples and analysis of 8 nuclear DNA (nDNA) microsatellites, revealing strong phylogeographic patterns consistent with local evolution of haplotypes nearly unique to false killer whales occurring nearshore within the Hawaiian Archipelago. Analysis of 21 additional samples collected during a 2010 shipboard survey in Hawaiian waters reveals significant differentiation in both mitochondrial DNA (mtDNA) and nDNA between false killer whales found near the MHI and the NWHI (Martien et al. 2011).

The 2013 SAR recognizes three stocks of false killer whales with overlapping ranges within Hawaiian waters: 1) the MHI insular stock (DPS), which includes animals inhabiting waters within 140 km (approx. 75 nmi) of the main Hawaiian Islands, 2) the NWHI stock, which includes animals inhabiting waters within 93 km (50 nmi) of the NWHI and Kauai, and 3) the Hawaii pelagic stock, which includes false killer whales inhabiting waters greater than 40 km (22 nmi) from the main Hawaiian Islands, including adjacent high seas waters.

As currently defined these stock ranges include several areas of overlap (Fig. 3). Waters within 40 km of Kauai and Niihau are an overlap zone between the NWHI and MHI insular stocks. The three stocks overlap within 40 km and 93 km around Kauai and Niihau, and the MHI insular and pelagic stocks overlap from 93 km to 140 km around these islands. The MHI insular and pelagic stocks overlap between about 40 km and 140 km from shore between Oahu and Hawaii Island. Finally, the NWHI and pelagic stocks overlap within 93 km of the NWHI, inside the Papahānaumokuākea Marine National Monument.

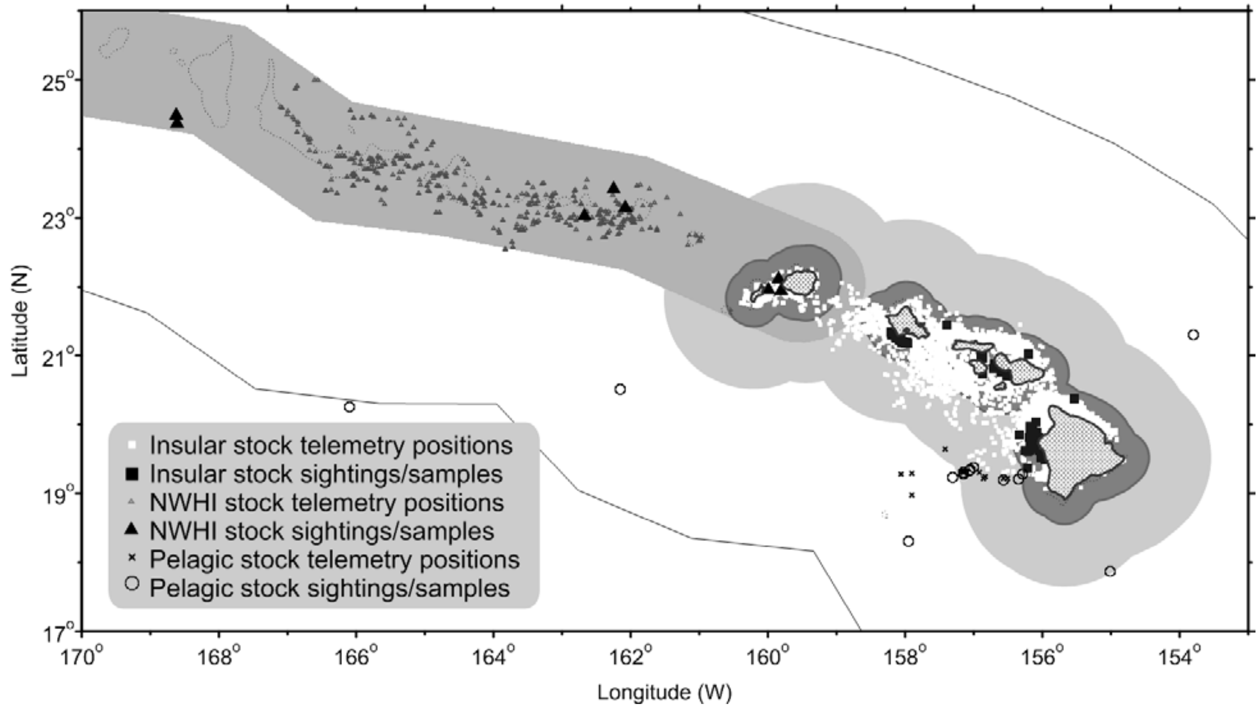

Figure **3**. Stock boundaries of MHI insular, NWHI insular, and pelagic false killer whales from the draft 2013 SAR. Also shown are sighting, biopsy, and telemetry records of false killer whale identified as being part of the MHI insular (square symbols), NWHI (triangle symbols), or pelagic (open and cross symbols) stocks. The darkest gray area is the 40-km MHI insular core area; lightest gray area is the 40-km to 140-km insular-pelagic overlap zone (Baird et al. 2010, Baird unpublished data; reproduced from Forney et al. 2010); medium gray area is the 50-nm (93-km) Papahanaumokuakea Marine National Monument boundary extended to the east to encompass Kauai, representing the NWHI stock boundary. The MHI insular, NWHI, and pelagic stocks overlap near Kauai and Niihau (Carretta et al. 2014).

Based on the information described above and the range as currently defined in the last Stock Assessment Report (Carretta et al. 2014), the endangered MHI IFKW DPS is considered to be the sole occupant of the waters in the MHI from shore to 40 km between Hawaii Island and Oahu (the "core range"). In addition the MHI insular stock of false killer whale shares habitat with the pelagic and NWHI stocks within 93 km of shore around Kauai and Niihau, and with the pelagic stock from 40 km to 140 km from the MHI. NMFS assumes all false killer whales offshore of 140 km are from the pelagic stock.

The 40-140 km boundary of the MHI insular/ pelagic stock overlap area was first defined in the 2010 Stock Assessment Report. The boundary was based on the maximum distance from shore that known MHI insular animals were documented (via satellite telemetry) to have traveled (112.8 km [60.9 nm]) at the time, and the minimum distance from shore that a pelagic stock animal was documented to have traveled (42 km [22.7 nm]) (Forney et al. 2010). NMFS included an additional geographic buffer because the sample size of satellite-tagged animals was considered sufficiently small that future deployment of satellite tags were expected to result in further increases in observed maximum distances (Forney et al. 2010). Additionally, sample sizes were considered insufficient to establish whether there might be differences in the offshore range of MHI insular false killer whales on the leeward versus windward sides of the islands.

Research on false killer whale movements has continued. Information that is currently being evaluated includes new satellite telemetry data from all three stocks. This more recent data, along with other relevant information, may support revision of the existing boundaries for insular and pelagic false killer whales, but new boundaries have not yet been determined. Nevertheless, we present the telemetry data here because it is available and may be informative to our biological opinion. First, the new data suggest asymmetrical usage of windward and leeward sides of the islands by MHI IFKW: the maximum distance from shore traveled by a MHI IFKW on the windward side of the islands is 51.4 km, while the maximum distance traveled on the leeward side is 114.9 km (Oleson et al. 2014). An important caveat is that the data are from individuals from two of the three identified social clusters of MHI IFKW; movement from one of the social clusters is unknown. Additionally, there is uneven seasonal distribution of the telemetry data. Second, recent data suggest that pelagic false killer whales may occur within the currently defined MHI insular "core area": at least one pelagic false killer whale was observed and satellite-tagged approximately 14 km off the island of Hawaii (Oleson et al. 2014). NMFS emphasizes that the new telemetry infomation represents an incomplete and unreviewed data subset on IFKW movements. While this information is useful to our understanding of the potential range of the MHI IFKW, we will continue to consider other relevant, peer-reviewed information in defining these boundaries. Accordingly, in this BiOp, NMFS has used the currently defined boundaries and bycatch to inform our jeopardy analysis, while taking into account the potential for a reduced range on the windward coast based on the new telemetry data.

Abundance of false killer whales in Hawaiian waters has been estimated using shipboard line-transect surveys of the entire Hawaiian EEZ, as well as aerial line-transect surveys and photographic capture-recapture analysis. The aerial surveys and photo-identification efforts covered only portions of the range of false killer whales in Hawaii. Hawaiian insular false killer whales were recognized as being separate from pelagic false killer whales only after the genetic results of Chivers et al. (2007) became available. Thus, many previously published estimates refer to both the insular and pelagic populations combined. Separate estimates of abundance are now available for both populations, with the best estimate of the size of the insular population being the result of capture-recapture analysis of photo-identification data. The 2009 Stock Assessment Report for MHI IFKW (Carretta et al. 2010) gives this as 123 individuals (CV = 0.72) (Baird et al. 2005). Recent reanalysis of the photographic data using more recent sighting histories and open population models has yielded new estimates of population size, for two time periods, 2000–2004, and 2006–2009. The new abundance estimate for 2000-2004 period is 162 (CV=0.23) animals (Carretta et al. 2014). Two estimates of population size were presented for the 2006–2009 period because two groups of whales photographed near Kauai have not been seen to associate into the social network of false killer whales seen at the other islands (Baird, 2009). The animals seen near Kauai have been associated with the NWHI stock and therefore not included in the most recent abundance estimate, which is 151 (CV=0.20) (Baird et al. 2013). The Hawaiian pelagic stock numbers 1,503 individuals (CV=0.66) based on current stock boundaries within the EEZ (Bradford et al. 2012). The population range for the Hawaii pelagic stock is known to extend into high-seas waters, though the specific boundaries of that population are unknown. The abundance estimate for the NWHI stock, based on a 2010 line transect survey is 522 (Carretta et al. 2014).

Mobley et al. (2000) estimated 121 MHI IFKW (CV = 0.47) obtained from aerial survey sightings pooled over 3 survey years (1993, 1995, 1997). That estimate was likely negatively biased because the survey aircraft did not allow detection of cetaceans directly below the plane, and no adjustment was made for availability bias (animals below the surface and unavailable for sighting as the aircraft passed overhead). Mobley et al. (2000) is considered out-of-date by NMFS standards (NMFS 2005b), although the results are similar to the 2000–2004 estimate referenced above.

In February 2009 a line-transect survey was conducted within 75 nmi of the main Hawaiian Islands where there were 6 sightings of false killer whales. Four on-effort sightings yielded a preliminary estimate of 635 false killer whales (CV = 0.77); however, not all groups could be attributed to the insular population. Some sightings occurred in the MHI IFKW/pelagic overlap zone, such that they may be pelagic false killer whales. Visual and acoustic observations of false killer whale behavior during the survey and the shape of the detection function (based on all sightings of false killer whale on previous NMFS Southwest Fisheries Science Center surveys) indicate that the estimate of 635 animals is an overestimation, particularly due to likely attraction of false killer whales to the survey vessel. Vessel attraction in other species has resulted in overestimation by as much as 400 percent (Turnock and Quinn II 1991). Without a correction factor, which is currently not available, the Pacific Scientific Review Group noted (PSRG 2010), and the Status Review of the MHI Insular False Killer Whale confirmed that this is not a reliable estimate of population because of a substantial upward bias for this species.

As identified in the 2013 SAR, the current minimum population estimate for the MHI IFKW is the number of distinct individuals identified during 2008-2011 photo-identification studies, or 129 false killer whales (Baird, unpublished data). Recent mark-recapture estimates (Oleson et al. 2010) of abundance are known to have a positive bias of unknown magnitude due to missed matches, and therefore are not suitable for deriving a minimum abundance estimate.

MHI IFKW are believed to have declined markedly during the 1990s, although their current population trajectory is unknown (Oleson et al. 2010). Historical population size is unknown. A survey was conducted in June and July 1989 on the leeward sides of Hawaii, Lanai, and Oahu to determine the minimum population size of false killer whales in Hawaiian waters. False killer whales were observed on 14 occasions with 3 large groups (group sizes 470, 460, and 380) reported close to shore off the island of Hawaii on 3 different days (Reeves et al. 2009). The largest group seen in 1989 is more than 3 times larger than the current best estimate of the size of the insular population. Although the animals seen during the 1989 surveys are assumed to come from the insular population, based on their sighting location near the Big Island, it is possible that they represent a short-term influx of pelagic animals to waters closer to the islands. For example, as noted above, recent data suggests that pelagic animals do move into what is currently considered the MHI insular stock core area (Oleson et al. 2014). The average group size during the 1989 survey (195 animals) is larger than the typical average group size for the insular population (25 animals for encounters longer than 2 hours) during more recent surveys (Baird et al. 2005), and is larger than that observed for the pelagic population (41 animals) (Bradford et al. 2014). However, because of the location of the sighting and lack of evidence at that time of pelagic animals occurring that close to the islands, it was considered likely that this group did consist of insular animals. Five systematic aerial surveys were conducted between

1993 and 2003 covering both windward and leeward sides of all of the main Hawaiian Islands, including channels between the islands, out to a maximum distance of about 46 km from shore (Mobley et al. 2000, Mobley 2004). A regression of sighting rates (8 groups in 1993, 9 in 1995, 1 in 1998, and none seen in 2000 and 2003 (Mobley et al. 2000, Mobley 2004) from these surveys suggests a significant decline in the population size ($p = 0.028$, $r_2 = 0.8429$ (Baird, 2009). The large group sizes observed in 1989, together with the declining encounter rates from 1993 to 2003 suggest that MHI IFKW have declined significantly in recent decades. Baird (2009) reviewed trends in sighting rates of false killer whales from aerial surveys conducted using consistent methodology around the MHI between 1994 and 2003 (Mobley et al. 2000) and found that rates during these surveys showed a statistically significant decline that could not be attributed to any weather or methodological changes.

The Status Review conducted by the NMFS Biological Review Team (BRT) (Oleson et al. 2010) presented a quantitative analysis of extinction risk using a population viability analysis (PVA). The modeling exercise was conducted to evaluate the probability of actual or near extinction, defined as a population reduced to fewer than 20 animals, given measured, estimated, or inferred information on population size and trends, and varying impacts of catastrophes, environmental stochasticity and Allee effects. The BRT chose to explore model structure uncertainty by using two separate types of models: the one rate-of-change model and the two-rate-of-change model. The BRT considered two hypotheses for population growth rate: 1) the decline apparent in the Mobley series (which occurred after the 1992 implementation of the longline exclusion area) results from ongoing mortality likely resulting from nearshore fisheries interactions, and 2) the decline apparent in the Mobley series results from a reduction in prey availability within the range of the insular population. The reduction-in-prey hypothesis results in the observed decline through the Mobley series but has a second growth rate for current and future growth based on a stable (but lower) prey base. Since there were only two time periods being used, three different distributions were used. The third distribution, which was considered the most plausible, constrained growth to biologically plausible values but allowed declines to the levels indicated by the Mobley time series (from an annual 10 percent decline to an annual 4 percent increase), which has an expectation of -3 percent population decline (Oleson et al. 2010). This model has a 19 percent probability of near extinction in 50 years, with the 1 growth-rate models ranging between 76 percent to 91 percent probability of near extinction (Oleson et al. 2010).

Since the historical population size is unknown, BRT members used density estimates from other areas together with the range inferred from telemetry data to suggest plausible ranges for historical abundance in the PVA. Using the estimated density of false killer whales around Palmyra Atoll, where the highest density of this species has been reported (Barlow and Rankin, 2007), and extrapolating that density to the 202,000 km2 area within 140 km of the MHI (which was proposed as a stock boundary for MHI IFKWs in the Draft 2010 Stock Assessment Report), a plausible historical abundance was around 769 animals. To allow for uncertainty in this estimate, higher potential abundances were also estimated as the mean plus one standard deviation (1392 animals), and the upper 95 percent lognormal confidence limit (2461 animals). There are several important caveats. Even though Palmyra has a density that is high relative to other areas, it is unlikely that this represented a pristine population during the 2005 survey on which the estimate is based. Given the depredation tendencies of false killer whales, known longlining in the Palmyra area, and the fact that false killer whales are known to become seriously injured or die as a result of interactions with longlines, the possibility that current densities are lower than historical densities cannot be discounted. Although Palmyra is situated in more productive waters than the Hawaiian Islands, we do not understand enough about the feeding ecology, behavior, and social system(s) of false killer whales to know how or whether productivity might be related to animal density for false killer whales.

The potential biological removal (PBR) level for the insular false killer whale stock is calculated as the minimum population estimate (129) times one half the default maximum net growth rate for cetaceans (½ of 4 percent) times a recovery factor of 0.1 (for a stock listed as Endangered under the ESA and with minimum population size less than 1500 individuals; Taylor et al. 2003) resulting in a PBR of 0.3 false killer whales per year (Carretta et al. 2014).

5.2.3.2 Life History Characteristics Affecting Vulnerability to Proposed Action

Depth preference, prey preference, and diving behavior affect the vulnerability of MHI IFKW DPS to Hawaii-based deep-set longline fishing. False killer whales may become hooked or entangled by longline gear, especially while depredating on bait or catch. NMFS considers both pelagic and MHI insular animals at risk of interacting with deep-set longline gear within the overlap zone, which is described below. From 2004-2014 observers recorded three interactions with false killer whales inside the range of the 140-km overlap zone with the deep-set fishery.

NMFS currently assumes (for purposes of attributing bycatch, among others) the endangered MHI IFKW solely occupies the shore to 40 km “core range” and overlaps with the pelagic and NWHI false killer whales as described above. As discussed below, there is inherent uncertainty in the data available to reach these conclusions. More recent data shows different usage of different sides of the islands by MHI IFKW. Based on telemetry data, the maximum distance from shore traveled by a MHI IFKW on the windward side of the islands is 51.4 km. The maximum distance traveled on the leeward side is 114.9km (Oleson et al. 2014). This data is based on recent tag data of MHI IFKW from two of the three identified social clusters of MHI IFKW; movement from one of the social clusters is unknown. In addition a pelagic false killer whale recently went to within 14 km off the island of Hawaii (Oleson et al. 2014). Based on this new information, revisions to the boundary lines for stock delineations may be appropriate for the purposes of prorating bycatch. However, until the working group has completed its analysis, the current boundaries and proration methods are the best available method for determining impacts to the stocks and DPS.

As discussed, satellite telemetry has documented MHI insular false killer whales to move up to 112.8 km (60.9 nm) offshore in certain areas, e.g., west-southwest of Oahu, Lanai, and Hawaii islands, but within the longline prohibited area. In other areas around the MHI, e.g., north-northeast of the MHI, this distance (60.9 nm) is sufficient to reach areas fished by the deep-set fishery, although telemetry has not shown insular whales to move into areas open to longline fishing. Still, due to the demonstrated potential of the whales to move this far from shore, the MHI insular population was considered vulnerable to hooking and entanglement from the deep-set fishery, in about 26 percent of its assumed (but not observed) range in the overlap zone that is not within the longline prohibited area (NMFS 2012c).

5.2.3.3 Threats to the Species

Here we provide a brief summary of the threats to false killer whales but more detailed information can be found in the SARs (available at: http://www.nmfs.noaa.gov/pr/sars/species.htm), the status review, and in the final listing rule.

As part of the Status Review the BRT qualitatively assessed the severity, geographic scope, and level of certainty of 28 potential individual threats to MHI IFKW. Because the severity and scope of individual threats may change through time, each threat was evaluated based on its historic impact and its current or future potential for impact. The factors believed to have had the greatest potential for contributing to the decline of MHI IFKW historically are reduced prey biomass and prey size, ecological interactions (competition) with commercial fisheries, accumulation of natural and anthropogenic contaminants, live captures prior to 1990, operational interactions with nearshore fisheries (primarily within about 40 km) and offshore longline fisheries (leading to injuries or deaths), and reduced genetic diversity. Some of the possible causes of the historic decline have been eliminated in recent times (e.g., live captures), and others have been limited in their geographic scope (e.g., interactions with the commercial longline fishery following the implementation of the 25–75 nmi longline exclusion boundary in 1991). Overall, the BRT considered the effects of small population size, hooking, entanglement, and intentional harm by fishers to be the most substantial threats to Hawaiian insular false killer whales.

Fisheries interactions were postulated to be a major threat because extensive unobserved troll, handline, and other hook-and-line fisheries target large pelagic fish and continue to operate at near-record levels in the Hawaiian insular false killer whale core area, and anecdotal but unconfirmed information on these fisheries suggests false killer whale interactions do occur. For these fisheries, the BRT assumed there is a high level of current and future risk assumed based on the scale and distribution of the troll and handline fisheries, and on anecdotal reports of depredation interactions by cetaceans, although interactions specific to false killer whales are known only for the troll fishery. The troll fishery has by far the greatest participation and effort in fishing days of any fishery within the known range of insular false killer whales, followed by the handline fishery, with the kaka- and shortline fisheries a distant third. The kaka-line and shortline fishing methods have been implicated as a threat based on the similarity of these fishing methods to the longline method. No other fishery has been identified as a fishery known to take MHI IFKW. Although there is no observer program and no self-reports have been submitted via the Marine Mammal Authorization Program, anecdotal information suggests depredation interactions have occurred off the north side of Maui and that the insular population may be impacted because of interactions with these small-scale fisheries (WPFMC 2009c). There are also reports of false killer whale depredation in other fisheries, and in 2013 a MHI IFKW stranded with hooks of several different types in its stomach, also suggesting there are interactions with other fisheries. The fish hooks have not been linked to the cause of death and therefore there has been no M&SI confirmed to be caused by any other fishery. Additionally the fisheries involved in this particular interaction have not been identified. For purposes of quantifying M&SI for all fisheries, the examples of fin scarring and hook ingestion do not, in the former case, establish whether the injuries were serious, and in the latter case, establish that the hooks were the cause of the animal's mortality. Nor do the examples cited lend themselves to systematic analysis such that other fisheries' M&SI may be reliably evaluated and quantified.

Hawaii Deep-set longline fishery impacts

The impact to the MHI IFKW from fisheries is difficult to determine due to the overlapping areas occupied by the three stocks. Impacts can only be identified to a stock from genetic samples or quality photos of the animals showing distinguishing features. Obtaining genetic samples from marine mammals hooked or entangled in longline fishing gear at sea presents risks to the safety of observers, fishermen, and the animals, and is only occasionally successful. Observers have not been able to obtain genetic samples from any false killer whale or unidentified blackfish that have been caught in the 140 km overlap zone. Likewise, obtaining reliable photos to compare to the photo-identification catalog of known animals has also been unsuccessful, mainly because most of the interactions occur at night, which is not optimal for obtaining necessary high quality photos.

When the stock identity of a false killer whale hooked or entangled by the longline fisheries within the MHI IFKW/ pelagic overlap zone cannot be determined, NMFS prorates the interaction to either the pelagic or MHI insular stock based on stock densities. Specifically, the model assumes densities of MHI insular stock animals decline and pelagic stock densities increase with increasing distance from shore (McCracken 2010). When this model was developed, the NWHI stock was not recognized as a separate stock and therefore was not included in the model, but will be incorporated into future models. For interactions with unidentified blackfish, NMFS prorates the animals to each species (false killer whale or short-finned pilot whale) based on a separate distance-from-shore model (McCracken 2010, Carretta et al. 2014).

Proration of false killer whales and unidentified blackfish takes within the overlap zone introduces additional uncertainty into the bycatch estimates. However, until methods of determining stock identity for animals observed taken within the overlap zone are available, and all animals taken can be identified to a population (e.g., from photos, tissue samples), this approach is the best available science for ensuring that potential impacts to all populations are assessed (Carretta et al. 2014).

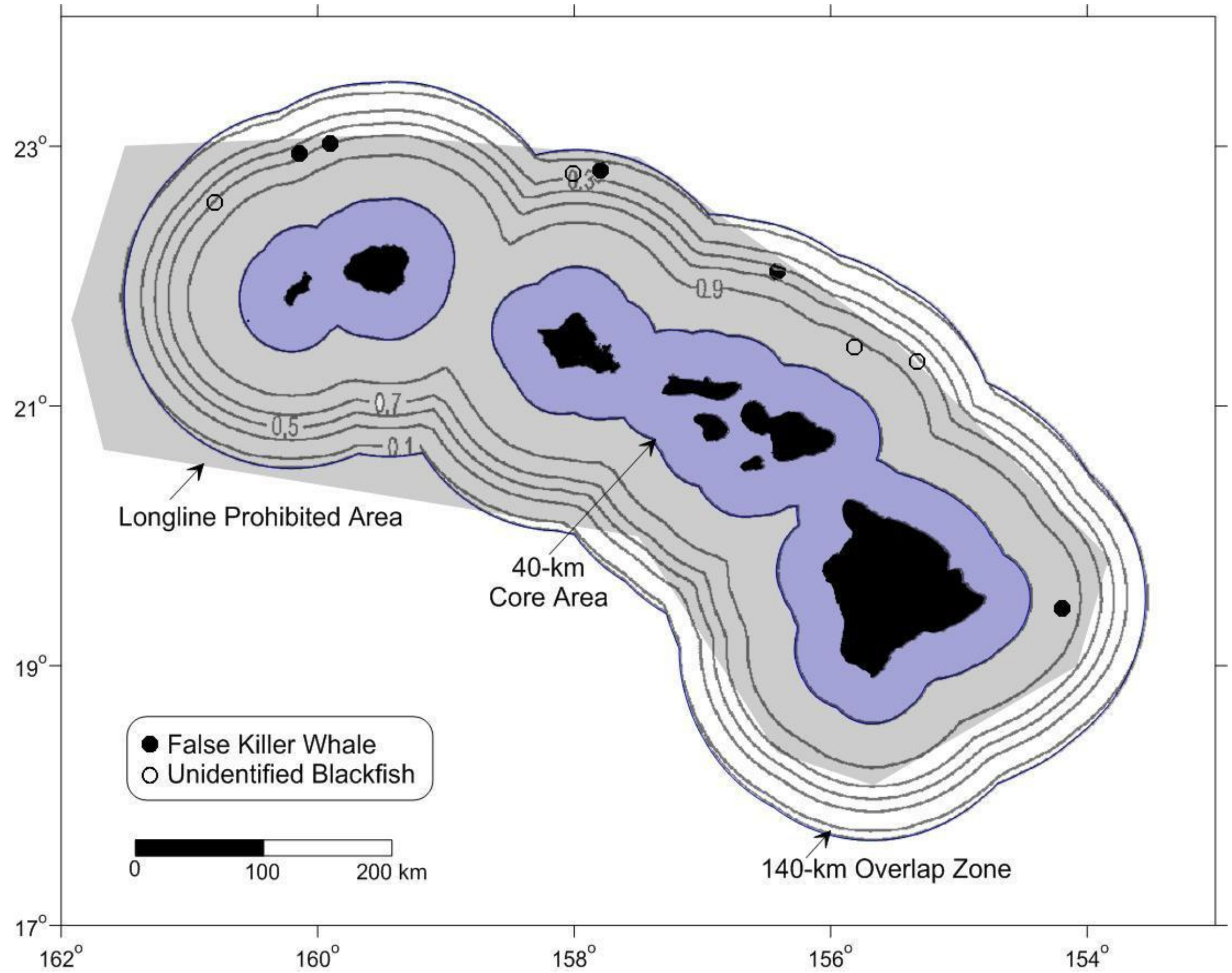

Figure 4. Observed interactions with false killer whales and unidentified blackfish in the overlap zone around the MHI, 2003-2012. The MHI Longline Fishing Prohibited Area is shaded in gray. Prior to implementation of the FKWTRP regulations in late 2012, the boundary contracted seasonally, allowing longline effort (and marine mammal interactions) to occur closer to the islands.

Based on the proration model described above and observed interactions and the expansion from observed interactions to an estimate of fleet-wide interactions based on the fishery's total effort and proration of blackfish and false killer whales of unknown stock identity (MHI insular versus pelagic), NMFS estimates that a total of 8.72 interactions occurred with MHI IFKW in the deep-set longline fishery from 2004-2014, which included both serious and non-serious injuries (Table 11). This is a conservative estimate that potentially overestimates the fishery's actual impact on MHI IFKW because the proration model does not account for the NWHI stock which was identified in 2011. For example, in 2012 two observed false killer whale interactions occurred in the area where the NWHI stock is known to occur (i.e., within the area where all three Hawaiian false killer whales stocks overlap), but at this time they can only be attributed (prorated) to a pelagic or MHI insular false killer whale. In addition, earlier interaction estimates are based on a much smaller abundance estimate for the pelagic false killer whale stock.

Injury determinations are made for all observed interactions. All estimated takes of false killer whales, and observed takes for which an injury severity determination could not be made, are prorated to deaths, serious injuries, and non-serious injuries based on the observed proportions of each outcome (Carretta et al. 2014).

The total of all known serious injury and mortalities to the MHI IFKW DPS as a result of commercial fishing operations for the time period from 2007 through 2011 is based on one observed unidentified blackfish interaction (either a false killer whale or short-finned pilot whale), resulting in an annual average M&SI of 0.1 animals. In 2012 there were two observed interactions with false killer whales in the insular/pelagic overlap zone, resulting in an annual average M&SI of 0.9 for 2008-2012 (NMFS 2014h).

Although more commonly observed in large whales, vessel strikes also have potential to kill or injure smaller cetaceans including false killer whales. Propeller strikes from large vessels are likely to be fatal, while propeller injuries from small boats may cause disfigurement of the dorsal fin or other parts of the body without killing the whale outright (Wells et al. 2008). There is also risk of blunt trauma injury from strikes. Slow-moving animals, or animals that associate with vessels, i.e., to ride the bow or depredate catch on fishing gear being retrieved, are likely the most susceptible to ship strikes. At least one juvenile killer whale that frequently approached vessels was killed by an injury caused by a ship off British Columbia, Canada (Laist et al. 2001). False killer whales in waters surrounding Hawaii (belonging to both insular and pelagic stocks) are known to ride the bow or stern wake of vessels and may come into proximity of propellers (Baird, pers.comm.). No ship-strike related injuries or deaths of false killer whales have been documented in Hawaiian waters, but Baird (2009) reported a fresh head wound on one individual from the insular population photographed off Oahu in September 2009 that may have been caused by a propeller strike.

False killer whales rely heavily on their acoustic sensory capabilities for navigation, foraging, and communicating with conspecifics. No specific studies or observations of the impacts of noise on wild false killer whales are available. However, intense anthropogenic sounds have the potential to interfere with the acoustic sensory system of false killer whales by causing permanent or temporary hearing loss, thereby masking the reception of navigation, forging or communication signals, or through disruption of reproductive, foraging, or social behavior. Experiments on a captive false killer whale have revealed that it is possible to disrupt echolocation efficiency in this species with the level of disruption related to the specific frequency content of the noise source as well as the magnitude and duration of the exposure (Mooney et al. 2009). In recent years there has been increasing concern that active sonar and seismic operations are harmful to beaked whales (Cox et al. 2006) and other cetaceans, including melon-headed whales (Southall et al. 2006) and pygmy killer whales (Wang and Yang, 2006). The use of active sonar from military vessels has been implicated in mass strandings of beaked whales, and recent mass-stranding reports suggest some delphinids are affected as well. False killer whales have been herded using loud sounds in drive fisheries off Japan (Kishiro and Kasuya 1993, Brownell et al. 2008), suggesting that high-intensity noise can affect the behavior of false killer whales in Hawaiian waters. The U.S. Navy Hawaii Range Complex surrounds the main Hawaiian Islands and is regularly used for training exercises that broadcast high-intensity, mid-frequency sonar sounds (U.S. Navy 2008). The specific location and timing of mid-frequency sonar training activities near Hawaii are not known, therefore, it is not possible to assess the frequency of occurrence of sonar use within the range of the MHI IFKW DPS.

Highly mobile species, such as marine mammals, can respond to effects of climate change more rapidly than their terrestrial counterparts (Harwood 2001). No significant climate change-related impacts to the MHI IFKW DPS have been reported to date. The relationship of climate change (e.g., global warming) on cetacean growth and mortality rates in the Hawaiian Archipelago is not well understood and requires further research in order to be taken into account by managers (IPCC 2007). The most likely impact of climate change on cetaceans will be changes in range related to migration, expansion, or contraction of the geographic thermal niche populations currently occupy, or changes in the distribution of prey species with particular thermal requirements. The ranges of 88 percent of cetaceans may be affected by changes in water temperature resulting from global anthropogenic climate change (MacLeod 2009). 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. A growing number of studies have demonstrated adverse impacts of acidification on marine organisms. These impacts include a reduction in the rate at which reef-building corals produce their skeletons, reduced ability of marine algae and free-swimming zooplankton to maintain protective shells, and reduced survival of larval marine species, including commercial fish and shellfish. Such impacts have serious implications for coral reef and pelagic ecosystem productivity. The net effect on higher trophic levels remains unclear, yet is likely to be negative. Acidification can be tenuously linked to increases in some gelatinous zooplankton (Richardson et al. 2009). Shifts in the pelagic species composition could have many direct and indirect effects on the distribution and abundance of large pelagic predators.

Due to a lack of scientific data, specific effects climate change will have on this species in the future are not predictable or quantifiable to any degree such as would allow for more detailed analysis in this consultation.

Based on instrumented longline catch rates (Boggs 1992) yellowfin tuna, marlins, spearfish, and mahi-mahi have generally shallower depth distributions during the day, and these may be more easily preyed upon by false killer whales, given the low frequency of deep dives by false killer whales. These species are also the main targets of the surface-fishing troll and handline fisheries. Many of the pelagic stocks in the Pacific are thought to be exploited at or near the maximum sustainable yield (MSY) level. Achieving MSY is the goal of fishery management, and in recent years concerns over the harvest of yellowfin and bigeye tuna (Harley et al. 2009, Langley et al. 2009) have prompted international action to reduce fishing mortality on these species. Similar concerns are motivating recommendations to limit fishing effort on striped marlin and albacore (ISC 2007). The status of skipjack tuna and swordfish stocks appear to be healthy. The status of populations of many other species is not known. Some studies indicate that as the biomass of tuna and billfish has been reduced by fishing, the abundance of intermediate trophic level species such as mahi-mahi and pelagic pomfrets has increased (Polovina et al. 2009). The BRT considered the impacts of reduced prey abundance to represent a medium risk for insular false killer whales both in the past and in the future. Although declines in prey biomass were more dramatic in the past when the insular false killer whale population may have been higher, the total prey abundance remains very low compared to the 1950s and 1960s as evidenced by CPUE data from Hawaii longline fisheries and biomass estimates from tuna stock assessments. A total of 2.9 to 3.9 millions of pounds per year is the estimated magnitude of prey required by the present-day MHI IFKW DPS. Competition with longline fishermen for potential prey within the insular false killer whale habitat seems to have represented a higher risk prior to the early 1990s when the longline fishery was harvesting many millions of pounds of fish per year, and where reported catch locations were almost all in what is now the longline exclusion zone. In the core nearshore habitat (< 40 km from shore) only the troll and handline fisheries now harvest as much as is estimated to be consumed annually by the MHI IFKW DPS.

5.2.3.4 Conservation of the Species

Considerable effort has been made in the last several years to reduce the impacts from the longline fisheries on false killer whales, including the MHI IFKW DPS. Pursuant to the MMPA, a take reduction team (TRT) was formed in 2010 and a take reduction plan (TRP) was published in late 2012 (NMFS 2012d) to address the issue of high levels of take and M&SI on false killer whales from the longline fisheries. The majority of the interactions were with the stock of pelagic false killer whales but the take with the MHI IFKW DPS was also addressed. The key outcome of the TRP included gear modifications to promote the release from the hook (i.e. use of a small diameter circle hook and strong leader, that will cause the hook to straighten under tension and come out of the animal, and release the animal with no or minimal trailing fishing gear), which is expected to result in interactions that cause fewer serious injuries. In order to avoid interactions with the MHI IFKW DPS the MHI longline fishing prohibited area which contracted seasonally was changed to remain closed year round. This closed area represents the entire core range, including the only observed areas where interactions with longline fisheries attributed to the MHI IFKW have occurred. This area may include the entire range on the windward side of the island as discussed above, of the insular DPS and over 74 percent of the insular/pelagic overlap zone leaving only a small area outside of their range on the south side of the islands where interactions could possibly occur (Figure 4).

5.2.4 Loggerhead Turtles

Information in this section is summarized primarily from the SEIS for Amendment 18 (WPFMC 2009), the 2009 Status Review (Conant et al. 2009), the 2011 Loggerhead DPS listing, the 2012 BiOp (NMFS 2012a) and other sources cited below.

The Services (NMFS and USFWS) determined that the loggerhead turtle (*Caretta caretta*) is composed of nine distinct population segments (DPSs) that constitute ‘‘species’’ that may be listed as threatened or endangered under the ESA. These loggerhead DPSs are the North Pacific Ocean, South Pacific Ocean, North Indian Ocean, Southeast Indo-Pacific Ocean, Southwest Indian Ocean, Northwest Atlantic Ocean, Northeast Atlantic Ocean, South Atlantic Ocean, and Mediterranean Sea. In the Pacific, the two loggerhead turtle DPSs, North Pacific and South Pacific, have been listed as endangered (76 FR 58868; September 22, 2011).

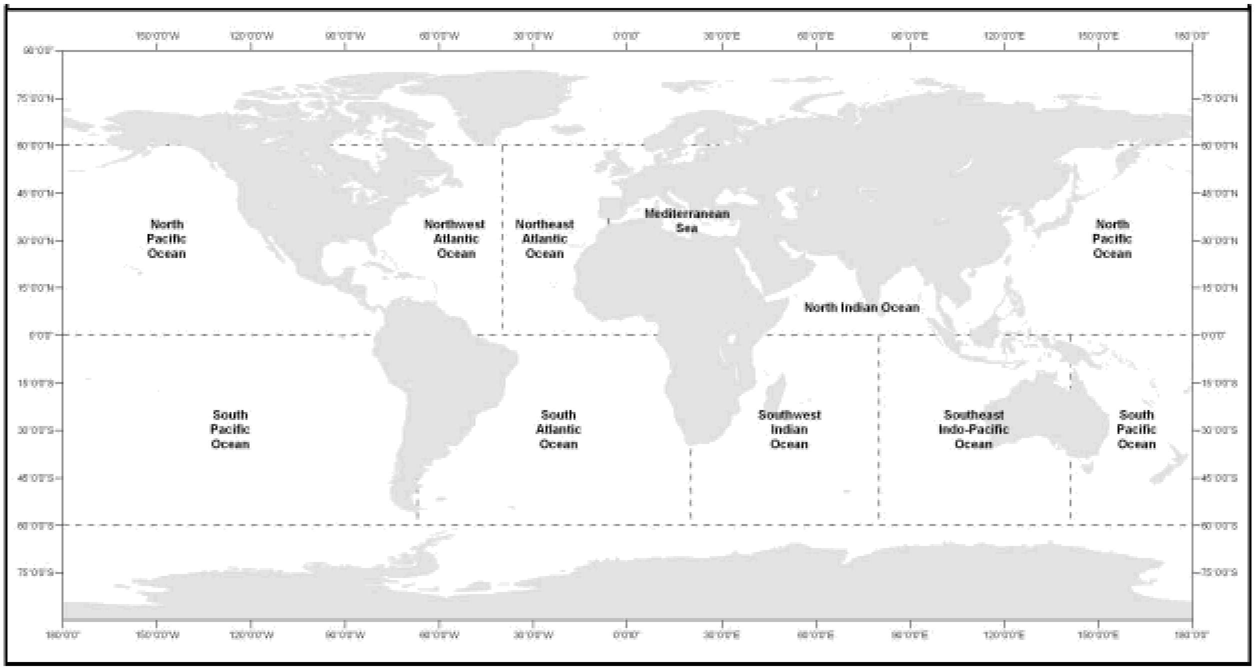

Figure 5. Map of Loggerhead Sea Turtle DPS boundaries (76 FR 58868; September 22, 2011).

5.2.4.1 Population Characteristics

Loggerheads are circumglobal, inhabiting continental shelves, bays, estuaries, and lagoons in temperate, subtropical, and tropical waters. Major nesting grounds are generally located in temperate and subtropical regions, with scattered nesting in the tropics. Natal homing of female loggerheads to nesting beaches maintains regional population structure. The North Pacific loggerhead DPS nests primarily in Japan (Kamezaki et al. 2003), although low level nesting may occur outside of Japan in areas surrounding the South China Sea (Chan et al. 2007, Conant et al. 2009). 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 40 percent 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 percent) in the size of the annual loggerhead 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 (STAJ 2008, 2009, 2010, 2012; Y.Matsuzawa, pers.comm 2014). The most recent 2013 nesting numbers do not take into account nests from Yakushima Island, which typically represents 40 percent of loggerhead nesting in Japan (Kamezaki et al. 2003).

For the 23-year period 1990-2013, the total number of nests per year for the North Pacific DPS ranged between 2,064 – 15,396 nests. Assuming a clutch frequency of four per female per year (Van Houtan 2011), the number of nesting females per year between 1990 and 2013 ranged between 516 – 3849. The total number of adult females in the population was estimated at 6,673 by Van Houtan (2013). Given that population estimates for sea turtles are problematic due to lack of demographic information of all life stages (NRC 2010), NMFS will use nesting or nesting female data as population indices in this opinion. A recent study determined that the current mean annual abundance of loggerheads in the North Pacific is 43,226 (Seminoff et al. 2014). In 2011 an analysis of nesting data was done using two different models for a population viability assessment (PVA), a classical model and a climate-based model (Van Houtan 2011, NMFS 2012). The classical PVA calculated population growth and its variability from time series of nest counts, and this model resulted in a population increase over the next 100 years. The second approach was the climate-based PVA that considers bottom-up climate forcing at two life stages, neonates and breeding females. The Climate-based PVA forecasted a population decline below the 50 percent QET-level set by Van Houtan (2011) within one generation (25 years). Although the model is considered highly accurate based on its ability to account for historical population changes, because of the difficulty predicting PDO, the model cannot forecast population trends beyond 25 years. While both the classical and climate-based approaches have limitations, NMFS believes that the climate-based model is more rigorous in applying actual data, and therefore generally more useful for assessing population trends.

5.2.4.2 Life History Characteristics Affecting Vulnerability to Proposed Action

Loggerhead 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. Satellite tracking of juvenile loggerheads indicates the Kuroshio Extension Bifurcation Region to be an important pelagic foraging area for juvenile loggerheads (Polovina et al. 2006, Kobayashi et al. 2008, and Howell et al. 2008). Other important juvenile turtle foraging areas have been identified off the coast of Baja California Sur, Mexico (Peckham and Nichols 2006, Peckham et al. 2007, Conant et al. 2009, Wingfield et al. 2011). After spending years foraging, potentially two decades (Tomaszewicz et al. 2014), in the central and eastern Pacific, loggerheads return to their natal beaches for reproduction (Resendiz et al. 1998, Nichols et al. 2000) and remain in the western Pacific for the remainder of their life cycle (Iwamoto et al. 1985, Kamezaki et al. 1997, Conant et al. 2009, Hatase et al. 2002, Ishihara et al. 2011). Based on tag-recapture studies, the East China Sea has been identified as a major habitat for post-nesting adult females (Iwamoto et al. 1985; Kamezaki et al. 1997, 2003; Kobayashi et al. 2008, 2011).

Given that the action area is oceanic, the main aspects of North Pacific loggerhead life history affecting their vulnerability to Hawaii-based deep-set longline fishing are juveniles foraging and migrating across the oceanic zone, as discussed below. The Hawaii-based deep-set fishery primarily interacts with juvenile loggerhead turtles (Van Houtan 2013). In the central North Pacific Ocean, foraging juvenile loggerheads 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. This boundary area is referred to as the Transition Zone Chlorophyll Front and is favored foraging and developmental habitat for juvenile loggerhead turtles (Polovina et al. 2001). Satellite telemetry of loggerheads also identified the Kuroshio Extension Current (KEC), specifically the Kuroshio Extension Bifurcation Region (KEBR), as a forage hotspot (Polovina et al. 2006, Kobayashi et al. 2008). The KEBR 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. lat. (Polovina et al. 2004) and 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). Data collected from stomach samples of juvenile loggerheads indicate a diverse diet of pelagic food items (NMFS 2006, Parker et al. 2005). The Pacific Coast of the Baja California Peninsula, Mexico is a foraging hotspot for loggerheads from nesting beaches in Japan (Seminoff et al. 2014).

Loggerheads are a slow-growing species that reach sexual maturity at 25 to 37 years of age, depending on the DPS (NMFS and USFWS 2007a). Conant et al. (2009) estimate age to maturity of the North Pacific DPS to be 30 years +/- 5 yrs. Van Houtan and Hailey (2011) estimated age at first reproduction to be 25 years. The North Pacific loggerhead range spans the entire North Pacific Ocean, hence migration of juveniles and adults between terrestrial (nesting), near-shore and pelagic habitats may result in criss-crossing of the action area during life stages. However, tagging studies indicate that juvenile loggerheads are shallow divers that do not forage frequently at depths fished by deep-set gear (<100 m; Polovina et al. 2003, 2004), spending 40–80 percent of time at surface and 90 percent of time at depths <15 m (Howell el al. 2010). Although juvenile loggerheads may forage within the action area, they do not typically forage at depths fished by the deep-set fishery; hence they are not as susceptible to interactions with deep-set gear as they are to shallow-set gear.

5.2.4.3 Threats to the Species

Global threats to loggerhead turtles are spelled out in the 5-year review (NMFS and USFWS 2007a), and threats to the North Pacific loggerhead DPS are described in more detail in Conant et al. 2009, and Van Houtan 2010. Major threats to the species, according to these sources, are fisheries bycatch, alteration of nesting habitat, and direct harvest and predation, which are briefly described below. In addition, anthropogenic climate change and marine debris appear to be a growing threat to this species, and are discussed briefly below.

Sources of mortality for North Pacific loggerheads in addition to U.S. western Pacific commercial longline fisheries include: human encroachment and egg harvest/predation on nesting beaches, nesting beach alteration (armament and habitat degradation), incidental capture in coastal and pelagic fisheries (including longline, drift gillnet, set-net, bottom trawling, dredge, and trap net) throughout the species' range (Conant et al. 2009; Dutton and Squires 2008; Peckham et al. 2007, 2008; Kudo et al. 2003; Ishihara 2009, Ishihara et al. 2011; Koch et al. 2006; Van Houtan and Halley 2011). Interactions and mortality with coastal and artisanal fisheries in Mexico and the Asian region likely represent the most serious threats to North Pacific loggerheads (Peckham et al. 2007, 2008; Ishihara 2009; Conant et al. 2009). Bycatch and fisheries-related standings' numbering in the thousands annually have been reported from gillnet and longline fisheries operating in loggerhead 'hotspots' off of Baja Mexico where intense coastal fishing pressure overlaps with high densities of loggerheads 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). In a recent study results suggest that up to 11 percent of the region's loggerhead 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 turtles (with a mean of 420 ± 274/yr) along 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. (Peckham et al., 2007, 2008). Recent efforts to estimate at-sea mortality of sea turtles in Baja using drifter experiments concluded that stranding probability estimates may vary between 5-20 percent, and it is likely that 150 loggerheads may have died during a 15 day period in 2010 (Koch et al. 2013). In 2012, a record level of loggerhead strandings occurred with 438 loggerhead sea turtles stranded, dead, along 43 kilometers of the shoreline of Playa San Lazaro, Baja California Sur in July 2012 (representing a 600 percent increase over the annual average of 78 loggerheads in July since 2003) which prompted the U.S. to identify Mexico under the MSRA for fishing interactions involving the bycatch of loggerhead turtles (Benaka et al. 2012, NMFS 2013b). Mexico now has two years (by January 2015) to demonstrate that they have a comparable regulatory program to reduce or end sea turtle bycatch or they could be negatively certified under the Magnuson Stevens Reauthorization Act (MSRA). A negative certification may result in prohibitions on the importation of certain fishery products into the United States or the denial of port privileges for vessels of that nation.

Preliminary research of coastal pound net fisheries in Japan also suggests high mortality to loggerheads and that these fisheries may pose a major threat to mature stage classes of loggerheads due to pound net operations offshore of nesting beaches in coastal foraging areas (Ishihara 2007, 2009). Pound nets in Japan operate nearshore in depths up to 100m and range in size measuring up to 10,000m^3. Nets consist of a leader set perpendicular to the coast that directs fish into standing nets that entrain fish into an enclosed trap mounted either at the surface or midwater. Fish are retrieved at regular intervals (usually daily) from poundnets, enabling live release of turtles and other bycatch from surface traps. However, pound nets with midwater traps prevent sea turtles from reaching the surface to breathe and thus can result in high mortality rates. Hence coastal pound net fisheries off Japan may pose a significant threat to the North Pacific DPS population (76 FR 58868; September 22, 2011).

In addition to interactions in Hawaii-based longline fisheries which resulted in 15 estimated loggerhead mortalities in the shallow-set fishery since 2005 (NMFS 2014a,b) and an estimated 16 loggerhead mortalities (two per year on average) in the deep-set fishery (NMFS 2014a), longline fisheries operating out of other countries are still using traditional methods (J style hooks with squid bait) and are likely injuring and killing at least many hundreds of turtles annually in the North Pacific (NMFS and USFWS 2007a).

Before 2001 in the North Pacific, longline fisheries operating out of Hawaii were estimated to capture an average of about 417 loggerheads a year (McCracken 2000). If we apply the old mortality rate of 40 percent (Gilman et al. 2007a), this would give us an estimated mortality of 167 (417*40 percent= 166.8) annually before the shallow-set portion of the fishery was closed in 2001. The Hawaii shallow-set longline fishery subsequently reopened in 2004 and was subject to a number of management measures that were designed to minimize bycatch and post-hooking mortality. The 2004 management measures have proven to reduce loggerhead interaction rates by 90 percent (Gilman et al. 2007a, WPFMC 2009b). Other U.S. fisheries that operate in the Pacific and interact with loggerhead sea turtles are the California/Oregon (CA/OR) drift gillnet fishery that targets swordfish and thresher shark off the west coast and the California longline experimental fishery. This fishery has been observed by the NMFS Southwest Region since 1990, with roughly 20 percent observer coverage. From 1990 to 2013, the CA/OR drift gillnet fishery was observed to incidentally capture 19 loggerheads (14 released alive, 1 injured, and 4 mortalities). Since 2000, restrictions have been in place to close areas to drift gillnet fishing off Southern California when loggerheads are expected to be in the area (i.e., closed during June, July and/or August during forecasted or occurring El Niño events). On July 25, 2014 NMFS announced the closure of the Pacific Loggerhead Conservation Area due to El Niño conditions (79 FR 43268). The California Oregon drift gillnet fishery has an incidental take statement for up to seven anticipated loggerhead interactions and four anticipated mortalities during a 5-year period (NMFS 2013a). The loggerhead regulations were modified in 2004 (69 FR 1844, April 7, 2004) and 2007 (72 FR 31756, June 8, 2007). Only two loggerheads have been observed taken incidentally in the California drift gillnet fishery since 2001, one in 2001 and one in 2006 (NMFS 2013a). The Eastern Tropical purse seine fishery caught five loggerheads in 2000 but the location of capture is unknown to determine if these were from the North or South Pacific DPS (NMFS 2005a).

Destruction and alteration of loggerhead nesting habitats are occurring throughout the species' range, especially coastal development, 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 and restricting beach migration in response to environmental variability. 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 (NMFS and USFWS 2007a). In Japan, where the North Pacific loggerhead DPS nests, many nesting beaches are lined with concrete armoring, causing turtles to nest below the high tide line where most eggs are washed away unless they are moved to higher ground (Matsuzawa 2006). Coastal development also increases artificial lighting, which may disorient emerging hatchlings, causing them to crawl inland towards the lights instead of seaward. Coastal development also improves beach access for humans, resulting in more vehicle and foot traffic on beaches, which may result in compaction of nests and reduction of emergence success (NMFS and USFWS 2007a). In Japan, threats to nesting and nest success include light pollution, poorly managed ecotourism operations, and trampling due to the thriving tourist economy on Yakushima Island, and increasing numbers of beachfront hotels and roadways (Kudo et al. 2003). Overall, the Services have concluded that coastal development and coastal armoring on nesting beaches in Japan are significant threats to the persistence of this DPS (76 FR 58868; September 22, 2011). On July 10, 2014, NMFS issued a final rule to designate critical habitat for the Northwest Atlantic Ocean loggerhead sea turtle DPS within the Atlantic Ocean and the Gulf of Mexico (79 FR 39856). Specific areas for designation include 38 occupied marine areas within the range of the Northwest Atlantic Ocean DPS. No marine areas meeting the definition of critical habitat were identified within the jurisdiction of the United States for the North Pacific Ocean DPS, and therefore no critical habitat has been designated for this DPS (NMFS 2014g).

Deliberate hunting of loggerheads for their meat, shells, and eggs is reduced from previous levels, but still exists. The North Pacific loggerhead DPS nests almost exclusively in Japan, especially on Yakushima Island. In 1973, a law was enacted on Yakushima Island prohibiting harvest of sea turtle eggs. A similar law was enacted in 1988 encompassing most of the other loggerhead nesting beaches in Japan, resulting in great reductions in egg harvest. The 1973 law may in part explain the increasing number of nesting turtles from 2001 to 2011, given that loggerheads mature in about 25 years (Ohmuta 2006). Predation of eggs also occurs, for example by raccoons and feral animals in Japan (NMFS and USFWS 2007a, STAJ 2010). While sea turtles have been protected in Mexico since 1990 (Conant et al. 2009), studies have shown that loggerheads continue to be caught, both indirectly in fisheries and by a directed harvest of juvenile turtles (Gardner and Nichols 2001; Koch et al. 2006; Peckham et al. 2007; Mancini et al. 2011).

Marine debris is also a source of mortality to all species of sea turtles because small debris can be ingested and larger debris can entangle animals leading to death. Marine debris is defined by NOAA as any persistent solid material that is manufactured or processed and directly or indirectly, intentionally or unintentionally, disposed of or abandoned into the marine environment. Manmade materials like plastics, micro plastics, and derelict fishing gear (e.g., ghost nets) that may impact turtles 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 (Arthur et al. 2009, Balazs 1985, Bjorndal et al. 1994, Bugoni et al. 2001, Doyle et al. 2011, Keller et al. 2004, Parker et al. 2011, Wabnitz and Nichols 2010). All marine turtles have pelagic stages; including when they leave the nesting habitat as hatchlings and enter a period known as the “lost years” that can last for years or decades (Lutz and Musick 1997, Zug et al. 2002). While the impact of marine debris to Pacific turtles during pelagic life stages is currently unquantified, it is quite 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 (Arthur et al. 2009, Doyle et al. 2011, Stewart et al. 2011, NMFS and USFWS 2007a, Hutchinson and Simmonds 1992, Law et al. 2010, Mrosovsky et al. 2009, Wabnitz and Nichols 2010). The addition of debris from the earthquake and tsunami that hit Japan in March 2011 increases concern due to the large amount of debris that entered the water in a short time. The Japanese government estimated that 25 million tons of debris was generated but there is no confirmed estimate of how much entered the water, and little information as to the type of debris that entered the water. It is believed that it is highly unlikely that the debris is radioactive for several reasons; the vast majority of the debris was many miles away from the reactor that leaked, the leak of contaminated water from the reactor into the sea started days to weeks after the debris was washed out to sea, and vessels coming into the U.S. from Japan were monitored for radiation, and readings were below the level of concern. The large debris field that was initially generated has broken up so it is no longer visible by satellite, which means that it can no longer be monitored so the location of the debris is unknown and projections of when it will reach shore can only be predicted using models that take into account oceanic and wind conditions (NOAA Marine Debris Program). For loggerheads the greatest risk is in the pelagic environment but there is no information to quantify what the impact will be.

As highly migratory, wide-ranging organisms that are biologically tied to temperature regimes, sea turtles are vulnerable to effects of climate change in aspects of their physiology and behavior

(Van Houtan 2011). Climate refers to average weather conditions, as well as associated variability. The term climate change refers to any distinct change in measures of climate lasting a long period of time, which means major changes in temperature, rainfall, snow, or wind patterns lasting for decades or longer. Climate change may result from: natural factors, such as changes in the Sun's energy or slow changes in the Earth's orbit around the Sun; natural processes within the climate system (e.g., changes in ocean circulation); and human activities that change the atmosphere's makeup (e.g., burning fossil fuels) and the land surface (e.g., cutting down forests, planting trees, building developments in cities and suburbs, etc.), also known as anthropogenic climate change (U.S. Environmental protection Agency). Impacts to marine turtle populations resulting from climate change may occur at different rates or at different levels between marine turtle species based on a number of factors.

Increasing temperatures at nesting beaches may impact sex ratios of hatchlings and/or increase embryonic mortality (Matsuzawa et al. 2002). The North Pacific DPS is estimated to have a 1:1 male to female ratio (NMFS and USFWS 2007a), and while nest temperatures in Japan may be within survival thresholds, high beach incubation temperatures have also occurred resulting in mortality of pre-emergent hatchlings in Japan (Matsuzawa 2006). This population may be less vulnerable to increases in sand temperature than those already highly skewed toward female or at the high end of thermal tolerance, but 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. In the future, increasing temperatures, sea level rise, changes in ocean productivity, and increased frequency of storm events are expected as a result of climate change and are all potential threats for loggerheads.

A final factor when considering the effects of future anthropogenic climate change is the role the Pacific Decadal Oscillation (PDO) plays in influencing turtle populations. A recent study mentioned above combined two factors of climate variability, changes in ocean circulation and sea surface temperatures (SST) on two different life stages of loggerhead sea turtles, (neonates[6] and adult females) to see how they influence population trends (Van Houtan and Halley 2011). This study found that changes in loggerhead nesting over at least the last several decades are strongly correlated with ocean oscillations due to environmental influences on juvenile recruitment (Van Houtan and Halley 2011, Van Houtan 2011). In the next 22 years, loggerheads are projected to decrease due to unfavorable conditions in the PDO in recent years. Beyond this time we do not have information to predict what the population will do (NMFS 2011, Van Houtan 2011). Arendt et al. (2013) found that historical climate forcing on the oceanic habitat of neonate sea turtles in the Atlantic explained only two-thirds of interannual variability and concluded that annual nest count trends are more influenced by remigrants than neophytes; however the same analysis has not been done for loggerheads in the Pacific. Juvenile recruitment appears to be strongly correlated with the PDO in the Kuroshio Bifurcation Extension Region where juveniles congregate (Polovina et al. 2006) as they are most susceptible to oceanographic variability given their limited ability to exploit their environment for food (Van Houtan and Halley 2011). SST in the months preceding nesting has been demonstrated to influence whether females nest due to the need for sufficient nutrients for yolk production (Van Houtan and Halley

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[6] Neonates are defined as hatchlings up to six months of age for the purpose of this study (Van Houtan, pers. Comm.).

2011). Additional studies that simulated changes in physical ocean properties in northern hemisphere westerly's in response to various future CO^2 emission scenarios predict that the area and primary production of the temperate oceanic biome in the North Pacific is anticipated to decrease by 34 percent over the next century (Polovina et al. 2011). The extent of the impact on species in the region, such as loggerheads, is unknown because we do not know how species may or may not adapt to changes over the long-term (Chaloupka et al. 2009).

5.2.4.4 Conservation of the Species

Considerable effort has been made since the 1980s to document and reduce loggerhead bycatch in Pacific Ocean fisheries, as this is the highest conservation priority for the species. NMFS has formalized conservation actions to protect foraging loggerheads in the North Pacific Ocean which were implemented to reduce loggerhead bycatch in U.S. fisheries. Observer programs have been implemented in federally-managed fisheries to collect bycatch data, and several strategies have been pursued to reduce both bycatch rates and post-hooking mortality. In Pacific Ocean fisheries these include developing gear solutions to prevent or reduce capture (e.g., circle hooks), implementing seasonal time-area closures to prevent fishing when turtles are congregated, modifying existing gear, and developing and promoting Sea Turtle Handling Guidelines (NMFS and USFWS 2007a). For example, switching to large circle hooks and mackerel bait in 2004 reduced the interaction rate by approximately 90 percent in the Hawaii shallow-set longline fishery (Gilman et al. 2007a, WPFMC 2009b). In 2003, NMFS implemented a time/area closure in southern California during forecasted or existing El Niño-like conditions to reduce the take of loggerheads in the California/Oregon drift gillnet fishery (68 FR 69962; December 16, 2003). On July 25, 2014, El Niño conditions off the coast of California were determined to have been met for the first time which closed the Pacific Loggerhead Conservation Area through August 31, 2014 to swordfish drift gillnet fishing (79 FR 43268; July 25, 2014). Efforts to elevate the awareness level of fishermen regarding protected species interactions and the potential impacts to their fisheries (i.e., closures if allowable take levels are exceeded in the shallow-set component of the fishery), and efforts to educate boat owners and operators via annual (and mandatory) protected species workshops cannot be discounted. NMFS has also developed a mapping product known as TurtleWatch that provides a near real time product that recommends areas where the deployment of pelagic longline shallow sets should be avoided to help reduce interactions between Hawaii-based pelagic longline fishing vessels and loggerhead sea turtles (Howell et al. 2008).

Since loggerhead interactions and mortalities with coastal fisheries in Mexico and Japan are of concern and are considered a major threat to North Pacific loggerhead 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, Sea Turtle Association of Japan (STAJ), and the Western Pacific Fisheries Management Council, a prominent Baja California Sur fleet retired its bottom-set longlines in 2007 (Peckham et al. 2008, Peckham and Maldonado-Diaz, 2012). Prior to this closure, the longline fleet interacted with an estimated 1,160-2,174 loggerheads annually, with nearly all (89 percent) of the takes resulting in mortalities (Peckham et. al. 2008). Because this fleet no longer interacts with loggerheads, conservation efforts have resulted in the continued protection of approximately 1,160-2,174 juvenile loggerheads annually (76 FR 58868; September 22, 2011). Additionally, stranding data collected since 2003 at Playa San Lazaro indicates a 60 percent reduction in standings' during 2010 compared to previous 2003-2009 averages (Peckham 2010). Led by the Mexican Wildlife Service, a federal loggerhead bycatch reduction task force, comprised of federal and state agencies and non-governmental organizations, was organized in 2008 to ensure loggerheads receive the protection they are afforded by Mexican law. In 2009, while testing a variety of potential solutions, ProCaguama's fisher-scientist team demonstrated the commercial viability of substituting bycatch-free hook fishing for gillnet fishing. ProCaguama, in coordination with the task force, is working to develop a market-based bycatch solution consisting of hook substitution, training to augment ex-vessel fish value, development of fisheries infrastructure, linkage of local fleets with regional markets, and concurrent strengthening of local fisheries management (Conant et al. 2009). As of 2012, a number of members of the gillnet fleet had retired their gear (a total of 140 gillnets), 18 crews have converted to hook and line fishing (a more sustainable practice in the 'hotspot' area that results in zero bycatch), and local NGO efforts were underway to implement the market-based solutions mentioned above to encourage consumption of sustainably caught sea food (Peckham 2014, Conant et al. 2009). Unfortunately, a record level of loggerhead strandings occurred in July and August 2012 that prompted the U.S. to identify Mexico under the MSRA for fishing interactions involving the bycatch of protected living marine resources (NMFS 2013b). As required under the MSRA, Mexico and the U.S. are now engaged in a bilateral effort to identify and implement a regulatory program that is comparable in effectiveness to that of the United States. However, market-based approaches to promote and support sustainable fisheries that use fishing gear, such as hand line, which have low (or no) sea turtle bycatch in Baja continues (Peckham 2014).

In Japan, due to concerns of high sub-adult and adult loggerhead mortality in mid-water pound nets, researchers with the STAJ, ProCaguama, and NMFS have begun collaborations, together with local fishermen of several Japanese prefectures, 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 February 2012 with the identification of three promising gear mitigation options for pound net fisheries (Matsuzawa et al. 2012). Continued collaborative efforts continue to progress development of mitigation measures that include refinement of solutions and testing in an operating net to assess target and non-target catch rates to secure industry uptake and buy-in (Ishihara et al. 2012).

Conservation efforts have also focused on protecting nesting beaches, nests, and hatchlings. Much of Japan's coastline is "armored" using concrete structures to prevent and minimize impacts to coastal communities from natural disasters. These structures have resulted in a number of nesting beaches losing sand suitable for sea turtle nesting, and nests often need relocating to protect them from erosion and inundation. In recent years, a portion of the concrete structures at a beach in Toyohashi City, Aichi Prefecture, was experimentally removed to create better nesting habitat (76 FR 58868; September 22, 2011). The STAJ along with various other organizations in Japan, are carrying out discussions with local and Federal Government agencies to develop further solutions to the beach erosion issue and to maintain viable nesting sites. The Ministry of Environment has supported the local NGO conducting turtle surveys and conservation on Yakushima in establishing guidelines for tourism to minimize impacts by humans on nesting beaches (Conant et al. 2009). Yet, beach erosion and armament still remain one of the most significant threats to nesting beaches in Japan (Conant et al. 2009). Since 2003, WPFMC has been contracting with STAJ to protect loggerhead nests and increase hatchling survivorship at several nesting beaches in southern Japan, including at the two primary beaches on Yakushima Island. Beach management activities include conducting nightly patrols during the summer nesting season to relocate nests from erosion prone areas, protecting nests from predators and people with mesh and fences, and cooling nests with water and shading to prevent overheating during incubation. STAJ has developed techniques for nest relocation that now result in an average of 60 percent hatchling success rates (compared to nearly zero survival of the same nests laid in erosion prone areas). Nest relocation in 2004-08 resulted in an estimated 160,000 hatchlings being released that otherwise may have been lost (76 FR 58868; September 22, 2011). Conservation efforts funded by the WPFMC continued through 2012 and benefits to the population are assumed to be commensurate with 2004-08 activities, but analysis is lacking of total estimated hatchlings released as a result of conservation efforts (2004-2012).

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

5.2.5 Leatherback Turtles

Information in this section is summarized primarily from the leatherback 5-year status review (NMFS and USFWS 2013), the May 2007 Leatherback focus issue of the journal Chelonian Conservation and Biology, the Turtle Expert Working Group's (TEWG) report on Atlantic leatherback (TEWG 2007), the 2012 BiOp (NMFS 2012a), the Pacific Leatherback Assessment

Working Group (PLAWG) convened by PIFSC and SWFSC (PLAWG 2012), and other sources cited below.

Although this species is listed globally (Table 1), it is difficult to characterize the global status and trend of leatherbacks on a global scale because the species consists of many discrete populations that may increase or decrease independently of one another. The leatherback 5-year status review (NMFS and USFWS 2013) does not make a determination regarding global status and trends, but rather limits its conclusions to the status and trends of populations for which information is available. Some populations are stable or increasing, but other populations for which information is available are either decreasing or have collapsed (NMFS and USFWS 2013, TEWG 2007, PLAWG 2012). The discovery of the world's fourth-largest leatherback nesting area on the Atlantic coast of Panama and Columbia (Patino-Martinez et al. 2008) supports the TEWG's conclusion that leatherback nesting is increasing in parts of the Atlantic and Caribbean (TEWG 2007).

5.2.5.1 Population Characteristics

Leatherbacks have the widest distribution of any sea turtle and can be found from the equator to subpolar regions in both hemispheres. In the Pacific, tagging studies have shown that leatherbacks can traverse entire ocean basins when foraging. Leatherbacks can forage in the cold temperate regions of the oceans, occurring at latitudes as high as 71° N and 47° S (Benson et al. 2011, Shillinger et al. 2008); however, nesting is confined to tropical and subtropical latitudes. The global leatherback population is not homogeneous because natal homing of female leatherbacks to nesting beaches maintains regional population structure. Leatherback populations occur in at least the western Pacific, the eastern Pacific, the Indian Ocean, Florida, the Caribbean, Africa, and Brazil, with further population structure at smaller spatial scales in some areas (e.g., the Caribbean), as described in the leatherback 5-year status review (NMFS and USFWS 2013) and the Turtle Expert Working Group's report on Atlantic leatherback (TEWG 2007), and outcomes of the Pacific Leatherback Assessment Working Group (PLAWG 2012). There are three demographic populations in the Pacific identified through genetic studies (Dutton et al. 1999, 2007): 1) a western Pacific population that nests in Papua Barat Indonesia, Papua New Guinea (PNG), Solomon Islands and Vanuatu, 2) an Eastern Pacific population that nests in Mexico and Costa Rica, and 3) a Malaysian population. All leatherback interactions with the Hawaii-based shallow-set longline fishery have been of western Pacific stock origin (Dutton unpublished) and all but one of the leatherbacks observed caught in the deep-set fishery have been from the western Pacific. In 1995 there was a juvenile leatherback caught south of the MHI that was from the E. Pacific population.

Western Pacific Leatherback Turtles

The Western Pacific leatherback subpopulation nests primarily in Indonesia, Papua New Guinea (PNG), Solomon Islands, and to a lesser extent in Vanuatu (Dutton et al. 2007). The marine habitat for this subpopulation extends north into the Sea of Japan, northeast and east into the North Pacific to the west coast of North America, 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). This western Pacific leatherback metapopulation harbors the last remaining nesting aggregation of significant size in the Pacific (Dutton et al. 2007, Hitipeuw et al. 2007) with approximately 75 percent of regional nesting occurring along the northwest coast of Papua

Barat, Indonesia (also known as the Bird's Head Peninsula) (Hitipieuw et al. 2007). Genetic results to date have found that nesting aggregations that comprise the western Pacific population all belong to a single stock (Dutton et al. 2007), and in 2007 this population was estimated to consist of approximately 2700–4500 breeding females (Dutton et al. 2007). Recent research indicates a continual and significant long term nesting decline of 5.9 percent per year at the primary western Pacific beaches in Indonesia, Jamursba medi and Wermon, since 1984 (Tapilatu et al. 2013). The once large nesting population in Terrengannu, Malaysia, is now functionally extinct (Chan and Liew 1995; NMFS and USFWS 2013).

The Bird's Head region consists of four main beaches, three that make up the Jamursba-Medi (JM) beach complex and a fourth which is Wermon beach (Dutton et al. 2007). Currently, it is estimated that approximately 500 females nest annually at these primary nesting sites of the Bird's Head Peninsula (Tapilatu et al. 2013). Satellite tagging studies of leatherbacks from this metapopulation indicate that turtles that nest during different times of the year have different migration patterns. Summer nesting turtles (July through September) have tropical and temperate northern hemisphere foraging regions, while winter (November through February) nesters traverse to tropical waters and temperate regions of the southern hemisphere (Benson et al. 2011). Turtles nesting in Papua Barat, Indonesia during the summer months migrate through waters of Malaysia, Philippines, and Japan, across the Pacific past Hawaii to foraging grounds in temperate waters off North America (Figure 7) (Benson et al. 2007a, b; Benson et al. 2011). This Bird's Head nesting population exhibits strong site fidelity to the central California foraging area (Benson et al. 2011) which puts them at risk of interacting with Hawaii-based 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 western Pacific Ocean nesting beaches (Benson et al. 2011).

All 50 leatherbacks sampled so far as bycatch in the Hawaii-based shallow-set longline fishery are from the western Pacific population, based on genetic analyses. One of the 24 leatherbacks sampled so far as bycatch in the deep-set fishery, was determined to be from the eastern Pacific population (Table 2), which represents 4 percent of the sampled leatherbacks. This interaction occurred in 1995 south of the MHI at 14°N latitude and 157° W longitude. One hundred percent of sampled leatherbacks have been from the western Pacific population since 2004 (Table 3). 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), whereas western Pacific females migrate northward to the North Pacific after nesting in Papua Barat (Benson et al. 2007a, b; Benson et al. 2011). A study of 46 tagged leatherbacks tracked over 12,095 cumulative tracking days demonstrated that eastern Pacific leatherbacks main migration corridor is south and to the east of the action area after nesting (Shillinger et al. 2008). Bycatch data in Peruvian and Chilean coastal artisanal fisheries indicate leatherbacks are present in coastal areas (Alfaro-Shigueto et al. 2011; Donoso et al. 2000;) and genetic analyses of juvenile and adult leatherbacks caught in fisheries off Peru and Chile also show a proportion originate from the western Pacific Ocean rookeries (NMFS and USFWS 2013). The Hawaii deep-set fishery interacts mostly with juvenile leatherback turtles (Van Houtan 2013), in waters south of the MHI (NMFS unpublished observer data).

Bird’s Head Component of the Western Pacific Population

Jamursba-Medi (JM) makes up approximately 38 percent of total estimated nesting for the western Pacific population (Dutton et al. 2007). Jamursba-Medi is comprised of three beaches that are monitored together as an index beaches. The other main beach on the north coast of the Bird’s Head peninsula is Wermon. There is also some additional nesting at beaches of the Manokawari region of the Birds Head peninsula. Due to seasonal patterns of beach erosion, nesting occurs primarily at Wermon during the winter months (November to February) when the three beaches at Jamursba-Medi are gone. During the summer months (May to September), the three beaches of JM are built up again and leatherbacks nest there at that time (Hitipieuw et al. 2007). Nest data were not collected consistently or reliably until the early 1990s, hence most reports of Jamursba-Medi nesting trends start at that time. However, anecdotal reports from the early 1980s suggest that nesting at Jamursba-Medi declined during the decade preceding initiation of nest counts in 1993 (Dutton et al. 2007, Hitipeuw et al. 2007). Recent published information indicates a long-term nesting decline at Jamursba-Medi by 29 percent from 2005 to 2011 and a 78 percent decline over the 27 year period since 1984 (Tapilatu et al. 2013). However a study done in 2011, using the same nesting data in a climate-based population viability assessment that considers bottom-up climate forcing at two life stages, neonates and breeding females, shows a future increase in the population due to favorable climate conditions (Van Houtan 2011).

Wermon beach produced approximately 30 percent of all western Pacific nests from 1999-2006 (Dutton et al. 2007). Peak leatherback nesting at Wermon occurs between November and March, with some variable levels of nesting in the summer, although significantly lower than Jamursba-Medi (Wurlianty and Hitipeuw 2007, Hitipeuw et al. 2007, Tapilatu et al. 2013). Winter post-nesting females from Wermon migrated westward around Bird’s Head Peninsula and then south into the Halmahera, Ceram or Banda Seas, or moved along the north side of New Guinea and then southeast into waters of the western south Pacific Ocean and Tasman Sea, whereas summer post-nesting females from Jamursba-Medi headed to the temperate North Pacific Ocean or into tropical waters of the South China Sea (Benson et al. 2011). Anecdotal information indicates that there may be a small number of animals that utilize both Wermon and Jamursba-Medi beaches during a nesting season (Tapilatu, pers. comm.; Benson, pers. comm. 2011), yet Tapilatu et al. (2013) did not confirm evidence of cross utilization of beaches.

Nesting data from Jamursba-Medi are highly variable from year to year, and no data are available from 1998 due to a lack of survey effort that year. For the 17-year period 1993-2010, nesting fluctuated annually, with the overall trend declining by a rate of 5.5 percent per year (Tapilatu et al. 2013). The total number of nests per year for the Jamursba-Medi leatherback 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). Tapilatu et al. (2013) estimated the clutch frequency at JM to be 3-10 nests per season with a mean of 5.5 +/- 1.6. This is in line with other clutch frequency ranges from Costa Rica (5.6 +/- 1.2 nests per female; Reina et al. 2002) and Mexico (5.5 +/- 1.9 nests per female; Sarti Martinez et al. 2007). Based on nest counts, approximately 290 females nested at JM during the 2011 April to September boreal summer nesting season. Given the number of nests laid, between 1,195-1,575 and 275-363 females nested per year between the 1996 high and 2011 low at JM. Nesting beach monitoring at Wermon began in November 2002 and ran through June 2003 with 1,788 nests recorded (Hitipeuw et al.

2007). Monitoring was conducted again from November 2003 through September 2004 which resulted in the highest number of nests recorded for 2003-04 totaling 2,881 nests (Hitipeuw et al. 2007). Monitoring resumed in November 2004 and continued year round thereafter. Nesting declined during 2005 to approximately 1,300 nests (Wurlianty and Hitipeuw 2007), although Tapilatu et al. (2013) report 1,497 nests laid at Wermon in 2005. Since 2005, nesting has declined by 11.6 percent per year at Wermon to 1,096 nests laid in 2011, representative of 189-249 females, or a 62 percent decline since monitoring began in 2002 (Tapilatu et al. 2013). It is possible there may have been a learning curve to overcome by the community-based rangers in the early years. Hence it is unknown if the apparent decline in nesting activity is due to such a learning curve, is an actual decline in nesting activity, or is representative of typical annual sea turtle nesting variability. Continued long-term standardized monitoring at Wermon will help quantify trends. Impacts from threats to the Wermon nesting aggregation are consistent with those occurring at Jamursba-Medi, although the mean hatching success rates are higher (e.g., Jamursba-Medi at 25.5 percent and Wermon at 47.1 percent) (Bellagio Steering Committee, 2008). In the Manokwari region, West Papua, Indonesia, nesting occurs year round and the number of nests recorded from 2008 through 2011 ranged from 84 to 135 (Suganuma et al. 2012), however survey effort was limited and not consistent across years. The total number of adult females in the Bird's head region is estimated to be 1,949 based on summer nests (April – September) by Van Houtan (2013) (from Tapilatu et al. 2013). In 2011 an analysis of nesting data was done using two different models for a population viability assessment (PVA), a classical model and a climate-based model (Van Houtan 2011, NMFS 2012). The classical PVA calculated population growth and its variability from time series of nest counts, and this model resulted in a population decrease over the next 100 years. The second approach was the climate-based PVA that considers bottom-up climate forcing at two life stages, neonates and breeding females. The Climate-based PVA forecasted a population increase over within one generation (25 years) (Van Houtan 2011, NMFS 2012). Although the model is considered highly accurate based on its ability to account for historical population changes, because of the difficulty predicting PDO, the model cannot forecast population trends beyond one generation. While both the classical and climate-based approaches have limitations, the climate-based model is considered more rigorous in applying actual data, and therefore is more useful for predicting population trends.

Non-Bird's Head Component of the Western Pacific Population

Besides the Bird's Head region, Dutton et al. (2007) reported leatherback nesting at 27 other sites in the western Pacific region (6 in Papua Barat, 10 in PNG, 8 in the Solomon Islands, and 3 in Vanuatu). Approximately 62 percent of leatherbacks nesting in 1999-2006 occurred at these 27 sites. Of the total western Pacific nesting metapopulation, 20 percent of nesting activity from 1999-2006 occurred in PNG, with the Huon Coast contributing approximately 15 percent of total western Pacific population nesting activity (Dutton et al. 2007). Within PNG, the Huon Coast hosts an estimated 50 percent of leatherback nesting in that country which occurs primarily between November and March (Benson et al. 2007). Nesting also occurs on Bougainville, the south coast of West New Britain Province and the north coast of the Madang Province (Benson et al. 2007). Benson et al. (2007) aerial surveys recorded 58 nests on Bougainville. In January 2009, an expedition to Bougainville Island to survey beaches identified 46 leatherback nests during the peak nesting period with a high level (83-100 percent) of nest harvest and relatively frequent harvest of adult leatherback turtles (Kinch et al. 2009). Post-nesting females satellite tagged in PNG migrated into the southern hemisphere, southward through the Coral Sea, into waters of the western south Pacific Ocean (Benson et al. 2011).

Anecdotal information in Quinn et al. (1983), Quinn and Kojis (1985), and Bedding and Lockhart (1989) suggest that 200 to 300 females nested annually between Labu Tali and Busama on the Huon Coast in the late 1980s (Hirth et al. 1993). The average remigration interval (period since last nesting season along the Huon coast) is 3 years, with substantial variation ranging from 1 to 7 years (Pilcher 2010a). During the 2010-11 nesting season, 79 leatherback turtles nested laying a total of 527 nests (Pilcher 2011). Of these females, 30 were remigrants (turtles from previous seasons), 15 were new turtles never tagged before, and 34 were renesting events for turtles already identified previously in the season (Pilcher 2011). Between 2003 and 2006 the Huon Coast Leatherback Turtle Conservation Program (HCLTCP) expanded to incorporate more nesting habitat at the Kamiali nesting area and six additional communities. As a result, nesting trends are reflective of increased monitoring effort. The most reliable trend information begins from the 2006 - 2007 nesting season, and since then nesting activity has been stable or slightly increasing (Pilcher 2011). During the 2011-2012 nesting season, monitoring efforts were reduced due to funding limitations and the number of nests recorded during the season dropped to 193. Due to the change in monitoring effort, nest counts from 2010-2011 and 2011-2012 cannot be directly compared. Monitoring efforts were restored during the 2012-2013 nesting season which resulted in 211 tracks recorded between October 2012 and February 2013 (Pilcher 2013). However, monitoring activities again halted during the 2013-2014 nesting season due to community discord and disagreements. Of nests laid in PNG during the 2012-2013 nesting season, 22 percent were lost to erosion, poaching or did not hatch. Overall, however, total nest counts for these years reflect a decline of approximately 93 percent in nesting activity since 1980 estimates (Pilcher 2009, 2013).

The Solomon Islands support leatherback nesting (Bellagio Steering Committee 2008) that 30 years ago was widely distributed across at least 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 percent of the total western Pacific 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 activities in these primary locations occur during November to March, although there are reports of nesting from May to August both within the Solomon Islands and PNG that warrant further investigation. Additionally, one of 37 foraging leatherbacks outfitted with a satellite transmitter in California waters migrated to the Solomon Islands and nested at Santa Isabel Island in May providing additional evidence of a summer breeding population linkage between the western Pacific region and California foraging habitats (Benson et al. 2011).

Nesting beach monitoring began in 1993 at Sasokolo by the Department of Fisheries where an average of 25 nesting females deposit approximately 100 nests per season (Ramohia et al. 2001, Pita 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 turtle nests are laid seasonally (MacKay 2005, Goby et al. 2010, Pilcher 2010b).

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. No information exists regarding populations trends over time, but it is believed that local consumption of turtles and eggs has reduced nesting populations over the last few decades (Bellagio Steering Committee 2008, NMFS 2008a).

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

Summary for Western Pacific Population

Population estimates for sea turtles are problematic due to a lack of demographic information. No population estimates are available for the Pacific leatherback populations, only nesting information, which is used as population indices. In 2007, the total number of adult females in the Jamursba-Medi population was estimated to be within the range of 2,110 – 5,735 adult females, and at that time that the Jamursba–Medi component represented approximately 38 percent of the population (Dutton et al. 2007). Resent research indicates that nesting trends in Papua Barat, Indonesia have declined and that approximately 500 females are nesting annually in the Bird’s Head region at JM and Wermon beaches. The total number of adult females in the Bird’s head region is estimated to be 1,949 based on summer nests (April –September) (Van Houtan 2014, Tapilatu et al. 2013). This represents about 75 percent of the nesting activity in the Western Pacific, therefore NMFS estimates that there are approximately 2599 nesting females in the region.

Eastern Pacific Leatherback Turtles

Of interactions in the Hawaii deep-set fishery, one interaction occurred with a leatherback turtle in 1995 from the eastern Pacific population. This interaction occurred south of Hawaii at 14° N latitude and 157° W longitude. The eastern Pacific leatherback population was recently identified as one of the eleven most endangered regional management units in the world (Wallace et al. 2011), where remnant populations occur along the Pacific coast of Mexico, Costa Rica, Nicaragua, and Baja California peninsula, Mexico (Eckert et al. 2012, NMFS and USFWS 2013).

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 percent 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).

Currently, 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 percent 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 percent and 25 percent in the five or six years following tagging). Spotila et al. (2000) calculated a mean annual mortality rate of 35 percent for leatherbacks nesting at Las Baulas, and revised to 22 percent by Santidrian Tomillo et al. (2007). For comparison, at St. Croix, US Virgin Islands nesting grounds, female leatherbacks returned approximately 60 percent over the same period with a mortality rate of 11 percent (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 percent CI: 0.75–0.80) or 22 percent 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 percent 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 percent 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 percent ± 2.5 percent), embryonic death was the main cause of low hatchling success in this population.

Nicaragua

Nicaragua encompasses the third leatherback nesting concentration in the eastern Pacific after Mexico and Costa Rica. Primary nesting beaches in Nicaragua include Veracruze, Jan Venado and Salamina with Veracruz having the longest time series of monitoring beginning in 2002. At all of these beaches, 100 percent of nests were harvested prior to implementation of the nesting beach program. An aerial survey conducted during the 1998-1999 season estimated a nesting density in Playa El Mogote of only 0.72 turtles per kilometer (IAC 2004). During the 2005-2006 nesting season, 488 nests were recorded during aerial surveys (Barragan 2006 in Urteaga et al. 2012). Between 2002 and 2010, a total of 420 leatherback nests were recorded and 48 individual females were identified (Urteaga et al. 2012). Of nests laid, approximately 94 percent 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.

Summary for Eastern Pacific Population

The precipitous decline of the eastern Pacific leatherback population during the past two decades has been extensively documented, and was recently 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 percent of all EP leatherback nesting concluded that nesting has declined more than 90 percent since the 1980s (Spotila et al. 2000, Pritchard 1982, MTSG 2012), from thousands of nesting females per year to no more than 1,000 adult females total in the population, with approximately 150 to 200 females nesting annually per year at primary and secondary nesting beaches (NMFS and USFWS 2013, MTSG 2012).

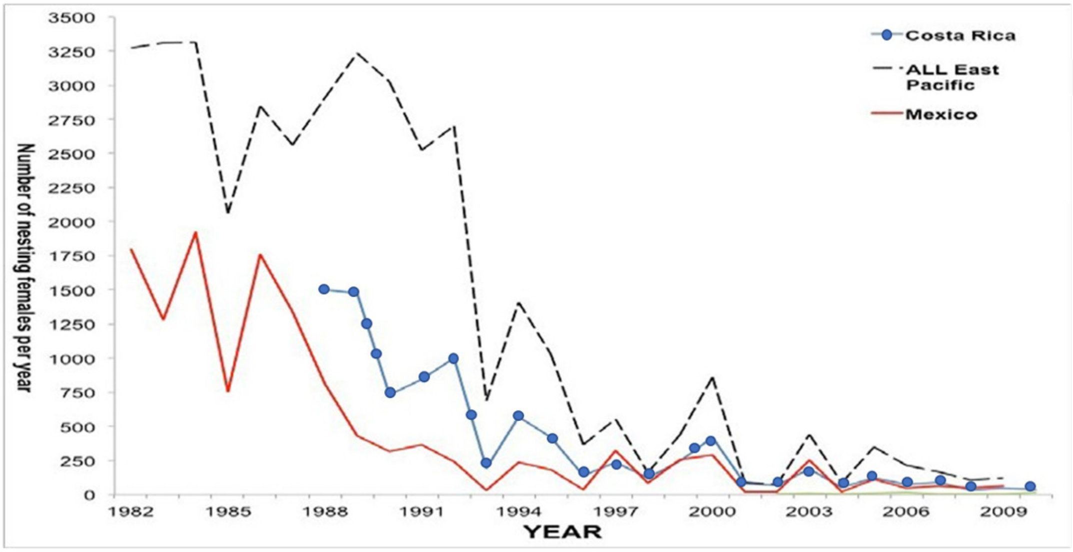

Figure 6. Nesting population abundance (shown in number of nests per year) at long-term (i.e. >10 years) index sites for the East Pacific leatherback subpopulation (MTSG 2012).

5.2.5.2 Life History Characteristics Affecting Vulnerability to Proposed Action

Leatherback life history is characterized by juvenile and adult life history stages occurring primarily in the oceanic zone. Adult leatherbacks range more widely across oceanic habitat than any other reptile, including into subpolar waters (NMFS 2004a, 2005a, 2006a; NMFS and USFWS 2013). Recent tagging studies have shown that adults sometimes migrate to highly productive upwelling areas near continental shelves, such as off Oregon and Washington (Benson et al. 2007a, 2011). Much less is known about juvenile and sub-adult leatherbacks, which is the life stage that the Hawaii deep-set interacts with predominantly (Van Houtan 2013). Given that the action area is oceanic, the main aspects of western Pacific leatherback life history affecting their vulnerability to Hawaii-based deep-set longline fishing are migration and foraging behavior, as discussed below.

In recent years, nesting females of the western and eastern Pacific populations have been tagged with satellite-linked transmitters, allowing tracking of their post-nesting migration routes. A portion of western Pacific leatherbacks nesting during June through August in Papua Barat go northeast, passing through the action area on their way to productive temperate waters off the west coast of the U.S. (Benson et al. 2007a, 2011). In contrast, leatherbacks nesting during November through March in Papua Barat, PNG, and Solomon Islands migrated southeast after nesting, towards Australian and New Zealand waters (Benson et al. 2007b, 2011). Additionally, seven of the 25 turtles with tracks of sufficient duration (28 percent) sampled in California waters migrated westward, presumably towards western Pacific nesting beaches (Benson et al. 2011). Satellite tracking and stable isotope analysis suggests that western Pacific leatherbacks are likely comprised of demographically discrete foraging ‘‘units’’ (Benson et al. 2011, Seminoff et al. 2012).

Eastern Pacific leatherbacks migrate south to foraging areas off South America in the eastern tropical Pacific and the southeastern Pacific, which is outside of the range of the action area (Shillinger et al. 2008, Bailey et al. 2012). However, much less is known about the juvenile foraging areas and in 1995 a DNA sample confirmed an interaction with an eastern pacific juvenile turtle south of the MHI. 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). Although both populations have large ranges within the Pacific Ocean, there is no spatial overlap between the EP and WP leatherback turtle populations that have been tracked, both in their migratory pathways and foraging grounds (Figure 6; Bailey et al. 2012). There are also data sets on at-sea distribution that were collected via observers and fishers onboard fishing vessels in the eastern Pacific. The primary dataset available was developed by IATTC and shows a wide distribution of leatherback turtles throughout the eastern Pacific, ranging from the Gulf of California, Mexico to Peru (Figure 6 In: IATTC, 2012). However, genetic analyses of juvenile and adult leatherbacks caught in fisheries off Peru and Chile indicates that a proportion (approximately 16 percent of sampled turtles) are from western Pacific rookeries (NMFS and USFWS 2013). Migratory routes of non-breeding adult females, and of adult males, are less understood for western and eastern Pacific leatherbacks, although 10 males were tagged with satellite-linked transmitters at California foraging grounds. Movements were similar to those of female leatherbacks tagged off California (Benson et al. 2011).

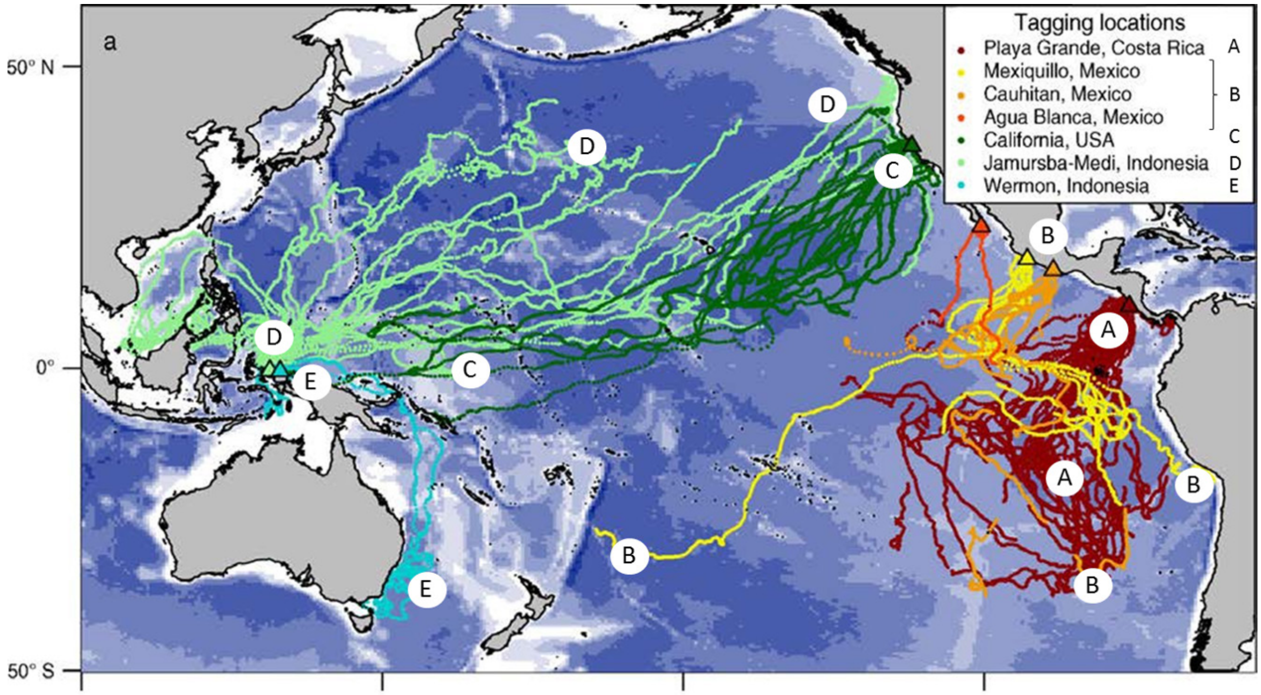

Figure 7. Satellite-tracked movements of 135 leatherback turtles in the Pacific Ocean coded by tagging location and overlaid on bathymetry (Figure from Bailey et al. 2012).

Leatherback age at maturity is uncertain. Estimates range widely between 9-15 years and 26-32 years based on skeletochronological analyses (see Jones et al. 2011 for review) 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 yr (Jones et al. 2011). Thus, a high degree of uncertainty remains about leatherback age at maturity in the wild. Likewise, leatherback lifespan is unknown.

Adult leatherbacks typically feed on pelagic soft-bodied animals, especially sea jellies, siphonophores, and tunicates. Despite the low nutritive value of their prey, leatherbacks 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. Leatherbacks 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 leatherbacks can dive deeper than any other reptile, most dives are less than 80 m (Shillinger et al. 2011). Migrating leatherbacks spend a majority of their time submerged and display a pattern of continual diving. Further, they appear to spend almost the entire portion of each dive traveling to and from maximum depth, suggesting continual foraging along the entire depth profile (NMFS 2006). Stable isotope analysis can complement satellite data of leatherback movements and identify important foraging areas that reflect regional food webs (Seminoff et al. 2012).

5.2.5.3 Threats to the Species

Global threats to leatherback turtles are spelled out in the 5-year review (NMFS and USFWS 2013), and threats to the western Pacific leatherback population are described in more detail in the proceedings of a 2004 leatherback workshop (WPFMC 2005), the Proceedings of the

Bellagio Sea Turtle Conservation Initiative (Bellagio Steering Committee 2008), and the PLAWG (2012).

Major threats to the species, according to these documents, are fisheries bycatch, alteration of nesting habitat, and direct harvest and predation, which are briefly described below. In addition, climate change and marine debris may be growing threats to this species, and are described below. Primary impacts to the western Pacific 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, NMFS and USFWS 2013, Bellagio Steering Committee 2008).

The annual survival probability of the PNG nesting population is estimated at 0.85 (95 percent CI: 0.66–0.95) which is lower than that of the Caribbean rookery 0.89) and the French Guiana rookery (0.91) (Pilcher and Chaloupka 2013). This survival probability may imply lower nesting beach fidelity, or might be reflective of higher mortality for the population due to anthropogenic hazards from subsistence hunting (Suarez and Starbird 1996) or incidental capture in fisheries (Lewison et al., 2004, Gilman et al. 2009, Wallace et al. 2010). Benson et al. (2011) found that post-nesting migrations of the PNG population headed southeast to the Eastern Australian Current and Tasman Front. Fisheries operating out of Australia and New Zealand are thought to result in high bycatch and high mortality rates of western Pacific leatherbacks that migrate there after nesting (MacKay 2014).

Before 2001 in the North Pacific, the Hawaii-based longline fishery was estimated to capture about 110 leatherbacks annually (McCracken 2000). If we apply the old mortality rate of 32 percent (Gilman 2007a), the result is an estimated mortality of 36 (110*32 percent= 35.2) annually before the shallow-set portion of the fishery was closed in 2001. The 2004 management measures have proven to reduce leatherback interaction rates by 83 percent (Gilman et al. 2007a, WPFMC 2009b). Since the shallow-set fishery re-opened in 2004, 17 (rounded up from 16.23) estimated leatherback mortalities occurred in the shallow-set fishery (NMFS 2014b). All of the leatherbacks caught were released alive; mortality estimates come from applying the NMFS post-hooking mortality criteria (Ryder et al. 2006) to interactions. The Hawaii-deep set fishery also interacts with leatherbacks and has an incidental take statement for up to 39 anticipated leatherback interactions and 18 anticipated mortalities over a three year period (NMFS 2005a). Until recently the level of annual interactions has remained well below this anticipated level. During the 2013-2014 winter season, December-March an unusually large number of interactions were observed in the fishery. During this time, six interactions were observed and in all other years less than 3 were observed with most years only having one or two interactions. The reason for the higher level of interactions is unknown; five of the six interactions occurred well south of the MHI. Fishing in the same area has occurred historically but with much lower interaction rates. It is possible that leatherbacks were congregating in eddies thus exposed more to fishing gear in a smaller area.

In the deep-set fishery from 2005-2014, 45 (rounded up from 44.49) mortalities are estimated to have occurred (NMFS 2014a). Since 2005, the Hawaii-based longline fisheries combined have reduced their estimated mortality to seven annually (NMFS 2014 a, b, c). However, other longline fisheries operating out of other countries are still using traditional methods (J style hooks with squid bait), and are likely killing at least hundreds of leatherbacks annually in the Pacific. The California Oregon drift gillnet fishery has an incidental take statement for up to 10 anticipated leatherback interactions and 7 anticipated estimated mortalities over a 5-year period (NMFS 2013a). Since 1990, 25 leatherbacks were observed taken and 16 of those were considered mortalities in the drift gillnet fishery (NMFS 2013a).

Destruction and alteration of leatherback 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 turtles, including those for the western Pacific population, occur in remote areas where development as described above is less prevalent although timber harvest, road construction, and village sprawl remain an issue in these remote areas (NMFS and USFWS 2013, Bellagio Steering Committee 2008 PLAWG 2012).

Harvest of leatherbacks 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 in the western Pacific egg harvest continues throughout the species' range, 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 western and eastern Pacific leatherbacks, for example by feral pigs in Papua Barat and feral dogs in PNG (NMFS and USFWS 2013, Bellagio Steering Committee 2008). Impacts and threats to leatherback 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 USFWS 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 leatherbacks 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 leatherbacks are opportunistically hunted for meat in some areas and leatherback eggs are occasionally collected from these beaches (Bellagio Steering Committee 2008, NMFS 2008a).

Marine debris is also a source of concern for leatherbacks due to the reasons described for loggerheads. Leatherbacks can ingest small debris and larger debris can entangle animals leading to death. For leatherbacks the greatest risk is in the pelagic environment but there is no information to quantify what the impacts are.

Although leatherbacks are probably already beginning to be affected by impacts associated with anthropogenic climate change in several ways, no significant climate change-related impacts to leatherback turtle populations in the western Pacific 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). In the same study described above for loggerheads, western Pacific leatherbacks show an increasing trend in the population over the next 22 years. The PDO was used to provide insight into neonate survival which was the same for loggerheads but instead of SST being used as a breeding cue for adult females, the ocean coastal upwelling index that describes the California Current dynamics was used (Van Houtan 2011). This study found that changes in leatherback nesting populations over the last approximately 20 years are correlated with ocean oscillations (PDO) due to environmental influences on juvenile recruitment. In the next 22 years, it is believed that leatherbacks in the western Pacific may increase by 82 percent due to favorable conditions in the PDO in recent years. Beyond 22 years we do not have information to predict what the population will do (Van Houtan 2011, NMFS 2012a). The study discussed above 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 leatherbacks. Increases in their primary prey source, sea jellies, due to ocean warming and other factors (Brodeur et al. 1999, Attrill et al. 2007, Richardson et al. 2009) although there is no evidence that any leatherback populations are currently food-limited. Even though there may be a benefit to leatherbacks 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 percent 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° C warming of the nesting beach was the primary driver of the modeled decline through reduced hatching success and hatchling emergence rates (Saba et al. 2012). Furthermore, climate change may compound the effects of interannual climate variability, as governed by El Niño Southern Oscillation (ENSO). Saba et al. (2007) showed that nesting females in Costa Rica exhibited a strong sensitivity to ENSO where as cool La Niña events correspond with a higher remigration probability and warm El Niño events correspond with a lower remigration probability. As a result, productivity at leatherback foraging areas in the eastern Pacific in response to El Niño/La Niña events result in variable remigration intervals and thus variable annual egg production. This phenomenon may render the eastern Pacific leatherback population more vulnerable to anthropogenic mortality due to longer exposure to fisheries than other populations (Saba et al. 2007).

Although the causes for decline of the eastern and western Pacific leatherback populations are not entirely clear, it is likely the result of historic intensive egg poaching on the nesting beaches, incidental capture of adults and juveniles in high seas fisheries, and natural fluctuations due to changing environmental conditions that influence prey abundance and distribution (Sarti Martinez et al. 2007; Santidrian Tomillo et al. 2007, 2008; Wallace et al., 2010; Saba et al., 2012). In Costa Rica the emergence of new threats from coastal development on key leatherback nesting areas present a serious challenge to efforts to protect leatherbacks in the East Pacific (Wallace and Piedra 2012, Tapilatu et al. 2013, NMFS and USFWS 2013), and egg consumption by humans and domestic animals (e.g., dogs) persist on Nicaragua nesting beaches where protection is incomplete (Urteaga et al. 2012). However, fisheries bycatch is still considered the major obstacle to population recovery (Wallace and Saba 2009, NMFS and USFWS 2013, MTSG 2012). Eckert and Sarti (1997) speculated that the swordfish gillnet fisheries in Peru and Chile contributed to the decline of the leatherback 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 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). A recent 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 subpopulation, followed by longlines (Wallace et al. 2013). Roe et al. (2014) highlight potential longline fishery bycatch hotspots in the Pacific that may affect populations at various stages of their life history. For western Pacific nesting populations, several areas of high risk were identified in the north and central Pacific, but the greatest risk was adjacent to primary nesting beaches in tropical seas of Indo-Pacific islands, in the Sulu, Sulawesi, and South China Seas. For eastern Pacific nesting populations, the greatest risk was identified in the South Pacific Gyre.

5.2.5.4 Conservation of the Species

Considerable effort has been made since the 1980s to document and address leatherback bycatch in fisheries around the world. In the U.S., observer programs have been implemented in most federally-managed fisheries to collect bycatch data, and several strategies have been pursued to reduce both bycatch rates and post-hooking mortality. These include developing gear solutions to prevent or reduce capture (e.g., circle hooks) or to allow turtles to escape without harm (e.g., turtle exclusion devices, but may be too small for adult leatherbacks), implementing seasonal time-area closures to prevent fishing when turtles are congregated, modifying existing gear (e.g., reducing mesh size of gillnets), and developing and promoting Sea Turtle Handling Guidelines (NMFS and USFWS 2013). For example, switching to large circle hooks and mackerel bait in 2004 with complimentary fishery-based outreach and education resulted in an approximately 83 percent reduction in the leatherback interaction rate in the Hawaii shallow-set longline fishery (Gilman et al. 2007a, WPFMC 2009b). Protected species workshops are required by NMFS annually of all Hawaii longline vessel operators to provide refresher trainings on the proper sea turtle handling guidelines, among other things. PIR offices in particular, have supported a significant number of international fishery-based projects to identify and promote effective sea turtle bycatch mitigation measures (e.g., circle hooks) or other gear modifications. In the Pacific, such projects have occurred in: Indonesia, Vietnam, Papua New Guinea, Solomon Islands, Malaysia, Palau, Marshall Islands, Federated States of Micronesia, and throughout Latin

America in association with the IATTC. Much of this work has been coupled with capacity-building, training, and implementation of regional observer programs aimed to improve the quality of catch and bycatch information from international fleets to better address the requirements of RFMO Conservation and Management Measures (CMMs) (of the WCPFC and IATTC). NMFS together with other regional partners will continue working within the context of RFMOs and U.S. laws to modify and improve international sea turtle bycatch mitigation requirements.

The NMFS and partners have been involved in leatherback turtle research and conservation activities in the western Pacific for nearly a decade supporting projects to understand and bolster survivorship, reduce harvest or predation, and to address other priority actions identified in the U.S. Pacific Leatherback Turtle Recovery Plan (NMFS and USFWS 1998b). Efforts to recover leatherbacks 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. The NMFS continues to work toward achieving support and developing fruitful partnerships for leatherback 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 PIT (Passive Integrated Transponder) tagging as a necessary tool to determine annual nesting estimates, undertaking aerial surveys and satellite telemetry research to assess habitat use, and utilizing innovative molecular techniques (genetics and stable isotopes) to assess stock structure and connectivity.

Community-based village rangers at Wermon and Jamursba-Medi in Papua Barat have been hired over the past decade to collect population demographic data (tag turtles and record nesting activity). Through their presence on the beach, projects have been able to guard leatherback 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 percent of nests laid at Wermon beach were lost as a result of harvest (60 percent) or predation (40 percent) (Starbird and Suarez 1996). Therefore, as a result of monitoring efforts the Wermon project may have protected over 12,000 nests that have been laid since the project's inception (NMFS 2011b). Community support in the form of scholarships and church repairs has been provided to encourage local participation in leatherback 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 conservation, and workshops convened to help the leatherback conservation program to develop stronger ties between the program and communities (Gjertsen and Pakiding 2012). From 2003 to 2007, the WPFMC supported a project at the Kei Kecil Islands of Papua Barat Indonesia to assess and help reduce traditional harvest of adult leatherbacks in coastal foraging habitats. Starbird and Suarez (1996) estimated that this traditional fishery captured at least 100 leatherbacks per year, however, the Kei Islands project acquired a more accurate harvest estimate of less than 50 turtles per year with the majority being juveniles or subadults (Lawalata and Hitipeuw 2006). In July 2012, Kei Kecil was designated as a marine protected area. It is hoped that the designation, which was supported by a program of USAID aimed at conserving marine resources, will help to address threats posed by forest clearance near nesting beaches, fishing activities, and hunting of turtles for meat (IOSEA 2013).

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

In the Solomon Islands, a program has been initiated at Sasakolo and Litogarhira to relocate nests that would otherwise be destroyed by beach erosion, high sand temperatures, illegal harvest and predation in order to increase hatchling production (a collaborative project between SWFSC and The Nature Conservancy, with additional funding support from the International Sustainable Seafood Foundation, the Ocean Foundation, and NMFS). Additionally, the Tetepare Descendents Association (TDA) has closed 13 km of beach to harvest, continues to protect and monitor nests, and is obtaining training, guidance and encouragement through collaborations with relevant NMFS staff and funding. Further, efforts are currently underway to launch assessment and monitoring activities with communities that have summer nesting activities.

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

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. percentSince the Mexican government instituted protective measures, there has been greater nest protection and nest success. The most recent results (2011-2012 nesting season) indicate that nearly 90 percent 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 percent or 142 nests). On the priority II beaches during the 2011-12 nesting season, nest protection ranged from 70.6 percent to 78.8 percent (López et al. 2012). This is a significant increase since 1996, when only 12 percent of nests were relocated. From 1982 to 2004 a total of 270,129 leatherback 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'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 percent 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 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 percent 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 subpopulation remains perilously low, and continues to decrease slowly toward extinction (MTSG 2012). Reducing leatherback bycatch has become a primary focus for many conservation projects around the world, and some mitigation efforts are showing promise (Watson et al. 2005, Gilman et al. 2009, Wang et al. 2010). However, fisheries bycatch is still considered the major obstacle to population recovery (Wallace and Saba 2009, Wallace et al. 2013, NMFS and USFWS 2013).

The conservation and recovery of leatherback turtles is facilitated by a number of regulatory mechanisms at international, regional, national and local levels, such as the FAO Technical Consultation on Sea Turtle-Fishery Interactions, the Inter-American Convention for the Protection and Conservation of Sea Turtles, CITES, and others. In 2008 the WCPFC adopted a Conservation and Management Measure (CMM 2008-03) to mitigate the impacts on turtles from longline swordfish fisheries in the Western Central Pacific Ocean. The measure includes the adoption of FAO guidelines to reduce sea turtle mortality through safe handling practices and to reduce bycatch by implementing one of three methods by January 2010. The three methods to choose from are: 1) use only large circle hooks, 2) use whole finfish bait, or 3) use any other mitigation plan or activity that has been approved by the Commission. As a result of these designations and agreements, many intentional impacts on sea turtles have been reduced: harvest of eggs and adults have been reduced at several nesting areas through nesting beach conservation efforts and an increasing number of community-based initiatives are in place to reduce the take of turtles in foraging areas (Gilman et al. 2007b, NMFS and USFWS 2013).

5.2.6 Olive Ridley Turtles

Information in this section is summarized primarily from the 2004 BiOp (NMFS 2004a), the 2005 BiOp (NMFS 2005a), the 2006 BiOp (NMFS 2006), the 2008 BiOp (NMFS 2008a), the olive ridley 5-year status review (NMFS and USFWS 2014), the 2012 BiOp (NMFS 2012a), and other sources cited below.

5.2.6.1 Population Characteristics

Olive ridleys are the most abundant sea turtle species and are known for major nesting aggregations called *arribadas* with tens of thousands to over a million nests annually, the largest of which occur on the west coasts of Mexico and Costa Rica, and on the east coast of India. Minor *arribadas* and solitary nesters are found throughout the remaining tropical and warm temperate areas of the world, including the western Pacific. 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 USFWS 2014). The eastern Pacific population includes nesting aggregations on the coast of Mexico, which are listed under the ESA as endangered. All other olive ridley populations are listed as threatened (Table 1). In 2014, the NMFS and USFWS (Services) completed a status review of the olive ridley sea turtle. Based on the best available information, the Services concluded the olive ridley sea turtle breeding colony populations on the Pacific coast of Mexico may warrant reclassification (NMFS and USFWS 2014). No change to the threatened populations is recommended. However, for the current population listings for the olive ridley (both endangered and threatened), information indicates an analysis and review of the species should be conducted in the future to determine the application of the DPS policy.

The eastern Pacific population is thought to be increasing, while there is inadequate information to suggest trends for other populations. The global status of olive ridleys is described in the 5-year status review (NMFS and USFWS 2014). Olive ridleys are the most common turtle species that interacts with the Hawaii-based deep-set longline fishery. In the deep-set fishery, 106 interactions have occurred between 1995 and 2014, of which 82 were from the eastern Pacific (77 percent) and 24 were from the western Pacific (23 percent) (Table 2), which is comprised of turtles that are genetically similar to turtles with haplotypes identified in Sri Lanka, Malaysia and India (Dutton pers. comm.). In the shallow-set fishery, fourteen genetic samples have been collected and analyzed since 1995; eight were from the eastern Pacific population and six were from the western Pacific population (Table 2).

Eastern Pacific olive ridleys nest primarily in large *arribadas* on the west coasts of Mexico and Costa Rica. Since reduction or cessation of egg and turtle harvest in both countries 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 (NMFS and USFWS 2014). 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 (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). The NMFS and USFWS (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, then 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 USFWS 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, gass exchange, and ammonica 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 USFWS 2014).

In the western Pacific, olive ridleys are solitary nesters and typically occur in tropical and warm temperate waters from Australia through southeast Asia (NMFS and USFWS 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 (NMFS and USFWS 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 USFWS 2014). Survey effort on India 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 USFWS 2014). During the 2012 nesting season, an estimated 100,000 olive ridleys 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 2009a). Data are not available to analyze trends (NMFS 2005a, NMFS and USFWS 2014).

Western Pacific Non-Arribada Beaches

In Indonesia, olive ridleys 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 USFWS 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 ridleys 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 USFWS 2014). In Australia, olive ridley nesting is scattered throughout northern Australia, with a few thousand females nesting annually (Limpus 2009a). 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 2009a, NMFS and USFWS 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 2009a).

Several methods have been used over time to estimate the number of turtles nesting during an arribada (NMFS and USFWS 2014). The olive ridley abundance estimates presented in Table 5 and adapted from NMFS and USFWS (2014) and other sources were likely 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. However, 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. It is unclear if adjustments have been made to account for multiple nesting events or arribadas, and the number of nesting females in locations is presented as a range in arribada size (e.g., 3,564 to 476,550 at Ostional), therefore the information in Table 5 are provided for a general sense of arribada size and not intended to be exact estimates of the nesting population.

The once large nesting populations of olive ridleys that occurred in peninsular Malaysia and Thailand have been decimated through long term over-harvest of eggs (Limpus 2009a). 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 2009a). While the Australian olive ridley nesting distribution and population size remains to be fully evaluated, a few thousand females may nest annually in the Northern Territory (Limpus 2009a). 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 2009a).

Table 5. Annual olive ridley population estimates at major arribada nesting sites

Location	Average number per arribada (population status)	Reference
La Escobilla, Mexico	1,021,500 – 1,206,000 nests 1,013,034 females (increasing)	NMFS and USFWS 2007c; NMFS and USFWS 2014
Ixtapilla, Mexico	2,900 – 10,000 nests (increasing)	NMFS and USFWS 2014
Moro Ayuta	10,000 – 100,000 nests (stable)	NMFS and USFWS 2014
Ostional, Costa Rica	3,564 to 476,550 females (increasing but declining recently)	Valverde et al., 2012; NMFS and USFWS 2014
Playa Nancite, Costa Rica	2,000-12,000 females; 256- 41,149 females (declining)	NMFS and USFWS 2007c; NMFS and USFWS 2014
Playa Chacocente, Nicaragua	42,500 nests 27,947 females (unknown)	NMFS and USFWS 2007c; NMFS and USFWS 2014
Playa La Flor, Nicaragua	521,440 females (stable)	NMFS and USFWS 2014
Isla Canas, Panama	8,768 females (declining)	NMFS and USFWS 2014
Gahirmatha, India (Orissa)	1,000 - 100,000 females and 10,000 - 250,000 (stable or declining)	NMFS and USFWS 2007c; IOSEA, 2013; NMFS and USFWS 2014
Rushikulya, India	23,561 – 172,402 females (stable)	NMFS and USFWS 2014

5.2.6.2 Life History Characteristics Affecting Vulnerability to Proposed Action

Life history of Pacific olive ridleys is characterized by juvenile and adult stages occurring in the oceanic zone. Along with leatherbacks, olive ridleys are the most pelagic of all sea turtle species (NMFS 2004a, 2005a, 2006a, 2008a; NMFS and USFWS 2014). Olive ridleys appear to have the shortest age to maturity at approximately 13 years of age (Zug et al. 2006). The Hawaii-based deep-set longline fishery encounters olive ridleys at rates greater than any other sea turtle due to the area of operation of the fishery and the diving behavior and distribution of the species as discussed below.

Olive ridleys occupy marine ecosystems that occur over vast areas and are considered nomadic in the eastern Pacific (Plotkin 2010). 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 ridleys appear to forage throughout the eastern tropical Pacific Ocean, often in large groups, or flotillas, and are occasionally found associated with floating debris (Arenas and Hall 1992 in NMFS and USFWS 2014). The direct impact of El Niños on olive ridleys is unknown, but olive ridleys appear to change migration pathways in response to shifts in food availability during El Niño (Plotkin 2010).

Polovina *et al.* (2003, 2004) tracked 10 olive ridleys caught in the Hawaii-based pelagic longline fishery. The olive ridleys 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 ridleys identified from the western Pacific associated with these major currents, suggesting that olive ridleys 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 ridleys revealed that turtles 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. Eguchi et al. (2007) estimated the density and abundance of the olive ridley 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 (MCP: 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 ridleys was 177,617 (CI: 150,762-204,471, CV: 17.2 percent, 95 percent), with the highest abundance recorded in winter in the oceanic region of Jalisco (N: 181,150, CI: 117,150-280,110, CV:21.4 percent). Martín del Campo et al. (2014) conclude that olive ridleys 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.

Olive ridleys 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 ridleys typically forage in deep water, often diving within the range that deep-set gear is fished. In addition, the distribution of this species in the North Pacific tends to be within the action area for the Hawaii-based deep-set longline fishery (Polovina et al. 2003, 2004; NMFS 2006).

5.2.6.3 Threats to the Species

Global threats to olive ridley turtles are spelled out in the 5-year status review (NMFS and USFWS 2014). Major threats to the species are 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, 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 USFWS 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 ridleys 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 adults and eggs was reduced in the late 1980s and early 1990s, but legal harvest of eggs continues in some parts of the species’ range, such as in Ostional, Costa Rica. Illegal harvest of eggs is common in much of the species’ range, such as throughout Central America, Western Pacific, and India (NMFS and USFWS 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 analysis of data that showed that 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. In relation to the estimated number of clutches laid, the estimated mean of clutches harvested was 21.2 percent (ranging from 1.5 percent to 102.4 percent- the percentage harvest of 102.4 percent resulted from the mathematical conversion of eggs to nests and the error in the estimated number of clutches laid. As per Valverde et al. (2012), this value suggests nearly complete egg harvest). 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 percent), high clutch destruction rates, and low recruitment (Valverde et al. 2012).

A major threat to olive ridley turtles is believed to be 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 ridleys that tens of thousands of dead adults wash up on the coast annually (NMFS and USFWS 2014). 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 ridleys annually (NMFS and USWFS 2014). Trawlers in Costa Rica are reported to catch over 15,000 sea turtles annually, and 90 percent of those are olive ridleys (Arauz et al. 1998). As s 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 ridleys caught, including 92,300 adult females and 23,000 green turtles (Dapp et al. 2013). Artisanal gillnet and longline fisheries of Peru and Chile are known to interact with olive ridleys (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. Almost 6,000 sea turtles were estimated to be captured annually, of which 240 were olive ridleys (Alfaro-Shigueto et al. 2011). Threats to olive ridleys 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 2009a). The Hawaii-deep set fishery interacts with olive ridleys and had an incidental take statement for up to 121 anticipated olive ridley interactions and 117 anticipated mortalities over a three year period (NMFS 2005a). The interactions that have occurred in the fishery since then were slightly lower than anticipated. Between 2005 and 2014 there were 277 olive ridley interactions in the deep-set fishery and from this the estimated mortality is 264 (McCracken 2006, 2007, 2008, 2009a, 2009b, 2010, 2011, 2012, 2013, 2014a; NMFS 2014a). The Hawaii shallow-set fishery rarely interacts with olive ridleys and since 2004, only four have been incidentally caught. All four were released alive. The California Oregon drift gillnet fishery has an incidental take statement for up to 2 anticipated olive ridley interactions and 1 anticipated estimated mortality every five years (NMFS 2013a). Since 2001 no olive ridleys have been captured in the California Oregon drift gillnet fishery and only one has been observed since 1990 (NMFS 2013a).

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 ridleys 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 1994 and 2010 in NMFS and USFWS 2014). As with leatherback turtles nesting in the eastern Pacific, olive ridley’s 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 ridleys due to the same reasons described for loggerheads. Olive ridleys can ingest small debris and larger debris can entangle animals leading to death. For olive ridleys the greatest risk is when they are in the pelagic environment but there is no data to quantify what the impacts are.

5.2.6.4 Conservation of the Species

Since large-scale direct harvest of adult olive ridleys became illegal, conservation efforts have focused on reducing bycatch in fisheries, especially those operating near *arribadas* such as the Pacific coast of Mexico/Central America and the east coast of India. Some areas offshore of Central American *arribadas* are closed to fishing in order to reduce turtle bycatch (Frazier et al. 2007), and trawl fishing which was estimated to catch over 15,000 turtles per year (90 percent of which were olive ridleys), was banned in Costa Rica in September 2013 (Arias 2013). Likewise, no mechanized fishing is allowed within 20 km of the *arribada* in India, and turtle excluder devices are mandatory on trawlers operating out of Orissa state (Shanker et al. 2003). However, enforcement is reported to be lacking in both areas (Frazier et al. 2007, Shanker et al. 2003).

In India, the Odisha Government has enacted a seven-month ban (November 1 to May 31) restricting fishing near the Gahirmatha marine sanctuary in Kendrapara district along the 20 km stretch of the Dhamra-Rushikulya river mouth to protect nesting olive ridleys. An estimated 26,000 traditional marine fishermen in coastal Kendrapara and Jagatsinghpur districts are likely to be affected by the measure. Trawl operators are prohibited in the protected zone, and orders are being enforced with nearly 100 trawls and vessels were seized and their crew arrested during the ban in 2011 (The Hindu Business Line News 2011).

Between 2004 and 2007, the IATTC coordinated and implemented a circle hook exchange program to experimentally test and introduce circle hooks and safe handling measures to reduce sea turtle bycatch in mahi-mahi and tuna/billfish artisanal longline fisheries in Ecuador, Peru, Panama, Costa Rica, Guatemala, and El Salvador. Almost all (99 percent) of fishery/turtle interactions identified by this program were with green and olive ridley sea turtles. By the end of 2006, over 1.5 million J hooks had been exchanged for turtle-friendly circle hooks (approximately 100 boats). Overall, circle hooks were found to reduce interaction rates by 40 to 80 percent in artisanal fisheries that switched gear types, with deep hookings reduced by 20 to 50 percent. Experiments to reduce longline gear entanglements were also successful. This project ended in 2007 and no follow up study has been initiated to assess continued use of circle hooks or dehooking and safe handling methods in fisheries where these measures were introduced.

The conservation and recovery of olive ridleys is facilitated by a number of regulatory mechanisms at international, regional, national, and local levels, such as the Indian Ocean Southeast Asian Marine Turtle Memorandum of Understanding, the Inter-American Convention for the Protection and Conservation of Sea Turtles, CITES, and others. Within the WCPFC, NMFS has worked to modify and improve international bycatch mitigation requirements and aided in establishing a binding Sea Turtle Conservation Measure implementing the FAO Guidelines (e.g., circle hooks and safe handling measures) which has likely helped reduce interactions and improve survivorship in international longline fisheries. As a result of these designations and agreements, many of the intentional impacts on olive ridleys have been reduced: harvest of eggs and adults have been reduced at several nesting areas through nesting beach conservation efforts and an increasing number of community-based initiatives are in place to reduce the take of turtles in foraging areas (Gilman et al. 2007b, NMFS and USFWS 2014).

5.2.7 Green Turtles

Information in this section is summarized primarily from the 2004 BiOp (NMFS 2004a), the 2005 BiOp (NMFS 2005a), the 2006 BiOp (NMFS 2006), the 2008 BiOp (NMFS 2008a), the 2010 BiOp (NMFS 2010b), the 2012 BiOp (NMFS 2012a), the green turtle 5-year status review (NMFS and USFWS 2007b), and other sources cited below.

5.2.7.1 Population Characteristics

Green turtle populations occur in at least the western, central, and eastern Atlantic, the Mediterranean, the western, northern, and eastern Indian Ocean, Southeast Asia, and the western, central, and eastern Pacific, according to the 5-year review (NMFS and USFWS 2007b). According to genetic samples obtained from incidentally caught green turtles in the deep set fishery the turtles interacting with the deep-set fishery come from the eastern, central, and western pacific populations. The eastern Pacific population includes turtles that nest on the coast of Mexico, Galapagos, Revillagidos Islands, and El Salvador to Costa Rica. In the 5-year review, the only archipelago included in the central Pacific was Hawaii (NMFS and USFWS 2007b). The western Pacific region includes the rest of the Pacific not previously described. In order to address the effect of take of green sea turtles on nesting populations in the western Pacific Ocean region, we analyzed three geographically distinguishable nesting aggregations, the central west Pacific Ocean, the southwest Pacific Ocean, and the central south Pacific Ocean. For purposes of this analysis we define the central west Pacific Ocean to include nesting areas of Commonwealth of Northern Mariana Islands (CNMI), Federated States of Micronesia, Guam, Indonesia, Japan (e.g., Chichijima), Marshall Islands, Palau, Papua New Guinea, and the Solomon Islands. For purposes of this analysis we define the west southwest Pacific Ocean to include nesting areas of eastern Australia, Coral Sea, New Caledonia, and Vanuatu. For purposes of this analysis we define the central south Pacific Ocean to include nesting areas of American Samoa, Cook Islands, Fiji, French Polynesia, Kiribati, Tokelau, Tonga, Tuvalu, and United Kingdom Overseas Territory.

The Hawaii-based longline fisheries occasionally interact with green turtles. Nineteen green turtles have been sampled from the Hawaii deep-set longline fishery since 1995, 13 were from the eastern Pacific population, 2 were from the western Pacific population, 3 were from the Hawaii component of the central Pacific, and 1 was of indeterminate origin (from either Hawaii or eastern Pacific populations) (Table 2). Seven turtles have been sampled so far as bycatch in the shallow-set fishery; four were from the Hawaii component of the central Pacific population, while the other three were from the eastern Pacific population, based on genetic analyses (Table 2).

The Hawaii component nests exclusively in the Hawaiian Archipelago, with over 95 percent of nesting at French Frigate Shoals (FFS) in the Northwestern Hawaiian Islands (Kittinger et al. 2013). Adults migrate greater than 1,000 km between foraging areas in the Main Hawaiian Islands (MHI) and the FFS nesting area. Since initial nesting surveys at the FFS index beach in 1973, there has been a marked increase in annual green turtle nesting (Balazs and Chaloupka 2004). The increase over the last 30+ years corresponds to an underlying near-linear increase of about 5.7 percent per year (Chaloupka et al. 2007). Between 1973 and 2011, 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 unpublished). In 2012, nesting activity decreased, as is typical, to 439 females. The total number of nesting females in this aggregation is currently 3,846 (NMFS and USFWS in Press). Subsequent surveys in 2013 (and likely beyond for some time) will not be possible due to a December 2012 storm that destroyed the FFS field station making residence for biologists no longer possible due to unsafe conditions. In-water abundance of green turtles is consistent with the increase in nesting trends (Balazs 1996; Balazs and Chaloupka 2004; Chaloupka et al. 2007). In addition, there has been a dramatic increase in the number of basking turtles in the MHI and throughout the Northwestern Hawaiian Islands (Balazs 1996, Balazs and Whittow 1982, Parker and Balazs 2010). Long-term monitoring of the population indicates a strong degree of island fidelity exists within the rookery, and tagging studies have shown that turtles nesting at FFS come from numerous foraging areas where they reside throughout the Hawaiian Archipelago (Balazs 1976; Balazs 1980, 1983; Dutton et al. 2008). This linkage has been firmly established through genetics, satellite telemetry, flipper tagging and direct observation (Balazs 1983, 1994; Leroux et al. 2003; Dutton et al. 2008). More information is available on green turtle population and trends in the 5-year review (NMFS and USFWS 2007b). The increase of the long-term nester trend can be attributed to increased survivorship (since harvesting of turtles in foraging grounds was prohibited in the mid-1970s) and cessation of habitat damage at the FFS rookery since the early 1950s (Balazs and Chaloupka 2004).

Eastern Pacific green turtles nest on at least the coasts of Mexico and elsewhere in Central America, with major nesting rookeries in Michoacán, Mexico and the Galápagos Islands (NMFS and USFWS 2007b). Lesser nesting sites are found in the Revillagigedo Islands off central Mexico and several small beaches along the Central American coastline, particularly from El Salvador through northwest Costa Rica (SWOT 2011). An estimated 3,319 – 3,479 eastern Pacific females nested annually (NMFS and USFWS 2007b), and nesting has been steadily increasing at the primary nesting sites in Michoacan, Mexico, and in the Galapagos Islands since the 1990s (Delgado and Nichols 2005; Senko et al. 2011). Nesting trends at Colola have continued to increase since 2000 with the overall eastern Pacific green turtle population also increasing at other nesting beaches in the Galapagos and Costa Rica (Wallace et al. 2010, NMFS and USFWS 2007b). Based on nesting beach monitoring efforts, the current adult female nester population for Colola, Michocan is 11,588 females, which makes this the largest nesting aggregation in the East Pacific, comprising nearly 58 percent of the total adult female population. The total for the entire Eastern Pacific nesting aggregation is estimated at 20,112 nesting females (NMFS and USFWS in Press).

Central west Pacific Ocean greens nest in the areas of Commonwealth of Northern Mariana Islands (CNMI), Federated States of Micronesia, Guam, Indonesia, Japan (e.g., Chichijima), Marshall Islands, Palau, Papua New Guinea, and the Solomon Islands. The estimated number of nesting females for this area is 6,518 (Kolinski *et al.* 2006, Palacios 2012, Kolinski *et al.* 2004, Wenninger 2012, Kolinski *et al.* 2001, Maison *et al.* 2010, J.Cruce pers. comm. 2013, Naughton 1991, Pritchard 1995, Smith et al. 1991, Kolinski 1994, Guam Division of Aquatic and Wildlife Resources 2011and 2012, Dermawan 2002, Hitipeuwand and Maturbongs 2002, H. Suganuma pers. comm. 2012, Chaloupka *et al.* 2007, McCoy 2004, Thomas 1989, NMFS and US FWS 1998, Palau Bureau of Marine Resources 2005, Palau Bureau of Marine Resources 2008, Barr 2006, NFA 2007, NMFS and USFWS in Press). There is insufficient information to adequately describe abundance and population trends for many areas of the central west Pacific. Limited information suggests a nesting population decrease in some portions of this region like the Marshall Islands, or unknown trends in other areas. The nesting Chichijima, Japan has shown an increasing trend (NMFS and USFWS in Press).

Southwest Pacific Ocean greens nest in areas of eastern Australia, Coral Sea, New Caledonia, and Vanuatu. The estimated number of nesting females for this area is 83,058 (NMFS and USFWS in Press). Nesting occurs in many islands throughout the Southwest Pacific aggregation, but 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 interannual 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 2009b). 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 2009b). From the mid-1990s to the mid-2000s, there has been a leveling off of the rate of increase (Chaloupka et al. 2007). The Heron Island, Australia, index count is derived from a tagging census of the total annual nesting population. There was a 3 percent per year increase in annual nesting abundance in the subset of data from 1974–1998 (Chaloupka and Limpus 2001) and a similar 3.8 percent per year increase from the 1974–2002 subset (Chaloupka et al. 2007). 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 2009b). The increase in annual nesting females at Heron Island is concurrent with an estimated increase of 11 percent 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 Raine and Heron Islands nesting indices do not fully describe the productivity of this nesting aggregation as there is important ecological and demographic information that is not captured in the nesting index. There was a significant decrease in the late 1990s and early 2000s in the mean carapace size (CCL) of nesting females at Raine Island and Heron Island (Limpus et al. 2002, 2003, 2007; Limpus 2009b). Although this decrease is only a few centimeters or less, it could indicate important population-level changes including disproportionate adult mortality (including possible over harvest which could result in a declining population), several strong year-classes beginning to nest (possibly resulting in an increasing population), changes in mean size of nesting group, or changes in maturation time. Nesters at Raine Island show an increase in the mean observed remigration interval (Limpus et al. 2002), though it is important to note that observed remigration intervals are influenced by tagging effort in previous years. Given that the remigration interval of females returning for only their second season is longer than that for turtles that have nested during multiple prior seasons (i.e., older turtles), the observed increase in mean remigration interval further supports the notion that fewer large turtles are present in the population (Limpus et al. 2002). The decrease in size of nesters at Raine Island coupled with the pattern of increase in remigration intervals is consistent with a reduction of older turtles in the population and may be an early warning that the Raine Island nesting population may be in the early stages of decline (Limpus et al. 2002). There are additional concerns about the long-term health of the Raine Island nesting population (Limpus et al. 2003). Total productivity is limited by reduced nesting and hatchling success, which at Raine Island appears to be depressed due to habitat issues. For Raine Island, mean nesting success (i.e., probability that a clutch will be laid when a turtle comes ashore for a nesting attempt) can be as low as 3.3 percent (range=1.72 to 4.88, n=2; see Table 7 of Limpus et al. 2007). Reduced recruitment can be caused by flooding of egg chambers by ground water, dry collapsing sand around egg chambers, and underlying rock which prevents appropriately deep egg chambers (Limpus et al. 2003). Death of nesting females occurs at Raine Island. Nightly mortality ranges from 0 to over 70 per night and is highest when nesting the previous night exceeds 1,000 (Limpus et al. 2003).

Central south Pacific Ocean greens nest in areas of American Samoa, Cook Islands, Fiji, French Polynesia, Kiribati, Tokelau, Tonga, Tuvalu, and United Kingdom Overseas Territory. The estimated number of nesting females for this area is 2,902 (NMFS and USFWS in Press). Green turtle temporal population trends in the Central South Pacific are poorly understood, with not even a single nesting site having five contiguous years of standardized monitoring that span entire nesting seasons. Partial and inconsistent monitoring from the largest nesting site in this aggregation, 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 percent 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 these above trends, and the general dearth of long-term monitoring and data from this nesting aggregation, presents significant challenges to any trend analyses.

5.2.7.2 Life History Characteristics Affecting Vulnerability to Proposed Action

Green turtle life history is characterized by early development in the oceanic (pelagic) zone followed by later development in the coastal areas. After hatching, juveniles spend at least several years in pelagic areas where they feed primarily on small invertebrates. Between 6 to 10 years of age, at approximately 40 cm curved carapace length, most green turtles recruit to coastal habitats. However, one recent study has shown that some green sea turtles don’t recruit to nearshore areas until they are around 70 cm; curved carapace length, moving between open ocean areas and nearshore regions. This appears to be most prevalent with eastern Pacific greens (Parker et al. 2011).

Neritic green turtles typically forage in shallow coastal areas, primarily on algae and seagrass. Unlike other sea turtle species, upon maturation adults do not typically undertake trans-oceanic migrations to breeding sites, but long migrations may still occur between foraging and nesting areas, such as those undertaken by Hawaiian green turtles between the MHI and FFS (NMFS 2004a, 2005a, 2006a; NMFS and USFWS 2007b). Galapagos nesters 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). Green turtles have long been considered to be strongly tied to coastal areas with abundant seagrass and/or marine algae pastures. However, results of satellite telemetry work (Seminoff et al. 2008) with at-sea observations (IATTC 2012) indicate that many 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.

However, as described above in Section 4 (Description of Action Area), the proposed action includes waters of the Hawaiian Islands although longline fishing does not occur within 75 nm from the Main Hawaiian Islands. Adults migrate directly between the MHI and FFS (Balazs 1994), but the proposed action is unlikely to encounter many migrating adult green turtles from the Hawaii component of the central Pacific population. Green turtles from this region are expected to reach maturity at approximately 80 cm straight carapace length (SCL) (Zug et al. 2002). Since 2004, seven turtles have been observed incidentally caught in the deep-set fishery and all were less than 75 cm SCL (PIRO Observer program, unpublished data); two were from the W. Pacific and four were from the E. Pacific, and one was from the C. Pacific (Table 3). Therefore we expect that the main aspect of green turtle life history affecting their vulnerability to Hawaii-based deep-set longline fishing are juveniles and sub-adults utilizing oceanic habitats.

5.2.7.3 Threats to the Species

Global threats to green turtles are spelled out in the 5-year review (NMFS and USFWS 2007b). Major threats to the species, according to this document, are alteration of nesting and foraging habitat, fishing bycatch, and direct harvest, 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. The 5-year review also identifies disease and boat collisions as threats to green turtles in Hawaii.

Destruction and alteration of green turtle nesting and foraging habitats are occurring throughout the species' global range, especially coastal development, beach armoring, beachfront lighting, and vehicular/ pedestrian traffic. While under natural conditions beaches can move landward or seaward with fluctuations in sea level, extensive shoreline hardening (e.g., seawalls) inhibits this natural process. 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 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. Sea level rise threatens to erode coastal habitat, including nesting habitats. In Hawaii, the majority of nesting occurs on FFS, a low-lying atoll vulnerable to increases in sea level (Baker et al. 2006). Neritic green turtles (juvenile, sub adult, and adults) are primarily herbivores that forage on seagrass and algae in shallow coastal areas. Contamination from runoff degrades seagrass beds, and introduced algae species may reduce native algae species preferred by green turtles (NMFS and USFWS 2007b).

Another threat to green turtles in Hawaii is fibropapillomatosis (FP), which causes debilitating tumors of the skin and internal organs. FP is the most significant cause of stranding and mortality in green turtles in Hawaii, accounting for 28 percent of standings', and has an 88 percent mortality rate in stranded afflicted turtles (Chaloupka et al. 2007). While the disease appears to have regressed over time (Chaloupka et al. 2009) it persists in the population at varying spatial scales (Van Houtan et al. 2010). Van Houtan et al. (2010) suggest a potential relationship exists between the expression of FP and the State's land use, waste-water management practices and invasive macroalgae.

Green turtles are also susceptible, in nearshore artisanal and recreational fisheries gear (Nitta and Henderson 1993; Chaloupka et al. 2007). These fisheries use a vast diversity of gears, including drift gillnets, long-lining, set-nets, pound-nets, trawls, and others, and are typically the least regulated of all fisheries while operating in the areas with greatest density of coastal foraging green turtles (NMFS and USFWS 2007b). Industrial fisheries also interact with green turtles, especially juveniles, like in the Hawaii-based deep-set and American Samoa longline fisheries.

The Hawaii shallow-set fishery rarely interacts with green turtles and since 2004, only seven have been incidentally caught. All seven were released alive. The Hawaii-deep set fishery occasionally interacts with green turtles and from 2005- 2014 there were seven observed, which is estimated to be 27 total green interactions in the deep-set fishery and from this the estimated mortality is 25 (rounded from 24.87) (McCracken 2006, 2007, 2008, 2009a, 2009b, 2010, 2011, 2012, 2013, 2014a; NMFS 2014b). The deep-set fishery currently has an incidental take statement for up to 21 anticipated green interactions and 18 anticipated mortalities over a three year period (NMFS 2005a), but has had fewer takes than previously anticipated in the 2005 BiOp. The California Oregon drift gillnet fishery has an incidental take statement for up to two anticipated green interactions and one anticipated estimated mortality every five years. Since 2001 no green turtles have been captured in the California Oregon drift gillnet fishery and only one has been observed since 1990 (NMFS 2013a).

Harvest of green turtles for their meat, shells, and eggs has been a major factor in past declines of green turtles, and continues to be a major threat globally (Humber et al. 2014). Despite increasing levels of protection, the direct take of turtles has continued legally in many regions and countries and is now characterized by culturally significant use by traditional coastal populations, or small-scale fisheries supplying local markets with meat and sometimes shell. Humber et al. (2014) found that currently 42 countries still permit the direct take of turtles and collectively take in excess of 40,000 turtles per year of which the majority (>80 percent) are green turtles. This legal take is focused in the wider Caribbean and Pacific Islands. Ten countries account for >90 percent of legal take, with the highest consumers being Papua New Guinea, Australia and Nicaragua. Although, within the 42 countries included in this study, there has been a significant decrease in take since the 1980s. On the Pacific coast of Mexico in the mid-1970s, >70,000 green turtle eggs were harvested every night. Globally, harvest of adults and eggs is reduced from previous levels, but still exists in some parts of the species' range. In Mexico, illegal adult harvest continues but at lower rates today than in the past (Gardner and Nichols 2001, Koch et al. 2006, Senko et al. 2011).The curio trade in Southeast Asia also harvests a large but unknown number of green turtles annually (NMFS and USFWS 2007b, Lam et al. 2012). Evidence from current seizure records and market surveys highlight a consistent illegal trade route to mainland China from the Coral Triangle region of South-east Asia (mainly the Philippines, Malaysia, and Indonesia). TRAFFIC reported 128 seizures involving the East Asian countries between 2000 and 2008, with a trade volume of over 9,180 marine turtle [primarily green and hawksbill turtle] products including whole specimens (2,062 turtles), crafted products (n = 6,161 pieces) and raw shell (Lam et al. 2012).

Green turtles forage in shallow areas, surface to breath, and often occur just below the surface. The majority of turtles in coastal areas spend their time at depths less than 5 m below the surface (Schofield et al. 2007, Hazel et al. 2009), and hence are vulnerable to being struck by vessels. A study completed in Australia found the proportion of green turtles that fled to avoid an approaching vessel increased significantly as vessel speed decreased (Hazel et al. 2007). Sixty percent of observed turtles encountered during low speed trials (2.2 knots) fled the approaching vessel. Flight response dropped to 22 percent and 4 percent at moderate (5.9 knots) and fast (10.3 knots) vessel speeds, respectively. Those that fled at higher vessel speeds did so at significantly shorter distances. The results implied that sea turtles cannot be expected to actively avoid a vessel traveling faster than 2.2 knots. The authors suggested that visual rather than auditory cues were more likely to provoke a flight response and that vessels transiting at slower speeds can assure a “turtle-safe” transit so both turtles and vessels have time to evade collisions (Hazel et al. 2007).

Although green turtles are probably already beginning to be affected by impacts associated with anthropogenic climate change in several ways no significant climate change-related impacts to green turtle populations have been observed to date. However, impacts from climate change are likely to influence biological trajectories in the future over the long-term, on a century scale (Paremsan and Yohe 2003). For example, increasing temperatures at nesting beaches may impact sex ratios of hatchlings (many rookeries already exhibit strong female bias (Binckley et al. 1998, Chan and Liew 1995, Godfrey et al. 1996, Godfrey et al. 1999, Godley et al. 2001, Kaska et al. 2006, Marcovaldi et al. 1997, Oz et al. 2004) and/or increase embryonic mortality (Matsuzawa et al. 2002). Increased nest mortality has also been linked to erosion due to increased typhoon frequency (Van Houtan and Bass 2007) and intensity, a predicted consequence of climate change (Webster et al. 2005). Seagrasses are a major food source for green turtles worldwide and habitat may suffer from decreased productivity and/or increased stress due to sea level rise and salinity and temperature changes (Short and Neckles 1999, Duarte 2002). Climate change induced shifts in ocean productivity linked to temperature changes (Harwood 2001; Edwards and Richardson 2004; Hays et al. 2005) may affect foraging strategies and therefore reproductive capacity for green turtles (Solow et al. 2002) similar to what has been observed during El Nino events in the western Pacific (Limpus and Nicholls 1994, Chaloupka 2001). While there are some available data on past trends, these data are limited, 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.

Marine debris is also a source of concern for greens due to the same reasons described for loggerheads. Green sea turtles can ingest small debris and larger debris can entangle animals leading to death. For greens the greatest risk is most likely when the debris is closer to the Northwest and main Hawaiian Islands where there are large numbers of turtles.

5.2.7.4 Conservation of the Species

Green turtles nesting in the U.S. have benefited from both state and federal laws passed in the early 1970s banning the harvest of turtles and their eggs. Protection and management activities since 1974 throughout the Hawaiian Archipelago and habitat protection at the FFS rookery since the 1950’s have resulted in increased population trends of both nesting and foraging turtles (Balazs and Chaloupka 2004). Elsewhere, the protection of nesting beaches from large-scale egg harvest appears to have reversed downward nesting trends in some cases. For example, nesting beach protection began at Colola, Mexico in 1979, and the number of nesting green turtles began to increase 17 years later in 1996 after reaching a low point in the late 1980s through the mid-1990s. Grupo Tortuguero (GT), a grassroots community-based network, is active in fifty coastal communities of Baja California, Mexico and mainland Mexico. Over the past 20 years, the GT (comprised of hundreds of local volunteers, many of whom are former poachers), has worked to protect and promote an appreciation for and pride in sea turtles. As a result of GT’s efforts to provide education and raise awareness to promote conservation activities, eastern Pacific green turtles are on the road to recovery (Delgado-Trejo and Alvarado-Díaz, 2012). Furthermore, encouraging trends in green turtle nester or nest abundance over the past 25 years has become apparent in at least six locations including Hawaii, Australia, Japan, Costa Rica and Florida (Chaloupka et al. 2007). Efforts to reduce fisheries bycatch of loggerheads, leatherbacks, and olive ridleys also benefit green turtles, such as improvements made in the Hawaii-based longline fisheries since 2004 (NMFS and USFWS 2007b).

Conservation and recovery of green 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. Within the WCPFC, NMFS has worked to modify and improve international bycatch mitigation requirements and aided in establishing a binding Sea Turtle Conservation Measure implementing the FAO Guidelines which has likely helped to reduce interactions and improve survivorship in international longline fisheries. As a result of these designations and agreements, many intentional impacts on sea turtles have been reduced: harvest of eggs and adults have been reduced at several nesting areas through nesting beach conservation efforts and an increasing number of community-based initiatives are in place to reduce the take of turtles in foraging areas (Gilman et al. 2007b, NMFS and USFWS 2007b).

5.2.8 Scalloped Hammerhead Indo-West Pacific DPS

Information in this section is summarized primarily from the 2014 Status Review Report (Miller et al 2014), the 2014 Final Rule (NMFS 2014d) and other sources cited below.

5.2.8.1 Population Characteristics

The scalloped hammerhead shark can be found in coastal warm temperate and tropical seas worldwide. The Indo-west Pacific DPS can be found throughout the entire Indian Ocean and in the western Pacific from Japan and China to New Caledonia, including throughout the Philippines, Indonesia, and off Australia. The scalloped hammerhead shark occurs over continental and insular shelves, as well as adjacent deep waters, but is seldom found in waters cooler than 22° C (Compagno 1984). It ranges from the intertidal and surface to depths of up to 450–512 m (Klimley 1993), with occasional dives to even deeper waters (Jorgensen et al. 2009). It has also been documented entering enclosed bays and estuaries (Compagno 1984). These sharks have been observed making migrations along continental margins as well as between oceanic islands in tropical waters (Kohler and Turner 2001, Duncan and Holland 2006, Bessudo et al. 2011, Diemer et al. 2011). Tagging studies reveal the tendency for scalloped hammerhead sharks to aggregate around and travel to and from core areas or “hot spots” within locations (Holland et al. 1993, Duncan and Holland 2006, Hearn et al. 2010, Bessudo et al. 2011), however they are also capable of traveling long distances (1941 km, Bessudo et al. 2011; 1671 km, Kohler and Turner 2001, Hearn et al. 2010; 629 km, Diemer et al. 2011). These long distance migrations have occurred over continental shelves and seamounts, and have not been seen over deep pelagic waters (NMFS 2014d).

The scalloped hammerhead shark is a high trophic level predator (trophic level = 4.1; Cortés 1999) and opportunistic feeder with a diet that includes a wide variety of teleosts, cephalopods, crustaceans, and rays (Compagno 1984, Bush 2003, Júnior et al. 2009, Noriega et al. 2011). In a study on feeding behavior in Kāne'ohe Bay, Bush (2003) found a nocturnal increase in the rate of foraging by juvenile scalloped hammerheads, with sharks consuming a mixture of crustaceans and teleosts. Stomachs of 466 scalloped hammerheads off the coast of Australia revealed the importance of bony fish as a prey item, followed by elasmobranchs, octopus and squid, and baitfish, with a positive correlation between shark length and the proportion of elasmobranchs in stomach contents (Noriega et al. 2011). The scalloped hammerhead shark is viviparous (i.e., give birth to live young), with a gestation period of 9-12 months (Branstetter 1987, Stevens and Lyle 1989), which may be followed by a one-year resting period (Liu and Chen 1999). Females attain maturity around 200-250 cm total length (TL) while males reach maturity at smaller sizes (range 128 – 200 cm (TL); Table 1); however, the age at maturity differs by region. Based on analysis of the available data, the scalloped hammerhead shark can be characterized as a long lived (at least 20 – 30 years), late maturing, and relatively slow growing species (based on Branstetter (1990), where $k < 0.1$/year indicates slow growth for sharks).

Current and Effective Population Size

Current effective population sizes are available for the scalloped hammerhead shark, but are considered qualitative indicators rather than precise estimates given their reliance on mutation rates and generation times (Duncan et al. 2006). Using two generation times (5.7 and 16.7 years), Duncan et al. (2006) calculated the effective female population (N_f) size of *S. lewini* for the major ocean basins. Based on a 1:1 sex-ratio (Clarke 1971, Chen et al. 1988, Stevens and Lyle 1989, Ulrich et al. 2007, White et al. 2008, Noriega et al. 2011), these calculations have been converted into total (both females and males) effective population size (N_e) by using the formula $N_e = 2(N_f)$. Results of N_e greatly varied within and between ocean basins, with the global N_e estimated at 280,000 using a generation time of 5.7 years, and 94,000 using a generation time of 16.7 years (NMFS 2014d).

Table 6. Estimates of current effective population size (Ne) of scalloped hammerhead sharks. (NMFS 2014 d, Adapted from estimates in Duncan et al. 2006)

Ocean Basin	**Population**	**Sample Size (n)**	**Ne (5.7 year generation time)**	**Ne (16.7 year generation time)**
Pacific	Baja	44	22,000,000	7,600,000
	Pac. Panama	8	62,000,000	2,000,000
	Hawaii	44	3,200	1,100
	Philippines	15	64,000	22,000
	Taiwan	20	15,600,000	5,200,000
	E. Australia	32	70,000	24,000
Indian	W. Australia	26	6,800	22,000
	Seychelles	12	16,200	54,004
	S. Africa	25	18,000	60,010
Atlantic	W. Africa	6	300,000	100,000
	East Coast USA	16	36,000,000	12,000,000
All	Total	271	280,000	94,000

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

5.2.8.2 Life History Characteristics Affecting Vulnerability to Proposed Action

Scalloped hammerheads occur over continental and insular shelves, as well as adjacent deep waters. Increased rates of foraging have been documented at night in studies on juveniles. They also have a varied diet, which consists of crustaceans, teleosts, elasmobranchs, octopus, squid, and baitfish (Miller at al 2014). Deep-set longline fishing occurs in their known range and the majority occurs at night when the hammerheads are foraging in pelagic waters. This overlap with longline fishing and their broad diet make it possible for hammerheads to become hooked or entangled in gear while foraging.

5.2.8.3 Threats to the Species

Overutilization by industrial/commercial fisheries, artisanal fisheries, and illegal fishing of the scalloped hammerhead shark are the most serious threats to the persistence of this DPS. Scalloped hammerhead sharks are both targeted and taken as bycatch in many global fisheries. They are targeted by semi-industrial, artisanal and recreational fisheries and caught as bycatch in pelagic longline fisheries, and purse seine fisheries. There is a lack of information on the fisheries prior to the early 1970s, with only occasional mentions in historical records. Significant catches of scalloped hammerheads have and continue to go unrecorded in many countries outside the U.S. In addition, scalloped hammerheads are likely under-reported in catch records as many records do not account for discards (example: where the fins are kept but the carcass is discarded) or reflect dressed weights instead of live weights. Also, many catch records do not differentiate between the hammerhead species, or shark species in general, and thus species-specific population trends for scalloped hammerheads are not readily available.

Total catches of the hammerhead family have increased since the early 1990s (prior years were not reported), from 377 tonnes in 1991 to a current peak of 5,786 mt in 2010. This is in contrast to the catches of scalloped hammerhead sharks, which have decreased, for the most part, since reaching a maximum of 798 tonnes in 2002. According to shark fin traders, hammerheads are one of the sources for the best quality fin needles for consumption, and fetch a high commercial value in the Asian shark fin trade (Abercrombie et al. 2005). In Hong Kong, the world’s largest fin trade market, scalloped hammerhead, and smooth hammerhead sharks are found under the “Chun chi” market category, the second most traded fin category in the market (Clarke et al. 2006a). Applying a Bayesian statistical method to the Hong Kong shark fin trade data, Clarke et al. (2006b) estimated that between 1 and 3 million hammerhead sharks, with an equivalent biomass of 60 – 70 thousand mt, are traded per year.

In the Pacific, there is a historical lack of shark reporting on logsheets for most fleets. In addition, if shark catch is reported, it is usually aggregated shark data. For example, in the Taiwanese large-scale and small-scale tuna longline fisheries, bycatch data were not reported until 1981 due to the low economic value of the bycatch in relation to the tunas (Liu et al. 2009). All shark data collected before 2003 was recorded in the logbooks under the category “sharks”. After 2003, species-specific information was recorded for the blue shark, mako shark, and silky shark, but all other sharks remained lumped in the category “other sharks” (Liu et al. 2009). Due to these data gaps, WCPFC recently revised their scientific data reporting requirements.

Beginning in 2011, WCPFC vessels are required to report species-specific catch information for the following shark species: blue, silky, oceanic white-tip, mako, thresher, porbeagle, and hammerheads (WCPFC 2011). Despite this requirement, recent catches of hammerheads have not been provided to the WCPFC for a number of longline fleets, including fleets from among the top twenty countries reporting Pacific shark catches to the FAO. The WCPFC also manages the active tuna purse seine fleet in this region, which has expanded significantly since the 1980s and experienced a sharp increase over the past 6 years. In the mid-1980s, the purse seine fishery accounted for only 40 percent of the total tuna catch, but in 2010, this percentage had increased to 75 percent (Williams and Terawasi 2011). The majority of the purse seine catch has historically been attributed to Japan, Korea, Chinese-Taipei and the USA fleets, however recently an increased number of Pacific Islands fleets as well as new fleets (from China, Ecuador, El Salvador, New Zealand, and Spain) have entered the WCPFC tropical fishery (Williams and Terawasi 2011). These new additions have brought the number of purse seine vessels up to 280, the highest it has been since 1972 (Williams and Terawasi 2011). However, WCPFC observer data, collected from 1994-2009, indicate that longline sets may pose more of a threat to non-target shark species than purse-seine sets in this convention area, but in terms of hammerhead sharks, observers reported only negligible catch but with high rates of finning in both types of sets (SPC 2010). In 2012, Bromhead et al. (2012) published a study that analyzed operational-level logsheet and observer data reported by fleets operating in the Republic of the Marshall Islands EEZ from 2005-2009. Although estimates of total annual longline catches of sharks ranged from 1,583 to 2,274 mt per year, only five scalloped hammerhead individuals were observed caught and subsequently discarded and finned during the study period (Bromhead et al. 2012).

Although range-wide abundance trends are missing in this DPS, CPUE data from South Africa and Australia suggest significant depletions of local populations. Declines of 58- 76 percent in the hammerhead population have been estimated for Australia’s northwest marine region, and a recent decline of 63 percent in *S. lewini* abundance was estimated for 2005-2010 based on data from a Queensland shark control program. Similarly, in South Africa, catch rates of *S. lewini* in beach mesh programs revealed significant declines in CPUE from 1978-2003. However, these programs were also assessed to have at least a medium causative impact on these localized depletions. High levels of commercial fishing that target sharks and catch sharks as bycatch occurs in this DPS. For example, in the Republic of the Marshall Islands EEZ, the tuna fishery alone accounted for annual longline catches ranging from 1583 to 2274 tonnes of sharks (over the period of 2005-2009) (Bromhead et al. 2012). Furthermore, four of the top five exporters of shark fins to Hong Kong (Singapore, Taiwan, Indonesia, and the United Arab Emirates) are located in this DPS’s range. The limited regulatory mechanisms to protect this DPS contribute to the risk of extinction due to overutilization by these various fisheries. For example, Indonesia, which at the beginning of the 21st century was the world’s leading elasmobranch producer accounting for 13 percent of the world total, currently has very few fishery regulations and in effect has created an open access fishery (Tull 2009). The heavy and unregulated artisanal and industrial fishing by both Indonesian and foreign vessels has depleted many of the large fish stocks, including sharks, in Indonesian waters (Field et al. 2009, Tull 2009). As a result, many Indonesian fishermen have moved south to illegally fish in Australian waters (Field et al. 2009). The level of management controls in Indonesia is not expected to increase because of the impact it would have on the livelihood of the many artisanal fisherman that operate in this area (Tull

2009). Likewise, many of the island countries in the western Pacific do not currently have the resources to implement or enforce protective fishery management measures, as any available funds are needed for important national needs, like health and education programs (Bromhead et al. 2012). Inshore fishing pressure is also of concern, as the schooling behavior of this species makes it susceptible to being taken in mass quantities on nursery grounds. Heavy exploitation of immature sharks has been observed in this DPS off the coasts of Madagascar, Queensland, and Southeast Asia (McVean et al. 2006, Harry et al. 2011, CITES 2010). The extinction risk analysis (ERA) team concluded that the limited management measures, large takes of immature *S. lewini*, and heavy fishing (both legal and illegal) on shark populations contributes significantly to the risk of this DPS's extinction, and these threats are likely to continue into the foreseeable future.

The Hawaii-based deep-set longline fishery catches scalloped Hammerhead sharks as bycatch at very low levels and the majority are from the central Pacific DPS, which is not listed as threatened or endangered. From 1995-2006, 56 scalloped hammerheads were caught on 26,507 observed sets in the HI longline fisheries (Walsh et al. 2009). Approximately 16 of those were caught by the deep-set fishery in the range of the Indo-west Pacific DPS, the rest were all in the range of the central pacific DPS. Since 2004, there have been three observed scalloped hammerhead sharks caught in the deep-set fishery in the Indo-west pacific DPS (Table 17) (PIRO Observer Program unpublished data). The American Samoa longline fishery has had an observer program since 2006, with coverage ranging between 6 percent and 8 percent from 2006-2009, and between 20 percent and 33 percent since 2010. Only eight scalloped hammerhead sharks have been observed caught in the fishery since the observer program started. The California fisheries operate within the range of the Eastern Pacific DPS and are therefore not expected to impact the Indo-west Pacific DPS of scalloped hammerhead sharks.

The threat of climate change was considered a low threat to scalloped hammerheads and an integrated risk assessment done by Chin et al 2010 provided evidence that they have low vulnerability to the threats from climate change (NMFS 2014d). The assessment took into account the in situ changes and effects that are predicted over the next 100 years in the Great Barrier Reef and assessed several species exposure, sensitivity, and adaptive capacity to a number of climate change factors including: water and air temperature, ocean acidification, freshwater input, ocean circulation, sea level rise, severe weather, light, and ultraviolet radiation (NMFS 2014d). The scalloped hammerhead had low vulnerability to each of the assessed climate change factors.

5.2.8.4 Conservation of the Species

The Hawaii longline fisheries are managed through the Pelagics FEP developed by the Western Pacific Fishery Management Council and approved by NMFS under the authority of the Magnuson-Stevens Act. Mandatory fishery observers have been monitoring both limited-entry fisheries (shallow and deep-set) since 1994, with observer coverage of at least 20 percent of trips since 2001 to provide a more comprehensive bycatch dataset. The Shark Finning Prohibition Act stopped shark finning in 2000 for the Hawaii-based longline fisheries, and a State of Hawaii ban on the possession of shark fins was imposed in 2010 (State of Hawaii SB2169).

At the international level NMFS identified the increasing number of shark fin bans as one potential effort to conserve the DPS, especially in Asian countries where the demand for shark fin soup is the highest. In addition scalloped hammerheads were just voted on to be added to Appendix II of the CITES, which means increased protection, but still allows legal and sustainable trade (NMFS 2014d). A number of Pacific Island countries (including U.S. territories) have also created shark sanctuaries, prohibited shark fishing, or have strong management measures to control exploitation of sharks in their respective waters, including Tokelau, Palau, Marshall Islands, American Samoa, CNMI, Cook Islands, and French Polynesia (NMFS 2014d).

6 Environmental Baseline

The environmental baseline for a biological opinion includes past and present impacts of all state, federal or private actions and other human activities in the action area, anticipated impacts of all proposed federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of state or private actions which are contemporaneous with the consultation in process (50 CFR 402.02). The Consultation Handbook further clarifies that the environmental baseline is “an analysis of the effects of past and ongoing human and natural factors leading to the current status of the species, its habitat (including designated critical habitat), and ecosystem, within the action area.” (USFWS and NMFS 1998). The purpose of describing the environmental baseline in this manner in a biological opinion is to provide context for effects of the proposed action on listed species.

The past and present impacts of human and natural factors leading to the status of the eight species addressed by this opinion within the action area include fishery interactions, vessel strikes, climate change, pollution, marine debris, and entanglement. The environmental baselines for the seven ESA-listed marine species addressed by this BiOp are described below.

6.1 Humpback Whales

Information in this section is summarized from the humpback whale Stock Assessment Reports (e.g., Allen and Angliss 2013), the humpback whale recovery plan (NMFS 1991), the SPLASH report (Calambokidis et al. 2008), the MMPA Section 101 (a)(5)(E)-Negligible Impact Determination (NMFS 2014e), the Global Review of Humpback Whales (Fleming and Jackson 2011), the 2012 BiOp (NMFS 2012a) and other sources cited below. The primary past and present impacts of human activities within the action area on the CNP humpback stock are fishery interactions and ship strikes.

The total of all known M&SI to the CNP stock of humpback whales as a result of fishing operations in Hawaii and Alaska for the time period from 2007 through 2011 is 46.75, resulting in a 5-year annual average take of 9.35 animals (Allen and Angliss 2013). This represents a summary of opportunistic reports of CNP humpback whale M&SI caused by entanglement from commercial and recreational fisheries and from onboard fisheries observers (Allen and Angliss 2013). The opportunistic reports come from a variety of ocean users who report the incident to the NMFS Alaska or Hawaii Regional Office Marine Mammal stranding center. In most cases these incidents cannot be attributed to a specific fishery because the interaction was not observed as it occurred but at some point later in time when the animal is spotted with gear attached. Therefore, these M&SI for CNP humpbacks represent minimum counts because they cannot be assigned to any one fishery and extrapolated out for total effort as is done for observed interactions onboard commercial fisheries. During this time period, one humpback was observed in the shallow-set longline fishery in Hawaii, which has 100 percent observer coverage; therefore, the one interaction represents the total number of interactions for that fishery. There were no interactions with CNP humpback whales observed in the deep-set longline fishery during this time period. The total number of mortalities attributed to ship strikes for the time period from 2007 through 2011 for CNP humpback whales is 21.24 with an annual average M&SI level of 4.25 animals/year. The total number of mortalities attributed to marine debris for the time period from 2007 through 2011 for CNP humpback whales is 11.25 with an annual average M&SI level of 2.25 animals/year (NMFS 2014e). An estimated annual total human-caused M&SI for the entire CNP stock of humpback whales for the 2007-2011 time-period is 16.20 animals (9.35 commercial fishery related + 0.35 recreational fishery related + 4.25 from vessel strikes + 2.25 from marine debris). Accordingly, total human-caused M&SI is well below the PBR (61.2) of this stock and is 26.47 percent of PBR (NMFS 2014e).

Because the CNP stock inhabit an area much larger than the action area, and fishing interactions with whales occur at a much lower rate in Hawaiian waters than in Alaskan waters (Allen and Angliss 2013), the combined impact of past and present fishing interactions and ship strikes within the action area is likely to be approximately seven whales per year (NMFS 2014e). In addition, impacts from anthropogenic sound such as military sonar and shipping (Ellison et al. 2011) within the action area are possible. Floating marine debris in the action area may present an entanglement hazard for humpbacks but is not likely to result in mortality. Whale- watching may affect humpbacks via vessel strikes and behavior disruption. The historic impact of whaling on this species is at most a minor part of the current environmental baseline, because; (1) the population has recovered from whaling, in terms of number of individuals, and (2) whaling was around the northern Pacific rim, thus little if any whaling occurred within the action area (NMFS 1991, Gilman et al. 2006, Calambokidis et al. 2008).

6.2 Sperm Whales

Information in this section is summarized from the sperm whale Stock Assessment Reports (e.g., Carretta et al. 2014), the sperm whale recovery plan (NMFS 2010c), the MMPA Section 101 (a)(5)(E)- Negligible Impact Determination (NMFS 2014e), and other sources cited below. The primary past and present impacts of human activities within the action area on the Hawaii sperm whale stock are fishery interactions and vessel strikes.

The estimated total of all known M&SI to the Hawaii stock of sperm whales as a result of commercial fishing operations for the time period from 2007 through 2011 is three (estimated based on one observed interaction in the deep-set longline fishery), resulting in an annual average of 0.7 M&SI (Table 9). There were no interactions with Hawaii sperm whales in the shallow-set longline fishery during this time period. The current PBR for this stock is 10.2 animals. Therefore, the total annual average incidental M&SI in commercial fisheries for Hawaii sperm whales for this timeframe is 6.86 percent of the PBR. There are no reported ship strikes for the Hawaii stock of sperm whales during this time period or for any other time frame. An estimated annual total human-caused M&SI rate from fisheries for the Hawaii stock of sperm whales for the 2007-2011 time period is 0.7. There was no other human caused mortality reported during this time frame. Accordingly, total human-caused M&SI is well below the PBR (10.2) of this stock and is 6.86 percent of PBR. The combined impact of past and present fishing interactions and ship strikes within the action area based on the analysis described above is likely to be approximately one sperm whale mortality from the Hawaii stock occurring annually in the action area.

In addition, impacts from anthropogenic sound such as military sonar and shipping (Ellison et al. 2011) within the action area are possible. Floating marine debris in the action area may present an entanglement hazard for sperm whales but is not likely to result in mortality.

6.3 Main Hawaiian Islands Insular False Killer Whales

Information in this section is summarized from the status review of Hawaiian Insular False Killer Whales under the ESA (Oleson et al. 2010), the final listing rule (NMFS 2012c), the final take reduction plan (NMFS 2012d), the Hawaiian Islands False killer whale Stock Assessment Reports (Carretta et al 2014), and other sources cited below. The primary past and present impacts of human activities within the action area on the MHI IFKW DPS are fishery interactions.

> The 5-year average of all known M&SI to the MHI insular stock incidental to commercial fishing operations for 2007-2011 is 0.1 (estimated based on one observed interaction of an unidentified blackfish in the insular/pelagic overlap area) whales per year. This single interaction was observed from the Hawaii deep-set longline fishery. There were no other sources of M&SI reported during this time period. However, fishery interactions are potentially an underestimate due to the lack of historical reporting and no observers in other fisheries such as the troll, handline, shortline, and kaka line fisheries. There are no M&SI of MHI insular false killer whales attributed to ship strikes during this time period. The estimated annual total human-caused M&SI rate for the MHI insular stock of false killer whales for the 2007-2011 time-period is 0.1.

Like other marine mammal species addressed by this opinion, past and present fisheries interactions have been, and continue to be, a threat to MHI IFKW DPS within the action area. Currently, the major type of fishing activity in the action area is longline fishing, except for nearshore fisheries that operate within the longline prohibited areas around the Hawaiian Islands, and only overlap with the transit areas of the deep-set fleet. It is unknown how these fisheries affect MHI IFKW as discussed in the threats to the species section.

In addition, impacts from anthropogenic sound such as military sonar and shipping (Oleson et. al 2010) within the action area are possible. Floating marine debris in the action area may present an entanglement hazard for MHI IFKW.

6.4 Loggerhead Turtles

Information in this section is summarized from the 2004 BiOp (NMFS 2004a), the 2005 BiOp (NMFS 2005a), the 2006 pelagics report (WPFMC 2006), the 2008 BiOp (NMFS 2008a), the 2009 Status Review, the 2011 Loggerhead DPS listing, the 2012 BiOp (NMFS 2012a), and the other sources cited below. Past and present fisheries interactions have been, and continue to be, a threat to loggerhead turtles within the action area. Currently, 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., MHI offshore handline mixed gear), but on such a small scale and with assumed low mortality rates as to be insignificant with regard to the loggerhead environmental baseline. Within longline prohibited areas around the Hawaiian Islands, numerous fisheries operate, but these do not affect loggerheads. Therefore, fisheries impacts on loggerheads in the action area are limited to longline fishing, past and present impacts of which are described below.

6.4.1 Longline Fishing

The action area lies entirely within the central North Pacific. Longline fishing is done by many countries in this region, and 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 3,500 and 6,000 for the last 30 years, this includes the 100-125 vessels in the Hawaii longline fisheries (the majority of which are involved in the deep-set fishery). The four main target species are yellowfin, bigeye, albacore tuna, and swordfish. The distribution of longline effort between 2000-2010 is shown in Figure 8 below. The action area is shown in the highlighted rectangle, and consists mostly of international waters.

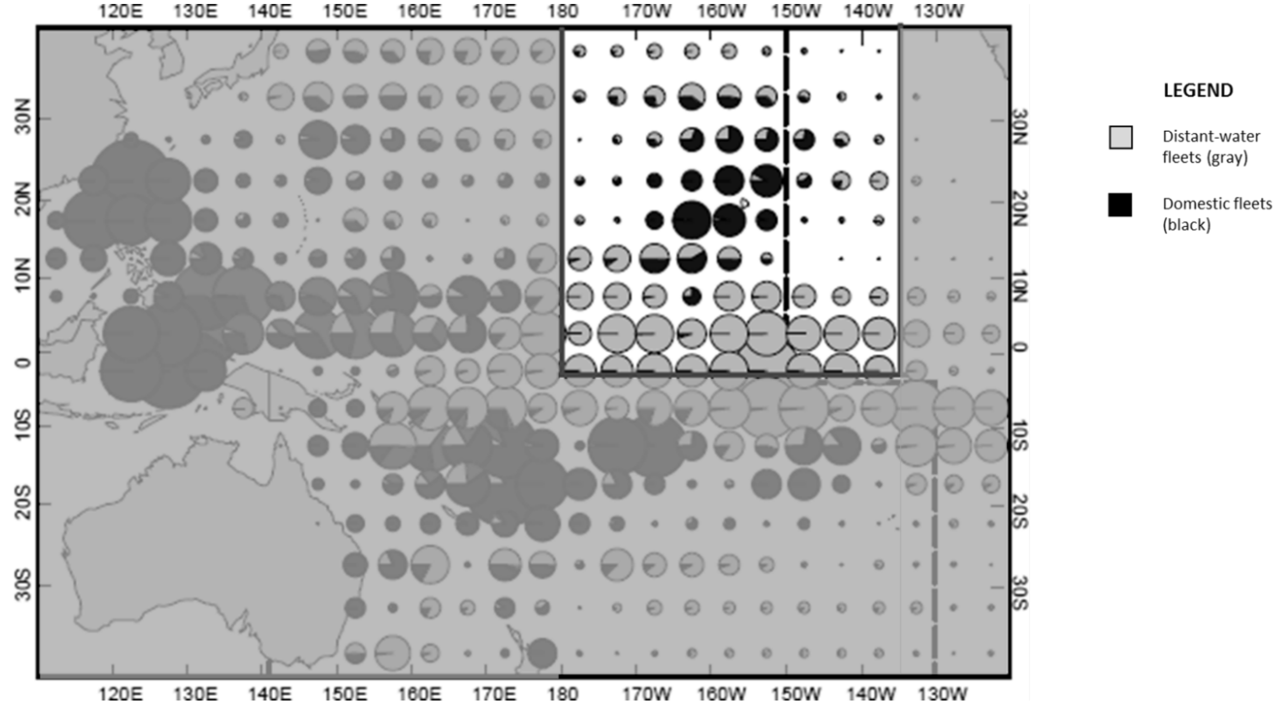

Figure 8. Distribution of longline effort of distant-water fleets (gray) and domestic fleets (black) for the period 2000-2010 (Williams and Terawasi 2011). Action area for Hawaii Deep-set longline fishery is highlighted (unshaded); remainder of map is shaded.

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. The deep-set longline fishery rarely catches loggerhead turtles. The shallow-set fishery operates further north and catches more loggerheads than the deep-set fishery. Other longline fisheries operating in the action area, such as the Taiwan and China tuna fisheries, have bycatch rates several times higher than the Hawaii-based deep-set fishery (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 and leatherback turtles in 2000. In the Pacific, they estimated 2,600 – 6,000 loggerhead juvenile and adult mortalities from pelagic longlining in 2000 (Lewison et al. 2004). However, using effort data from Lewison et al. (2004) and bycatch data from Molony (2005), Beverly and Chapman (2007) estimated loggerhead and leatherback longline bycatch to be approximately 20 percent of that estimated by Lewison et al. (2004), or 520 – 1,200 juvenile and adult loggerheads annually. Chan and Pan (2012) estimated that there were approximately 1866 total turtle interactions in 2009 in the central and North Pacific by comparing swordfish production and turtle bycatch rates from fleets fishing in the central and North Pacific area. From this we estimate that approximately 989[7] were loggerhead interactions and about 495 occurred in the central and North Pacific in 2009 (NMFS 2012a). A similar study has not been done comparing tuna production due to the more complex nature of the international tuna fisheries. We anticipate that the interactions with loggerheads and international fleets in the action area for the deep-set fishery would be lower than the shallow-set fishery because more of the effort occurs in warmer water.

For purposes of providing the environmental baseline for loggerheads in this opinion, NMFS estimates that longlining in the action area has killed 10 percent of the Pacific totals estimated by Beverly and Chapman (2007) and Lewison et al. (2004): 50 – 120 (10 percent of Beverly and Chapman’s 2007 estimate) to 260 – 600 (10 percent of Lewison et al.’s 2004 estimate), or 50 - 600 North Pacific juvenile and adult loggerheads annually.

The deep-set fishery has traditionally interacted with fewer loggerhead turtles than the shallow-set-set fishery, although mortality rates of turtles in shallow-set gear is lower than in deep-set gear. The reason for the higher mortality rates in the deep-set fishery is due to the gear being at greater depths, which does not allow the turtles to reach the surface to breath. From 2005-2014 there have been 4 observed Loggerhead interactions with an estimate of 24 total (NMFS 2014a). In 2013 there were two observed interactions with the deep-set fishery, which is unusual. Loggerheads are particularly susceptible to shallow-set gear and in the 1990s the Hawaii-based shallow-set fishery interacted with several hundred loggerheads annually in the action area. However, the shallow-set fishery was closed in 2001 and only re-opened in 2004 after instituting measures for reducing turtle interactions. This reformation of the Hawaii-based shallow-set fishery, including gear modifications and reduced effort, has resulted in an approximately 97 percent reduction in the average number of loggerhead interactions in this fishery since the 1990s (McCracken 2000, NMFS 2012a). From 2005-2014, turtle bycatch in Hawaii longline fisheries

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[7] Chan and Pan 2012 calculated that there were 1866 total turtle interactions in the Central and North Pacific in 2009. Approximately 53percent of turtle interactions in the Hawaii shallow-set fishery are with loggerheads; this percent was applied to the total number in order to estimate loggerhead interactions (1866 * 53percent=989).

(shallow-set and deep-set combined[8]) within the action area is estimated to have resulted in mean annual mortality of two loggerheads per year (NMFS 2014c).

6.4.2 Other Impacts

As mentioned in Section 5.2.4, and described in further detail below, climate change and marine debris may be affecting pelagic loggerhead habitat within the action area. Lower breeding capacity of North Pacific loggerheads in years following higher sea surface temperatures may reflect reduced ocean productivity during warmer years within the action area (Chaloupka et al. 2008a). In addition, marine debris may entangle or be ingested by turtles, leading to injury or possibly starvation, and derelict fishing gear may cause entanglement and possibly drowning. Data are not available to estimate the number of loggerhead mortalities resulting from climate change and marine debris in the past few years in the action area.

6.5 Leatherback Turtles

Information in this section is summarized from the 2004 BiOp (NMFS 2004a), the 2005 BiOp (NMFS 2005a), the 2006 pelagics report (WPFMC 2006), the 2008 BiOp (NMFS 2008a), the 2012 BiOp (NMFS 2012a), and other sources cited below. Like other sea turtle species addressed by this opinion, past and present fisheries interactions have been, and continue to be, a threat to leatherback turtles within the action area. Currently, the major type of fishing activity in the action area is longline fishing, except for nearshore fisheries that operate within the 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, hence instituting a global prohibition in 1992. Other types of fishing may occur in the action area outside of the longline prohibited areas (e.g., MHI offshore handline mixed gear), but on such a small scale and with assumed low mortality rates as to be insignificant with regard to the leatherback environmental baseline. Within the longline prohibited areas around the Hawaiian Islands, numerous fisheries operate, but these do not affect leatherbacks. Therefore, the fisheries impact on leatherbacks in the action area is limited to longline fishing, the past and present impacts of which are described below.

6.5.1 Longline Fishing

Longline fishing from domestic and international fleets within the action area is described in Section 6.4.1 and represented in Figure 8. Estimating the total number of sea turtle interactions in Pacific-wide longline fisheries is difficult because of low observer coverage and inconsistent reporting from international fleets. However, 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 and leatherback turtles in 2000. In the Pacific, they estimated 1,000 – 3,200 leatherback mortalities from pelagic longlining in 2000 (Lewison et al. 2004). An estimate of 626 adult female mortalities from pelagic longlining in 1998 was made by Kaplan (2005), or roughly 2,500 juveniles and adults. However, using effort data from Lewison et al. (2004) and bycatch data from Molony (2005), Beverly and Chapman (2007) estimated

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[8] The shallow-set fishery does not occur entirely in the action area. For purposes of this biological opinion the two fisheries are considered to have an overlap of approximately 25percent; therefore 25percent of the mortalities from the shallow-set fishery are added to the environmental baseline. The total number of mortalities resulting from the shallow-set fishery are discussed in the status of the species sections for all turtles.

loggerhead and leatherback longline bycatch to be approximately 20 percent of that estimated by Lewison et al. (2004), or 200 – 640 juvenile and adult leatherbacks annually. In a more recent study it was estimated that the number of interactions for all species operating in the central and North Pacific was approximately 1866 (Chan and Pan 2012). By comparing that to the shallow-set fishery action area it was estimated that approximately 375 leatherback interactions occurred in the same area in 2009 (NMFS 2012a).

For purposes of providing the environmental baseline for leatherbacks in this opinion, NMFS estimates that longlining in the action area has killed 10 percent of the Pacific totals estimated by Beverly and Chapman (2007), Kaplan (2005), and Lewison et al. (2004): 20 – 64 (10 percent of Beverly and Chapman's 2007 estimate) to 100 – 320 (10 percent of Lewison et al. 2004 estimate), or 20 - 320 western Pacific leatherback juveniles and adults annually.

Before 2001 in the North Pacific, the Hawaii-based longline fishery was estimated to capture about 110 leatherbacks annually (McCracken 2000). If we apply the old mortality rate of 32 percent (Gilman 2007), this would give us an estimated mortality of 36 (110*32 percent= 35.2) annually before the shallow-set portion of the fishery was closed in 2001. Changes to the fishery have resulted in an approximately 90 percent reduction in the average number of leatherback interactions annually in this fishery since the 1990s (McCracken 2000; NMFS 2012a). Since the shallow-set fishery re-opened in 2004, 17 (rounded up from 16.23) estimated leatherback mortalities occurred in the shallow-set fishery (NMFS 2014b). All of the leatherbacks caught were released alive; mortality estimates come from applying the NMFS post-hooking mortality criteria (Ryder et al. 2006) to interactions. The Hawaii-deep set fishery also interacts with leatherbacks and has an incidental take statement for up to 39 anticipated leatherback interactions and 18 anticipated mortalities over a three year period (NMFS 2005a). In the deep-set fishery from 2005-2014, there were 112 interactions and 45 (rounded up from 44.49) mortalities are estimated to have occurred (NMFS 2014a). From 2005-2014, turtle bycatch in Hawaii longline fisheries (shallow-set and deep-set combined[9]) within the action area is estimated to have resulted in mean annual mortality of six leatherbacks per year (NMFS 2014c). Other longline fisheries operating in the action area, such as Taiwan and China tuna fisheries, are thought to have bycatch rates several times higher than the Hawaii-based deep-set fishery (NMFS 2008a, Kaneko and Bartram 2008).

6.5.2 Other Impacts

As mentioned in Section 5, and described in further detail below, climate change and marine debris may be affecting pelagic leatherback habitat within the action area. Leatherbacks may be particularly susceptible to ingesting of marine debris because plastic bags resemble sea jellies, their primary prey. Derelict fishing gear may cause entanglement and drowning. Data are not available to estimate the number of leatherback mortalities resulting from climate change and marine debris in the past few years in the action area.

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[9] The shallow-set fishery does not occur entirely in the action area. For purposes of this biological opinion the two fisheries are considered to have an overlap of approximately 25percent; therefore 25percent of the mortalities from the shallow-set fishery are added to the environmental baseline. The total number of mortalities resulting from the shallow-set fishery are discussed in the status of the species sections for all turtles.

6.6 Olive Ridley Turtles

Information in this section is summarized from the 2004 BiOp (NMFS 2004a), the 2005 BiOp (NMFS 2005a), the 2006 BiOp (NMFS 2006), the 2008 BiOp (NMFS 2008a), the olive ridley 5-year status review (NMFS and USFWS 2014), the 2012 BiOp (NMFS 2012a), and other sources cited below. Past and present fisheries interactions have been, and continue to be, a threat to olive ridley turtles within the action area. Longline fishing as described above is the most important past and present impact on olive ridleys. Much less attention has been paid to the effects of longline fishing on this species than has been given to similar impacts on loggerheads and leatherbacks, hence no estimates are available for olive ridley mortality from longline fishing in the Pacific. Olive ridleys are susceptible to deep-set longlining because of their deep foraging behavior (loggerhead interactions are rare in deep-set fishing because of shallow foraging) (Polovina et al. 2003, 2004). In the Hawaii-based deep-set longline fishery, the bycatch rate of olive ridleys is higher than other species (McCracken 2006, 2007, 2008, 2009a, 2009b, 2010, 2011, 2012, 2013, 2014a). In addition, mortality of bycaught olive ridleys is higher than the other sea turtle species, most likely because they are hooked in deep water and unable to reach the surface to breath. Bycatch rates in foreign deep-set fisheries (for tuna) are >10 times higher than in the Hawaii-based deep-set fishery, and constitute much more fishing effort than the Hawaii-based fishery (Beverly and Chapman 2007). Thus it is likely that thousands of olive ridley mortalities occur annually in the Pacific via longlining.

Due to the abundance of this species and the amount of longlining occurring within the action area by all fleets combined, at least several hundred olive ridley mortalities are thought to have occurred annually, and are still occurring annually via longlining (most from the eastern Pacific population, but some from the western Pacific population).

From 2005-2014, turtle bycatch in Hawaii longline fisheries (shallow-set and deep-set combined[10]) within the action area is estimated to have resulted in mean annual mortality of 27 olive ridleys per year (NMFS 2014c).

As mentioned in Section 5, and described in further detail below, climate change may be affecting pelagic olive ridley habitat within the action area. Marine debris and derelict fishing gear may cause entanglement and possibly drowning. Data are not available to estimate the number of olive ridley mortalities resulting from climate change and marine debris in the past few years in the action area.

6.7 Green Turtles

Information in this section is summarized from the 2004 BiOp (NMFS 2004a), the 2005 BiOp (NMFS 2005a), the 2006 BiOp (NMFS 2006), the 2008 BiOp (NMFS 2008a), the green turtle 5-year status review (NMFS and USFWS 2007b), the 2012 BiOp (NMFS 2012a), and the other sources cited below. Past and present fisheries interactions have been, and continue to be a threat to green turtles within the action area. However, unlike loggerheads, leatherbacks, and olive

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[10] The shallow-set fishery does not occur entirely in the action area. For purposes of this biological opinion the two fisheries are considered to have an overlap of approximately 25percent; therefore 25percent of the mortalities from the shallow-set fishery are added to the environmental baseline. The total number of mortalities resulting from the shallow-set fishery are discussed in the status of the species sections for all turtles.

ridleys, green turtles are affected by both longline fishing and nearshore fishing, and other anthropogenic threats within the action area. As explained in Section 5, this is because juvenile green turtles in the Hawaiian population recruit to nearshore areas throughout the Hawaiian Archipelago, hence juveniles are affected by longline fishing while utilizing pelagic habitats, and by nearshore fishing during the adult nearshore life history stage. In addition, climate change and marine debris may be affecting this species in the action area.

Much less attention has been paid to the effects of longline fishing on green turtles than has been given to similar impacts on loggerheads and leatherbacks, thus no estimates are available for green turtle mortality due to longline fishing in the entire Pacific. While few green turtle interactions occur in Hawaii-based fisheries, general turtle bycatch rates in foreign deep-set fisheries (for tuna) are >10 times higher than in the Hawaii-based fisheries (Bartram and Kaneko 2004), and constitute much more fishing effort than the Hawaii-based fisheries. Therefore it is likely that within the action area, up to several hundred juvenile green turtle mortalities occur annually by longlining. From 2005-April 2014, turtle bycatch in Hawaii longline fisheries (shallow-set and deep-set combined[11]) within the action area is estimated to have resulted in mean annual mortality of 3 greens per year (NMFS 2014c).

Interactions in nearshore fisheries in the MHI (e.g. lay gillnets, hook-and-line, etc.) sometimes result in entanglement and drowning of green turtles. Of many kinds of nets used in Hawaii, gillnets are most problematic for turtles, because they are left untended, and entangled animals usually drown. Revised State of Hawaii regulations governing lay gillnets began in March 2007: they cannot be set after sunset and during daylight hours, can be legally left untended in ½ hour increments, must be inspected completely every two hours, and may not be used for more than four hours during any set. However the likelihood of turtle entanglement and drowning still persists even if all fishers comply with State regulations. Hook-and-line fishing from shore or boats also hook or entangles green turtles, although the chance of survival is higher than if caught in a gillnet (Chaloupka et al. 2008b). Turtles drowned in fishing gear do not typically ‘strand’ (come ashore to die, or wash up on shore dead), so there are no estimates for the total number of green turtle mortalities that occur annually from fishing interactions (NMFS 2008c). Between 1982 and 2013 the most common known cause of green turtle standings was the tumor-forming disease, fibropapillomatosis (27 percent) with an 88 percent mortality rate of stranded afflicted turtles, followed by hook-and-line fishing gear-induced trauma (11 percent) and gillnet fishing gear-induced trauma (5 percent) (PIFSC TRP Turtle Stranding Database 2014). Since 2002, there has been a steady increase in the rate of hook and line fishing induced standings’ from 20 turtles per year to over 40, ranging from 10 percent to 20 percent of reported standings’ (PIFSC TRP unpublished quarterly stranding report to PIRO).

Total annual green sea turtle mortalities from 1998-2007 in the main Hawaiian islands by boat collisions was estimated by NMFS (2008e) based on stranded turtle mortalities (Chaloupka et al. 2008b, PIFSC TRP Turtle Stranding Database 2007). An estimate of 10 stranded turtle

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[11] The shallow-set fishery does not occur entirely in the action area. For purposes of this biological opinion the two fisheries are considered to have an overlap of approximately 25percent; therefore 25percent of the mortalities from the shallow-set fishery are added to the environmental baseline. The total number of mortalities resulting from the shallow-set fishery are discussed in the status of the species sections for all turtles.

mortalities from boat collisions in the MHI (see Figure 3, p. 25, NMFS 2008c) was determined to represent 20-40 percent of all annual green turtle mortalities in the MHI by boat collisions, resulting in a range of 25-50 turtle mortalities per year. Thus the average number of green turtle mortalities per year by boat collisions was estimated at 37.5 (NMFS 2008c). Between 1982 and 2013, seventy-three records of turtle standings' exist from Ala Moana Regional Park to Kaka'ako Waterfront Park and Kewalo Basin, which is a busy area for boaters, the location where longline boats harbor and transit to and from port, and an area with a high abundance of green sea turtles. Of stranding cases, only two strandings involved a boat collision as the determined cause of stranding (PIFSC TRP Turtle Stranding Database 2014), which may mean that earlier estimates are exaggerated.

As mentioned in Section 5 and described in further detail below, climate change may be affecting pelagic green turtle habitat within the action area. Marine debris and derelict fishing gear may cause entanglement and possibly drowning. Data are not available to estimate the number of green turtle mortalities resulting from climate change and marine debris in the past few years in the action area.

6.8 Scalloped Hammerhead sharks

Information in this section is summarized primarily from the 2014 Status Review Report (Miller et al 2014), the 2014 Final Rule (NMFS 2014d) and other sources cited below. As described earlier the greatest threats to scalloped hammerhead sharks are overutilization from artisanal and commercial fisheries and illegal fishing. These threats are greater outside of the action area of the deep-set longline fishery. Only a small portion of the Indo-West Pacific DPS range falls within the action area (Figures 2,7), with the majority occurring in the range of the central Pacific DPS, which was not listed.

Within the action area the Hawaii-based deep-set longline fishery catches very low numbers of scalloped Hammerhead sharks as bycatch. Approximately 16 scalloped hammerheads were caught by the deep-set fishery in the range of the Indo-west Pacific DPS since 1995. Since 2004, there have been 3 observed scalloped hammerhead sharks caught in the deep-set fishery in the Indo-west pacific DPS, all of which were caught from 2004-2007 (PIRO Observer Program, unpublished data). Based on the three observed and the observer coverage levels in those years we estimate that the total catch of scalloped hammerheads from the Indo-Pacific DPS was approximately 14, which is approximately 2 annually (rounded from 1.4) during the 2004-2014 time period (Table 17). There are no observed reports of scalloped hammerhead sharks caught by the shallow-set fishery in the area of the Indo-West Pacific DPS. Therefore we estimate that there are no more than 2 scalloped hammerheads from the Indo-West Pacific DPS caught annually by the Hawaii deep-set longline fishery. As discussed earlier the historic level of reporting by most fisheries was minimal, and where it did exist it was not precise enough to determine species of hammerhead, or even shark in most cases. Therefore there is not much available information on the level of take of scalloped hammerheads by foreign fishers in the action area.

6.9 All species: impacts associated with climate change

Global mean temperature has risen 0.76°C over the last 150 years, and the linear trend over the last 50 years is nearly twice that for the last 100 years (Solomon et al. 2007). Climate change is a global phenomenon so resultant impacts have likely been occurring in the action area, although scientific data describing any impacts that have occurred from climate change in the action area are lacking. As discussed in the Threats Section, no significant climate change-related impacts to humpback whale, sperm whale populations, or the MHI Insular false killer whale DPS have been reported to date. As also discussed in the Threats Section, climate change is likely beginning to affect sea turtles found in the action area through the impacts of rising sand temperatures, rising sea level, increased typhoon frequency, and changes in ocean temperature and chemistry.

While sea turtle hatchling sex ratios vary naturally within and among seasons and nesting locations, several species already exhibit female bias throughout their major rookeries worldwide, in many cases producing anywhere from 60 – 99 percent females (Chan and Liew 1995, Godfrey et al. 1996, Marcovaldi et al. 1997, Binckley et al. 1998, Godfrey et al. 1999, Godley et al. 2001, Oz et al. 2004, Kaska et al. 2006). Monitoring data over a long enough timescale to discern climate change related trends in sea turtle sex ratio have not been collected in the action area. Sea level rose approximately 17 cm during the 20th century (Solomon et al. 2007) and further increases are expected. There are several predictions for potential future sea turtle nesting habitat loss due to sea level rise (Fish et al. 2005; Baker et al. 2006; Fuentes et al. 2009), however available data are insufficient to determine an existing correlation between past sea level rise and sea turtle population dynamics (Van Houtan 2010).

Global climate change-induced elevated temperatures, altered oceanic chemistry, and rising sea level may be contributing to changes to coral reef and seagrass ecosystems (as described above in Status of the Species) which provide resting and foraging habitat for some sea turtles, although it is difficult to distinguish impacts of climate-related stresses from other stresses that produce more prominent short term effects (Parry et al. 2007). Climate change-induced shifts in ocean productivity linked to temperature changes (Harwood 2001, Edwards and Richardson 2004, Hays et al. 2005) may affect foraging strategies and therefore reproductive capacity for sea turtles (Solow et al. 2002, Chaloupka et al. 2007, Van Houtan and Halley 2011, Van Houtan 2011), similar to what has been observed during El Nino events in the Pacific (Limpus and Nicholls 1994, Chaloupka 2001, Saba et al. 2007, Reina et al. 2008). These shifts in abundance of foraging resources are also directly linked to observed modifications in phenology for sea turtles such as longer re-migration intervals and temporal shifts in nesting activity (Weishampel et al. 2004, Hawkes et al. 2007). However, at this time it is only possible to speculate as to the implications of such impacts, as findings raise numerous follow up questions (listed by Weishampel et al. 2004) including whether earlier nesting will affect overall fecundity, clutch size, incubation length, hatch success, mating synchrony, and sex ratio. Recent studies have demonstrated that climate conditions influence juvenile recruitment and impact population trends in the North Pacific loggerhead DPS, Northwest Atlantic loggerhead DPS, western pacific leatherbacks, and Gulf of Mexico hawksbills (Van Houtan and Halley 2011, Van Houtan 2011, del Monte-Luna et al. 2012). Changes in reproductive capacity and temporal shifts of nesting activity associated with changing environmental conditions have not been studied specifically in the action area.

Additional potential effects of climate change on sea turtles include range expansion and changes in migration routes (Robinson et al. 2008). Leatherbacks have extended their range in the Atlantic north by 330 km in the last 17 years as warming has caused the northerly migration of the 15°C SST isotherm, the lower limit of thermal tolerance for leatherbacks (McMahon and Hays 2006). Similar studies on changes in migration routes for loggerheads, leatherbacks, olive ridleys, and greens have not been done in the Pacific. Therefore, it is not possible to say with any degree of certainty whether or how their migration routes and ranges have been or are currently affected.

The ranges of 88 percent of cetaceans may be affected by changes in water temperature resulting from global climate change, however the humpback whale and sperm whale are cosmopolitan species ranging throughout the world's oceans and thermal and prey limitations related to climate change are unlikely to impact the range of these species (MacLeod 2009). Whilst oceanic cetaceans are unlikely to be directly affected by rising in sea level, important habitats for coastal species and species that require coastal bays and lagoons for breeding, such as humpback whales, could be adversely affected in the future (Simmonds and Elliot 2009). Humpback whales that feed in polar regions may also encounter reduced prey.

Attempting to determine whether recent biological trends are causally related to anthropogenic climate change is complicated because non-climatic influences dominate local, short-term biological changes. However, the meta-analyses of 334 species and the global analyses of 1,570 species show highly significant, nonrandom patterns of change in accord with observed climate warming in the twentieth century. In other words, it appears that these trends are being influenced by climate change-related phenomena, rather than being explained by natural variability or other factors (Parmesan and Yohe 2003). The details discussed previously in this section support the probability that recently observed changes in sea turtle phenology, sex ratio, and foraging characteristics in studied populations may be influenced by climate change-related phenomena. However, the implications of these changes are not clear in terms of population level impacts, and data specific to the action area are lacking.

As discussed in the status of the species section climate change is not considered to be a major threat to scalloped hammerhead sharks.

In summary, several factors of climate change are impacting turtle populations or may impact populations in the future. Climate variability from year to year influences juvenile recruitment and influences nesting for several populations of turtles; turtles have encountered this type of climate variability throughout their entire existence but changes in climate variability due to anthropogenic climate change is a less understood issue. There are different life stages that will be affected by different aspects of climate change, some may be positive and others negative. Since it is anticipated that changes due to increasing temperatures are expected to occur slowly over the next century, species may adapt as they have done with a variable climate throughout their existence.

7 Effects of the Action

In this section of a biological opinion, NMFS assesses the probable effects of the proposed action on threatened and endangered species. 'Effects of the action' refers to the direct and indirect effects of an action on species or critical habitat, together with the effects of other activities that are interrelated or interdependent with that action that will be added to the environmental baseline. "Direct effects" are those affects that are caused directly by the action. "Indirect effects" are those that are reasonably certain to occur later in time (50 CFR 402.02). The 'Effects of the action' are considered within the context of the 'Status of Listed Species' and together with the 'Environmental Baseline' sections of this opinion, along with 'Cumulative Effects' to determine if the proposed action can be expected to have direct or indirect effects on threatened and endangered species that appreciably reduce their likelihood of surviving and recovering in the wild by reducing their reproduction, numbers, or distribution (50 CFR 402.02), otherwise known as the jeopardy determination.

Approach. NMFS determines the effects of the action using a sequence of steps. The first step identifies stressors (or benefits) associated with the proposed action with regard to listed species. The second step identifies the magnitude of stressors (e.g., how many individuals of a listed species will be exposed to the stressors; *exposure analysis*). In this step of our analysis, we try to identify the number, age (or life stage), and gender of the individuals that are likely to be exposed to a proposed action's effects, and the populations or subpopulations those individuals represent. The third step describes how the exposed individuals are likely to respond to these stressors (e.g., the mortality rate of exposed individuals; *response analysis*).

The final step in determining the effects of the action is establishing the risks those responses pose to listed resources (*risk analysis*). The risk analysis is different for listed species and designated critical habitat. Our jeopardy determinations must be based on an action's effects on the continued existence of threatened or endangered species as those species have been listed, which can include true biological species, subspecies, or distinct population segments of vertebrate species. Because the continued existence of listed species depends on the fate of populations that comprise them, viability (probability of extinction or probability of persistence) of listed species depends on viability of their populations. Similarly, the continued existence of populations are determined by the fate of individuals that comprise them; populations grow or decline as individuals that comprise the population live, die, grow, mature, migrate, and reproduce (or fail to do so).

Our risk analyses reflect these relationships between listed species and the populations that comprise them, and the individuals that comprise those populations. We begin by identifying the probable risks the action poses to listed individuals that are likely to be exposed to an action's direct and indirect effects. Our analyses then integrate those individuals risks to identify consequences to the populations those individuals represent. Our analyses conclude by determining the consequences of those population-level risks to the species those populations comprise.

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. In particular, we examine the scientific and commercial data available to determine if an individual's probable responses to an action's effects 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.

Reductions 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 is itself 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. 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 fitness reductions are likely to be sufficient to reduce the viability of the populations those individuals represent (measured using changes in the populations' abundance, reproduction, spatial structure and connectivity, growth rates, or variance in these measures to make inferences about the population's extinction risks). In this step of our analyses, we use the population's base condition (established in the 'Status of Listed Species', 'Environmental Baseline', and 'Cumulative Effects' sections of this 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.

This introduction summarizes stressors and interactions resulting from the proposed action. It is included here to set the stage for following sections.

Potential Stressors. Potential stressors associated with the proposed action are listed here, and then described in more detail for each species in the following sections. The proposed action is the continued operation of the Hawaii deep-set longline fishery. The greatest stressor associated with this action on the eight listed species considered in this opinion is interactions with fishing gear. Another potential stressor associated with the proposed action is collisions with fishing vessels. Vessels travel through areas with dense concentrations of some listed species, such as when vessels travel to and from port, passing through nearshore waters where green turtles occur. While additional effects may occur due to the proposed action (e.g., exposure to waste from fishing vessels), they are not considered likely to adversely affect individuals of listed species, and thus are not considered stressors. The potential direct stressors of interactions and collisions are described in detail below in the species sections, because they vary considerably between species.

Exposure

Even though the Hawaii longline fisheries have been in operation for many years the deep-set and shallow-set fishery have only been operating as two distinct fisheries since the shallow-set fishery re-opened in April 2004. Protected species interactions that have been observed in the deep-set fishery from 2004-April 2014 and the total estimated number of interactions based on the observed interactions are shown below in Table 7. Exposure for each species will be explained in greater detail below.

Table 7. Number of observed and estimated total interactions[12] between listed species and the Hawaii deep-set fishery from 2004- June 30, 2014. Observed interactions from PIROP annual and quarterly reports. Estimated total interactions from McCracken 2006, 2007, 2008, 2009a, 2009b, 2010, 2011, 2012, 2013, 2014a; NMFS 2013c.

Year	**Interactions**															
	Humpbacks		Sperm		Insular false killer whale[13]		Loggerheads		Leatherbacks		Olive Ridleys		Greens		Scalloped Hammerheads	
	Obs	Est	Obs	Est	Obs	Est	Obs	Est	Obs	Est	Obs	Est	Obs	Est	Obs	Est
2004	1	5.5	0	0	0	0	0	0	2	14	13	45	1	4	2	9
2005	0	0	0	0	1	0.7065	0	0	1	4	4	17	0	0	0	0
2006	0	0	0	0	1	2.7222	0	0	2	9	11	55	2	6	0	0
2007	0	0	0	0	0	0	1	6	2	4	7	26	0	0	1	5
2008	0	0	0	0	0	0	0	0	1	11	3	17	0	0	0	0
2009	0	0	0	0	0	0	0	0	1	4	3	18	0	0	0	0
2010	0	0	0	0	0	0	1	6	1	6	4	10	1	1	0	0
2011	0	0	1	6.3870	1	0.8920	0	0	3	14	7	36	1	5	0	0
2012	0	0	0	0	2	4.89	0	0	1	6	6	34	0	0	0	0
2013	0	0	0	0	0	0	2	10	3	15	9	44	1	5	0	0
2014	0	0	0	0	0	0	0	0	6	42	5	23	1	4	0	0
Total	**1**	**5.5**	**1**	**6.3870**	**3**	**9.21**	**4**	**22**	**23**	**129**	**72**	**325**	**7**	**25**	**3**	**14**

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[12] Interactions includes all interactions whether they are non-serious, serious, or mortalities.

[13] Number of observed and estimated is based on pro-ration of observed false killer whales and unidentified blackfish in the insular/ pelagic overlap area as defined in section 5.2.3.

7.1 Humpback Whales

The stressors, exposure, response, and risk steps of the effects analysis for humpback whales with regard to implementation of the proposed action are described below. The following information was used to conduct these analyses of the proposed action on humpback whales: the 2012 BiOp (NMFS 2012a), the humpback whale Stock Assessment Reports (e.g., Allen and Angliss 2013), the humpback whale recovery plan (NMFS 1991), the SPLASH report (Calambokidis et al. 2008), the Global Review of Humpback Whales (Fleming and Jackson 2011), and other sources cited below.

7.1.1 Stressors

The primary stressor of the Hawaii deep-set longline fishery on humpback whales is entanglement with fishing gear. Humpbacks are present in the action area as they migrate to and from waters surrounding the Hawaiian Islands. However, the deep-set longline fishery generally occurs at locations where humpbacks are uncommon. Thus, interactions between the Hawaii-based longline fishery and humpback whales are rare and unpredictable events. Vessel strikes are not considered a stressor since most of the fishing occurs where humpbacks are uncommon and vessels are only briefly transiting through the grounds where they are common. There have been six observed fishery interactions between humpbacks and the entire Hawaii-based longline fleet since observer coverage began, three of which were with the deep-set fishery (Table 8).

Table 8. Summary of observed interactions between humpback whales and the Hawaii-based longline fisheries from 1995-2013 (Forney 2010, Bradford & Forney 2013). Seriousness of injuries was assessed under MMPA serious injury guidelines for interactions that occurred between 1995-2006 (Angliss and DeMaster 1998). For interactions occurring after 2007 revised serious injury determination guidelines were used (NMFS 2012b).

Hawaii Longline Fishery	Date	EEZ	NMFS-Determined Severity
Deep Set	2/11/2001	Hawaii	Not serious
Deep Set	10/12/2002	Outside	Not serious
Deep Set	2/16/2004	Outside	Not serious
Shallow Set	2/19/2006	Outside	Serious
Shallow Set	12/29/2007	Outside	Not serious
Shallow Set	11/13/2011	Outside	L10 (Prorate 0.75 Serious)

According to descriptions of these interactions by NMFS fishery observers, the whales were entangled in the mainline. In each instance, efforts were taken to disentangle the whale, and all whales were either released or able to break free from the gear without noticeable impairment of the animal's ability to swim or feed. However, if entanglement results in the gear wrapping around the animal and breaking off, the animal may trail the gear for a long period of time. Available evidence from entangled humpback whales indicates that while it is not possible to predict whether an animal will free itself of gear, a large proportion of them are believed to extricate themselves based on scarring observed among apparently healthy animals. A study in southeast Alaska on the central North Pacific stock of humpback whales estimated that about 71 percent had been entangled at some point in their life and survived from a variety of gear types, but with the majority most likely coming from pot gear and gillnets (Neilson et al. 2009).

7.1.2 Exposure

Since 2004 (when the deep-set and shallow-set became two separately managed fisheries), there has been one observed interaction with a humpback whale which was estimated to be a total of 5.5 interactions[14] (Table 7). In order to predict the future level of interactions with humpback whales and such low numbers of historic interactions, NMFS used a conditional binomial model to predict future interaction levels (McCracken 2014b). The proposed action is defined as approximately 1,305 trips, and18,592 sets with 46,117,532 hooks annually, thus the number of humpback whales that are likely to be entangled as a result of interactions with longline gear associated with the proposed action is 2 (rounded from 1.3) annually[15] (McCracken 2014b).

7.1.3 Response

NMFS rates the severity of marine mammal interactions with fishing gear using serious injury guidelines developed for the MMPA. Seriousness of injuries from 1996-2006 were assessed under MMPA serious injury guidelines developed in 1998 (Angliss and DeMaster 1998). For interactions occurring after 2007 revised serious injury determination guidelines were used (NMFS 2012b). Of the six interactions of humpbacks with the Hawaii-based longline fisheries since 2001 (3 in deep-set, 3 in shallow-set), four were 'not serious', one was 'serious', and one was prorated to be 0.75 serious (Table 5) (Forney 2010). The future level of M&SI was determined using pooled interaction data from sperm whale, humpback, and Bryde's whale interactions since there were so few with each species individually. These three species are not known to depredate catch or bait and their observed interactions have been entanglements, pooling their injury conditions likely introduces little, if any bias; whereas, without pooling, the predictions of M&SI would be less accurate as a result of each species having few observed interactions (McCracken 2014b).

In the exposure analysis above, humpback exposure to the proposed action is estimated to result in less than two entanglements annually. It is estimated that the annual M&SI will be 0.7 based on the past M&SI determinations (McCracken 2014b). As described earlier (footnote number 5) all serious injury is considered mortality. Therefore, the proposed action is expected to kill one (rounded up from 0.7) humpback whale every year from the CNP stock.

7.1.4 Risk

The mortality and serious injury of CNP humpback whales due to the proposed action is estimated to be less than one whale every year. This estimated number would not approach the stock's PBR, which is 61.2 animals. PBR is a calculation that is used under the MMPA to evaluate impacts to stocks and is used in this analysis to help inform our ESA jeopardy analysis, but is not the sole basis for the decision. A stock's PBR is defined by the MMPA as the

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[14] Interactions includes all interactions whether they are non-serious, serious, or mortalities.

[15] A sensitivity analysis was done by McCracken to account for variability in the level of effort based on trips. The number analyzed was between 1,300 and 1,525 trips with the same level of sets and hooks per trip (McCracken 2014a). The level of hook effort observed in 2013 was approximately 1percent higher than used in the predictive model and this level of hook effort corresponds to 1473 trips as described in the analysis (McCraken pers comm.). The number of interactions at this number of trips falls within the range projected for humpback whales and below the 2 animals used in the BiOp analysis (Figure 20 of McCracken 2014a).

maximum number of animals, not including natural mortalities, that may be removed from a marine mammal stock while allowing that stock to reach or maintain its optimum sustainable population. A stock's OSP is further defined by the MMPA as the number of animals which will result in the maximum productivity of the population or the species, keeping in mind the carrying capacity of the habitat and the health of the ecosystem of which they form a constituent element. (16 U.S.C. 1362(3)(9)). 50 CFR 216.3 explains that the optimum sustainable population is a population size which falls within a range from the population level of a given marine mammal species or stock which is the largest supportable within the ecosystem to the population level that results in maximum net productivity. Accordingly, a marine mammal stock's OSP may be significantly different than a population size below which a species may be at risk of jeopardy.

The stock size is relatively large (about 7,500 to 10,000, depending upon which model is used for the abundance estimate) and is growing at a rate that is nearly double the default maximum rate of growth for cetaceans (4 percent). The mortality of up to one individual humpback whale every year from the CNP stock is not expected to affect the growth rate in any appreciable way and is thus not expected to increase the risk of extinction for this stock and therefore population of which it is a part. That is, NMFS does not expect the proposed action to result in an appreciable reduction in the numbers, distribution, or reproduction of the North Pacific population of humpback whales. Therefore NMFS does not expect the proposed action to result in an appreciable reduction in the numbers, distribution, or reproduction in the species the population comprises.

7.2 Sperm Whales

The stressors, exposure, response, and risk steps of the effects analysis for sperm whales with regard to implementation of the proposed action are described below. The following information was used to conduct these analyses of the proposed action on sperm whales: the sperm whale Stock Assessment Reports (e.g., Carretta et al 2014), the sperm whale recovery plan (NMFS 2010c), and other sources cited below.

7.2.1 Stressors

The primary stressor of the Hawaii-based deep-set longline fishery on sperm whales is entanglement with fishing gear. 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 (Table 9).

Table 9. Summary of observed interactions between sperm whales and the Hawaii-based longline fisheries from 1995-2013 (Forney 2010). Seriousness of injuries were assessed under MMPA serious injury guidelines (Angliss and DeMaster 1998) and the new policy (NMFS 2012b) for interactions that occurred after 2007 (Bradford and Forney 2013).

Hawaii Longline Fishery	Date	EEZ	NMFS-Determined Severity
Mixed (swordfish target)	5/22/1999	Inside	Not serious
Experimental	4/21/2002	Outside	Not serious
Deep Set	5/27/2011	Inside	L10 (prorated .75 SI)

One interaction has been observed in the deep-set fishery (Table 9). The observer was not able to determine the nature of the entanglement/hooking due to the mainline parting. The branchline, leader and hook remained attached to the animal after the mainline parted according to the observer's description. In addition, approximately 25-30 feet of mainline may have been still attached. Further analyses of these interactions by NMFS using the serious injury guidelines (NMFS 2012b) were prorated using the L10 criterion which means there was evidence of an entanglement but no further details to support a more specific injury category, and was prorated as 0.75 serious injury (Bradford and Forney 2013).

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 shallow-set 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).

7.2.2 Exposure

Since 2004, there has been one observed interaction with a sperm whale resulting in an estimated total of 6.3870 sperm whale interactions with the deep-set fishery (Table 7). In order to predict the future level of interactions with sperm whales and given such low numbers of historic interactions, NMFS used a conditional binomial model to predict future interaction levels (McCracken 2014b). The proposed action is defined as approximately 1,305 trips, and 18,592 sets, with 46,117,532 hooks annually, thus the number of sperm whales that are likely to be entangled as a result of interactions with longline gear associated with the proposed action is less than three[16] (rounded up from 2.8) annually (McCracken 2014b).

7.2.3 Response

NMFS rates the severity of marine mammal interactions with fishing gear using serious injury guidelines developed for the MMPA as described in the humpback whale section above. Of the three interactions of sperm whales with the Hawaii-based longline fisheries since 1994 (one in deep-set, two with vessels targeting swordfish), two were 'not serious,' and 0.75 was 'serious' (Table 9).

In the exposure analysis above, sperm whale exposure to the proposed action is estimated to result in less than three entanglements annually. Of the three interactions that have occurred since 1994 in the Hawaii-based longline fisheries, one resulted in a prorated serious injury of 0.75. It is estimated that the annual M&SI will be 2 (rounded from 1.4) based on past M&SI determinations as described in the humpback whale section above (McCracken 2014b). Therefore, the proposed action is expected to kill up to two sperm whales annually from the Hawaii stock.

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[16] A sensitivity analysis was done by McCracken to account for variability in the level of effort based on trips. The number analyzed was between 1,300 and 1,525 trips with the same level of sets and hooks per trip (McCracken 2014a). The level of hook effort observed in 2013 was approximately 1percent higher than used in the predictive model and this level of hook effort corresponds to 1473 trips as described in the analysis (McCraken pers comm.). This number of trips falls within the range projected for sperm whales and below the 3 animals used in the BiOp analysis (Figure 19 of McCracken 2014a).

7.2.4 Risk

The mortality and serious injury of Hawaii sperm whales due to the proposed action is estimated to be less than two whales every year. This would not approach the stock's Potential Biological Removal (PBR), which is 10.2 animals. As described more fully in 7.1.4, PBR is a calculation that is used under the MMPA to evaluate impacts to stocks and is used in this analysis to help inform our ESA jeopardy analysis, but is not the sole basis for the decision. Here, the stock size abundance estimate is 3,354 (Carretta et al. 2014). Thus, the mortality of up to two individual sperm whales every year from the Hawaii stock is 0.079 percent (2÷ 3,354 *100= 0.060) of the stock, which would not have a meaningful affect on the stock's ability to achieve its OSP, and therefore negligible. This negligible impact to the stock, which is only a small portion of the population, is not expected to increase the risk of extinction for this population, and therefore the species. That is, NMFS does not expect the proposed action to result in an appreciable reduction in the numbers, distribution, or reproduction of sperm whales.

7.3 Main Hawaiian Islands Insular False Killer Whale

The stressors, exposure, response, and risk steps of the effects analysis for false killer whales with regard to implementation of the proposed action are described below. The following information was used to conduct these analyses of the proposed action on false killer whales: the status review of Hawaiian Insular False Killer Whales under the ESA (Oleson et al. 2010), the final listing rule (NMFS 2012c), the final take reduction plan (NMFS 2012d), the Hawaiian Islands False killer whale Stock Assessment Reports (Carretta et al. 2013), and other sources cited below.

7.3.1 Stressors

The primary stressor of the Hawaii-based deep-set longline fishery on the MHI IFKW DPS is hooking and entanglement with fishing gear. MHI IFKW are present in the action area. However, the longline fishery operates outside of their core range where they are uncommon and false killer whales from other stocks are also present (i.e. NWHI and pelagic false killer whale stocks). As previously described (section 5.2.3.3), interactions are prorated based on stock densities. Interactions between the deep-set longline fishery and MHI IFKW are rare and unpredictable events. Since 2005, there have been six observed possible interactions between MHI IFKW and the deep-set longline fleet. (Tables 10,11), based on their location in the overlap zone.

Table 10. Summary of observed interactions between false killer whales and unidentified blackfish in the overlap zone with the Hawaii-based deep-set longline fishery from 2005-June 2014 (Forney 2010, Bradford and Forney 2013). Seriousness of injuries were assessed under MMPA serious injury guidelines (Angliss and DeMaster 1998, NMFS 2012b). 2012 from NMFS 2014h.

Species	Date	NMFS-Determined Severity
Blackfish	10/21/2005	Serious
Blackfish	10/24/2006	Serious
False Killer Whale	11/11/2006	Not Serious
Blackfish	10/12/2011	Serious
False Killer Whale	10/29/2012	Serious
False Killer Whale	12/25/2012	Serious

Table 11. Table showing number of observed interactions with unidentified blackfish and false killer whales within the range of the MHI IFKW, and then total after proration to species and stock and expansion to a fleet-wide estimate (NMFS 2013c, NMFS SARs 2006-2013).

Year	Observed Interactions FKW/Blackfish	percent Observer Coverage	Total Estimated MHI IFKW M&SI
2005	0/1	26.10	0.71
2006	1/1	21.20	2.72
2007	0/0	20.10	0
2008	0/0	21.70	0
2009	0/0	20.60	0
2010	0/0	21.10	0
2011	0/1	20.30	0.8920
2012	2/0	20.40	4.89
2013	0	20.40	0
2014	0	14.30*	0

*This level of observer coverage is for the first quarter only (January 1-March 30, 2014).Coverage in the overlap area was approximately 22.4 percent

Table 11 gives the numbers by year of six deep-set fishery interactions with false killer whales and unidentified blackfish inside the overlap zone, i.e., from 40 to 140 km of shore from 2005-2014. All six of the deep-set fishery interactions with false killer whales and unidentified blackfish from 2005-2012 occurred between October 1 and December 31, when NMFS allowed longline fishing closer to the MHI in the seasonally-contracted MHI longline prohibited area. Effective December 31, 2012, the False Killer Whale TRP revised the MHI Longline Fishing Prohibited Area to remove the seasonal contraction of the boundary. Five of the six interactions occurred in the area that is now closed to longline fishing year-round. A single interaction with an unidentified blackfish (2006) occurred 0.84 nm offshore of the now-year-round longline prohibited area boundary.

The 2011 and 2012 Pacific SARs provide M&SI estimates (from 2005-2010) as rounded whole numbers of animals, but estimate the 5-year average of M&SI (from the same period), based on the unrounded annual estimates (Carretta et al. 2012, 2013). Table 11 provides unrounded estimates of total annual interactions in the deep-set fishery from NMFS' PIFSC during 2005-2014. NMFS has determined that the vast majority of the interactions with false killer whales are M&SI. As described previously, at the recommendation of the FKWTRT, the FKWTRP removed the seasonal boundary change from October 1 through January 31 each year, for the purpose of reducing impacts to MHI IFKW. The MHI Longline Fishing Prohibited Area includes over 74 percent of the 40-140 km overlap zone. That percentage does not include the additional core range shoreward of 40 km around the MHI, which has been closed to longline fishing year-round since 1992.

7.3.2 Exposure

Since 2005, there have been three observed interactions with false killer whales and three with unidentified blackfish in the overlap area. The proposed action is defined as approximately 1,305 trips, 18,592 sets with 46,117,532 hooks annually, however the area of operation where the vessels may interact with the MHI IFKW is a very small area. Therefore NMFS anticipates that the maximum number of hooks that may be set in the open area which would expose MHI IFKWs is approximately 600,000 hooks[17]. NMFS expects that future interaction rates will be less than observed in the past because of the FKWTRP that was fully implemented on February 27, 2013, which included both gear modifications and area closures to minimize the frequency and severity of injuries resulting from longline interactions[18]. NMFS cannot accurately predict the number of false killer whale interactions that may occur in the fishery managed under the FKWTRP, and because the FKWTRP was only recently implemented, cannot assess the immediate impact on interaction rates. However, to support the Negligible Impact Determination (NID), and analysis for this biological opinion NMFS conducted several analyses to estimate future interaction (take) levels. Because these analyses necessarily relied on interaction rates prior to the implementation of measures under the FKWTRP, NMFS expects actual rates to be less than those modeled.

To estimate a future take rate, NMFS developed a Bayesian model; which uses observed data from past years and assumes that the conditions will be the same in the future (McCracken 2014b). The model uses the false killer whale takes that were observed within 140 km from the MHI from 2008-2012, which had the most consistent systematic sampling of observer coverage at 15 percent for each quarter. This time period included three interactions: two takes that were identified as false killer whales, and one that was identified only as a blackfish but that was estimated to have a 0.97 probability of being a false killer whale based on its location (McCracken 2010, 2014b). Additionally, a subsequent comparison of the model to a variety of other models that considered takes over a longer period and/or wider areas (e.g., the Hawaii EEZ, or the entire fishery) showed that the selected model (using takes and effort inside 140 km from 2008-2012) better predicted past observed takes inside the 140 km boundary (McCracken 2014b).

The model was used to predict the annual number of takes of false killer whale (all populations combined) that might occur in the portion of the MHI insular/pelagic overlap zone that is currently (i.e., post-TRP implementation) open to longline fishing, if the effort in future years remained similar to reported effort from 2008-2012 (McCracken 2014b). The take prediction for the portion of the MHI insular/pelagic overlap zone that is currently open to longline fishing is an average of 0.3 (rounded from 0.2953) insular false killer whale (MHI insular false killer whale) takes per year (McCracken 2014b).

The take prediction done by McCracken (2014b) could only take into account the revised year-round closure implemented by the TRP. Other measures that were introduced by the TRP to

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[17] In 2013 there were 98 trips with 248 sets containing 344,808 hooks in the open area, which is the first year of the new closure based on the implementation of the FKWTRP. In addition there were no observed interactions with false killer whales or unidentified blackfish in the area and observer coverage was 22.4percent in the open area (McCracken pers comm..).

[18] The FKWTRP had a two part implementation; most of the plan went into effect on December 31, 2012, including the change to the MHI Longline Fishing Prohibited Area, described above; the remaining requirements pertaining to deep-set longline fishery gear went into effect on February 27, 2013.

reduce interactions and M&SI could not be incorporated into the quantitative model. Therefore we assessed these changes in a qualitative manner. Other measures for the deep-set longline fishery include a requirement for all vessels to use only circle hooks with a maximum wire diameter (thickness) of 4.5 mm and a maximum offset of 10 degrees; monofilament leaders and branch lines with a minimum 2.0 mm diameter, and a minimum breaking strength of 400 lb for any other material used in the construction of a leader or branch line. The FKWTRP also includes additional training on marine mammal mitigation techniques at protected species workshops, requirements to post placards on marine mammal handling and release information on all Hawaii-based longline vessels, and a requirement that the captain of the longline vessel supervise the handling and release of any hooked or entangled marine mammal. These measures could not be incorporated into the models to predict the future levels of take and M&SI, however other studies have been conducted on the use of circle hooks to reduce interactions and M&SI. The new gear requirements may reduce the future take by up to 6 percent (Forney 2010). When taking this into consideration, we anticipate the future level of take to be 0.2696 (i.e., 6 percent reduction from 0.2953) (NMFS 2014e). The other gear requirements and training are also anticipated to reduce the impact but cannot be in any way quantified, therefore we anticipate the take and M&SI predicted under the model to be the maximum impact to the DPS from the proposed action.

7.3.3 Response

NMFS rates the severity of marine mammal interactions with fishing gear using serious injury guidelines developed for the MMPA. Seriousness of injuries was assessed under MMPA serious injury guidelines (Angliss and DeMaster 1998) and the new policy (NMFS 2012b) for interactions that occurred after 2007 (Bradford and Forney 2013). Of the six interactions of false killer whales and blackfish with the Hawaii-based longline fishery since 2005, five were ‘serious,’ and one was ‘not serious’ (Table 10) (Forney 2010).

In the exposure analysis above, MHI IFKW exposure to the proposed action is estimated to result in 0.2696 interactions annually or one approximately every three years. The anticipated M&SI, which is based on previous determinations from false killer whales and unidentified blackfish, is expected to be 0.2460 annually, or one M&SI approximately every four years (McCracken 2014b). Therefore, the proposed action is expected to result in up to one MHI IFKW mortality every four years.

7.3.4 Risk

The M&SI of MHI IFKW due to the proposed action is estimated to be one whale every four years (.2460 M&SI x 4=.981, round to 1). This level is just under but close to the stock’s Potential Biological Removal (PBR), which is 0.3 (rounded from 0.258) animals annually. As previously discussed, a stock’s PBR is defined by the MMPA as the maximum number of animals, not including natural mortalities, that may be removed from a marine mammal stock while allowing that stock to reach or maintain its optimum sustainable population. A stock’s optimum sustainable population (OSP), in turn, is defined by the MMPA as the number of animals which will result in the maximum productivity of the population or the species, keeping in mind the carrying capacity of the habitat and the health of the ecosystem of which they form a constituent element (16 U.S.C. 1362(3)(9)). 50 CFR 216.3 further explains that the optimum sustainable population is a population size which falls within a range from the population level of a given marine mammal species or stock which is the largest supportable within the ecosystem to the population level that results in maximum net productivity.

We do not expect that the removal of one IFKW every four years will interfere with the stock's ability to achieve its OSP. We acknowledge that other human-caused sources of mortality may occur that have not been quantified, such as from pollutants and state-managed fisheries. Nevertheless, we anticipate that real-time monitoring of the longline fishery's impacts, along with the above-described mitigation measures, will allow NMFS to monitor any unanticipated impacts to the species' condition. Accordingly, NMFS concludes that the mortality of close to one individual MHI IFKW every four years is not expected to increase the risk of extinction for this population. That is, NMFS does not expect the proposed action to result in an appreciable reduction in the numbers, distribution, or reproduction of the MHI IFKW.

7.4 Loggerhead Turtles

Stressors, exposure, response and risk steps of the effects analysis for loggerhead turtles with regard to implementation of the proposed action are described below. Loggerhead turtles directly affected by interactions resulting from the proposed action are from the North Pacific Ocean Distinct Population Segment (DPS). Direct and indirect effects of the action on this DPS are related to the base condition of the DPS in the Integration and Synthesis of Effects (Section 9). The following information was used to conduct these analyses of the proposed action on loggerheads: the 2004 BiOp (NMFS 2004a), the 2005 BiOp (NMFS 2005a), the 2006 BiOp (NMFS 2006), 2008 BiOp (NMFS 2008a), the 2011 Loggerhead DPS listing, the 2012 BiOp (NMFS 2012a), and other documents cited below.

7.4.1 Stressors

Longline fishing affects loggerhead turtles primarily by hooking, but also by entanglement and trailing of gear that remains attached to an animal. Deep-set longlining is done during the day and loggerheads generally feed at shallower depths than the gear is fished which makes them less susceptible to deep-set gear than shallow-set longline gear. Hooking may be external, generally in the flippers, head, beak, mouth, or internal, when the animal has attempted to forage on the bait, and the hook is ingested. 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 turtle by piercing the esophagus, stomach, or other organs, or by pulling organs from their connective tissue. Once the hook is set and pierces an organ, infection may ensue, which may result in death of the animal. If a hook does not become lodged or pierce an organ, it can pass through to the colon, and be expelled (NMFS 2004a, 2005a, 2008a).

Loggerheads also become entangled in fishing gear but not as frequently as becoming hooked (NMFS 2014 a, b). Entanglement in monofilament line (mainline or branchline) or polypropylene (float line) can result in substantial wounds, including cuts, constriction, or bleeding on any body part. In addition, entanglement can directly or indirectly interfere with mobility, causing impairment in feeding, breeding, or migration. 'Trailing line' refers to line that is left on a turtle after it has been incidentally caught and released, particularly line trailing from a hook. Turtles may swallow line trailing from a hook, which may block the gastrointestinal tract and cause other serious injuries. Trailing line can also become snagged on a floating or fixed object, entangling or further entangling the turtle, or the drag can cause the line to constrict around a turtle's appendage until the line cuts through it (NMFS 2004a, 2005a, 2008a).

7.4.2 Exposure

Loggerhead turtles are expected to be exposed to interactions directly caused by the proposed action due to hooking and entanglement by fishing gear deployed by the Hawaii-based deep-set longline fishery. This exposure can be quantified as the expected annual number of interactions. The proposed action would result in approximately 1,305 trips, and 18,592 sets with 46,117,532 hooks annually. Based on analysis of interactions that occurred between 2008-2014, percentNMFS[19] estimates that 18,592 sets with 46,117,532 hooks would result in 3 (rounded from 2.61[20]) loggerhead interactions annually (Table 12). McCracken (2014b) did an analysis to determine the future level of take based on observer data from 2008-2012 (this time frame was selected because it had the most consistent systematic sampling of observer coverage at 15 percent for each quarter), and estimated that the mean would be 3 annual interactions and the median was 2 annual interactions. In 2013 a slightly higher number of loggerheads were observed in the fishery and therefore we reanalyzed the future take levels to include the more recent data. The level of loggerhead exposure to the direct effects of the proposed action is 3 loggerhead interactions annually, which is the same mean calculated by McCracken (2014).

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[19] McCracken 2014b analyzed observer data from 2008-2012 to predict future take levels, however a higher number of turtles were observed in 2013 than any other year so this new information was incorporated into the future interaction level (Table 12).

[20] Because the number of hooks in the proposed interaction is necessarily based on prediction, we also considered exposure based on a one percent variation in the expected number of annual hooks. This results in 2.66 annual loggerhead interactions, an insignificant change in the expected level of interactions and mortalities.

Table 12. Table showing the number of observed interactions with loggerheads, two different total estimates by year, and the total number of interactions per 1000 hooks set. McCracken estimates were used where available for all calculations; the other estimated interaction using the expansion factor, was only used when McCracken's estimate was not available (McCracken 2009 a, 2009b, 2010, 2011, 2012, 2013, 2014a).

Year	Observed	Estimated McCracken	% Observer Coverage	Expansion Factor[b]	Estimated Interactions[c]	Total Hooks	Turtles per 1000 hooks
2008	0	0	21.70	4.61	0	40,063,212	0.0000000
2009	0	0	20.60	4.85	0	38,177,005	0.0000000
2010	1	6	21.10	4.74	5	37,244,654	0.0001611
2011	0	0	20.30	4.93	0	40,022,142	0.0000000
2012	0	0	20.40	4.9	0	44,163,002	0.0000000
2013	2	11	20.40	4.9	10	46,769,514	0.0002352
2014[a]	0	NA	14.30	6.99	0	2,646,427	0.0000000
Total	3				15		
Average	0.48				2.4		0.0000566
Future	46,117,532 * 0.0000566/1000=2.61 annually						

[a] This is for (Jan1-April 30, 2014[21]); observer coverage for the first quarter Jan 1-March 31; was all that was available at the time. The total hooks was based on observer program quarterly report for observed hooks in the first quarter and an estimated number of hooks observed through April 30, 2014 based on the number of vessels that departed and the average number of hooks observed in the first quarter (McCracken pers comm.). There were no observed loggerhead turtle interactions through the end of the second quarter (June 30, 2014) so the time frame that was analyzed previously remains representative for the first two quarters; the preliminary summary of the number of hooks observed and the observer coverage reported in quarter two does not materially change our analysis.

[b] 100 ÷ observer coverage. E.g., for 2010, 100/21.10 = 4.74

[c] (Observed interactions) x (Expansion factor). E.g., for 2010, 1(4.74) = 5.

7.4.3 Response

The response to stressors that can be quantified in the proposed action is the number of mortalities that can be expected to result from interactions with fishing gear from the Hawaii deep-set fishery. Loggerhead response to the predicted exposure (three interactions annually) from the proposed action can be converted to the annual number of estimated mortalities resulting from this exposure. Since there are so few observed interactions in the deep-set fishery with loggerhead turtles and there have been no major gear modifications to the fishery that are known to impact sea turtles, all historic interactions were used to calculate the post-hooking mortality rate. There have been eight loggerhead interactions observed in the deep-set fishery from 1994 through 2014 and based on NMFS' post-hooking mortality criteria (Ryder et al. 2006), post-hooking mortality of loggerheads in this fishery is 72.0 percent (NMFS 2014a). Using this post-hooking mortality rate, three interactions annually would lead to 2.16, (round to three) loggerhead mortalities. However, in order to estimate the risk that the proposed action poses to the North Pacific loggerhead DPS, an analysis was done by NMFS (Van Houtan 2013, 2014) to determine the number of adult females removed from the DPS. Adult females are the only component of the DPS for which data are available, from counts of adult females on nesting beaches. The response to the population from three interactions must be quantified in terms of adult females in order to interpret the population assessment. As explained below, three

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[21] There were no more observed loggerheads through the second quarter, which ended June 30th so no updates were necessary for 2014 through the second quarter.

loggerhead turtle interactions equate to an estimate of 0.18 adult females annually, which equates to one adult female mortality every 5.4 years (Van Houtan 2014).

The deep-set fishery interacts with male and female loggerheads, and most of these are juveniles. In order to estimate the number of adult female mortalities that would occur if there were three interactions, two adjustments must be applied to the calculation above: (1) the proportion of females in the adult population; and (2) the adult equivalent represented by each juvenile interaction. These adjustments are described in greater detail below.

The proportion of females in the adult population.
We assume the sex ratio of the North Pacific loggerhead population is 50:50 (Conant et al 2009, Van Houtan 2013). Therefore, we estimate that approximately half of animals incidentally caught are females.

The adult equivalent represented by each juvenile interaction.
Vaughan (2009) estimated the relationship between age and size (straight carapace length or SCL) for loggerheads in the North Atlantic. Assuming similar loggerhead growth in the North Pacific, the Hawaii deep-set fishery described under the proposed action would be expected to interact most frequently with juvenile loggerheads (Van Houtan 2013). Age at first reproduction (AFR) (maturity) for this DPS is estimated at 25 years (Van Houtan and Halley 2011). NMFS applied a conversion formula to determine the annual effect of the action on adult females. In order to estimate adult equivalents that will be affected by the action, survival rates (Snover 2002) were applied to three distinct life stages that would occur before the AFR estimate of 25 years. The three survival rates applied to convert juveniles to adults were 0.81, 0.79, and 0.88 (Snover 2002; Van Houtan 2011, 2013). Instead of assigning all bycatch to an average demographic by size, and therefore age, the estimated annual female mortality is split among individuals observed in the fishery from 2004 to July 2013, which provides an exact demographic match to the fishery interaction records and avoids potential bias in the adult nester equivalent (ANE) calculation (Van Houtan 2013). Three juvenile mortalities results in the annual removal of the equivalent of 0.18 adult females which is analogous to incurring a single adult female mortality every 5.4 years for loggerheads (Van Houtan 2014).

In the above calculations, NMFS applied precautionary assumptions to all variables to ensure that the action's maximum impact was accounted for, so as to protect the listed species. We used the maximum potential number of interactions per year (three) rather than a potential scenario of less than three interactions per year. For example, the maximum number of loggerhead interactions that has been observed in a year in the deep-set fishery since 2008 has been two, and the mean annual number of actual interactions for the 6.25 -year period 2008 -April 2014 was 0.48 (3 / 6.25 years)(Table 5). By projecting that the proposed action will result in approximately 1,305 trips, 18,592 sets with 46,117,532 hooks annually, we assume as many as three loggerhead interactions per year. The post-hooking mortality rate of 72.0 percent is based on application of current NMFS criteria[22] (Ryder et al. 2006) to the 8 observed interactions that have been observed in the deep-set fishery (NMFS 2014a).

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[22] Although NMFS continues to evaluate modifications to post-interaction mortality rates, the approved NMFS 2006 criteria represents the best available information at this time.

7.4.4 Risk

The response of loggerheads to interactions with gear deployed by the Hawaii-based deep-set fishery is considered to be the mortality of 0.18 adult females annually or one adult female every 5.4 years. The risk posed by this level of mortality to the North Pacific loggerhead population was assessed by Van Houtan (2013, 2014) for application to this opinion. Quantitative population viability analysis (PVA) modeling (i.e. classical PVA or a climate-based PVA as described above), works with discrete numbers; whole numbers and not partial individuals. Rounding the calculated ANE estimate up to a single individual would overestimate the fishery impacts by a factor of five, therefore no further quantitative modeling of the fishery impacts were recommended (Van Houtan 2013, 2014). This ANE estimate represents under 0.003 percent of breeding females (6,673) in the North Pacific loggerhead DPS (less than 1 in 36100) and in terms of population level significance this ANE has a negligible population influence (Van Houtan 2014).

7.5 Leatherback Turtles

The stressors, exposure, response, and risk steps of the effects analysis for leatherback turtles with regard to implementation of the proposed action are described below. Leatherback turtles directly affected by fishing interactions resulting are expected to be almost entirely from the western Pacific population. Direct effects of the action on this population, and any indirect effects on other populations, are related to the species as a whole in the Integration and Synthesis of Effects (Section 9). The following information was used to conduct these analyses of the proposed action on leatherbacks: the 2004 BiOp (NMFS 2004a), the 2005 BiOp (NMFS 2005a), the 2006 BiOp (NMFS 2006), 2008 BiOp (NMFS 2008a), the 2012 BiOp (NMFS 2012a) and other documents cited below.

7.5.1 Stressors

Due to morphological and behavioural differences between loggerhead and leatherback turtles, effects of longline fishing on leatherbacks are somewhat different than those on loggerheads. Entanglement and foul hooking are the most common primary effects of longline fishing on leatherbacks, whereas internal hooking is more prevalent in hardshell turtles, especially loggerheads. Leatherbacks seem to be more vulnerable to entanglement and foul hooking, possibly due to their morphology (large size, long pectoral flippers, and lack of a hard shell), their attraction to gelatinous organisms and algae that collects on buoys and buoy lines at or near the surface, or some combination of these and/or other reasons. However, internal hooking in leatherbacks has been documented in the Hawaii-based longline fishery. The effects of entanglement on leatherbacks are similar to those described above for loggerheads: substantial wounds and reduced mobility, causing impairment of feeding, breeding, or migration of the entangled individual. Besides entanglement and foul hooking, the other two primary effects of longline fishing on leatherbacks are internal hooking and trailing line, the effects of which are similar to those described above for loggerheads. Because leatherbacks have more delicate skin and softer tissue and bone structures than hardshell turtles, their risk from longline-related injury is considered to be higher (NMFS 2004a 2005a, 2006a, 2006b, and 2008a).

7.5.2 Exposure

Leatherback turtles are expected to be exposed to interactions directly caused by the proposed action, due to hooking and entanglement by fishing gear deployed by the Hawaii-based deep-set longline fishery. This exposure can be quantified as the expected annual number of interactions. The proposed action would result in approximately 1,305 trips, with 18,592 sets and 46,117,532 hooks annually. Based on the number of hooks set and the number of leatherback interactions between 2008-2014, which had the most consistent systematic sampling of observer coverage at 15 percent for each quarter; this level of effort is expected to result in up to 24 (rounded from 23.81[23]) leatherback interactions annually (Table 13). McCracken (2014b) did an analysis to determine the future level of take based on observer data from 2008-2012 (this time frame was selected because it had the most consistent systematic sampling of observer coverage at 15 percent for each quarter), and estimated that the mean level of interactions would be 11 annual interactions with a median of eight. However, in 2013 and 2014, a higher than average number of leatherbacks were observed in the fishery. NMFS cannot determine at this time whether this recent "spike" in interaction rates represents a statistical anomaly or a change in the impacts for the fishery. Accordingly, we reanalyzed the future take levels to include the most recent data, over 6.25 years (Table 13). Therefore, leatherback exposure to the effects of the proposed action is considered to be 24 leatherback interactions annually.

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[23] Because this is a dynamic fishery and the number of hooks is necessarily based on prediction we also considered exposure based on a one percent increase in the expected number of annual hooks. This results in 24.3 leatherbacks, an insignificant change in the expected level of interactions that does not appreciably change the expected level of annual interactions, or more importantly the number of mortalities.

Table 13. Table showing the number of observed interactions with leatherbacks, two different total estimates by year, and the total number of interactions per 1000 hooks set. McCracken estimates were used where available for all calculations; the other estimated interaction number was only used when McCracken estimate was not available (McCracken 2009a, 2009b, 2010, 2011, 2012, 2013, 2014a).

Year	Observed	Estimated McCracken	percent Observer Coverage	Expansion Factor[c]	Estimated Interactions[d]	Total Hooks	Turtles per 1000 hooks
2008	1	11	21.70	4.61	5	40,063,212	0.0002746
2009	1	4	20.60	4.85	5	38,177,005	0.0001048
2010	1	6	21.10	4.74	5	37,244,654	0.0001611
2011	3	14	20.30	4.93	15	40,022,142	0.0003498
2012	1	6	20.40	4.9	5	44,163,002	0.0001359
2013[a]	3	15	20.40	4.9	15	46,769,514	0.0003207
2014[b]	6	NA	14.30	6.99	42	2,646,427	0.0022672
Total	**16**				**92**		
Average[e]	2.56				14.72		0.0005163
Future	46,117,532 * 0.0005163/1000=23.81 annually						

[a] One turtle was caught in 2013 but counted in 2014, when the vessel arrived.

[b] This is for (Jan1-April 30, 2014[24]); observer coverage for the first quarter (Jan 1-March 31) was used since that was all that was available at the time. The total hooks was based on observer program quarterly report for observed hooks in the first quarter and an estimated number of hooks observed through April 30, 2014 based on the number of vessels that departed and the average number of hooks observed in the first quarter (McCracken pers comm.). There were no observed leatherback turtle interactions through the end of the second quarter (June 30, 2014) so the time frame that was analyzed previously remains representative for the first two quarters; the preliminary summary of the number of hooks observed and the observer coverage reported in quarter two does not change our analysis in any significant manner. For leatherbacks the number of turtles per 1000 hooks decreases slightly since there were more hooks observed throught the end of the second quarter but with no leatherback interactions observed. Since this number (turtles/1000 hooks) can fluctuate significantly when only looking at data on a per quarter basis, we base our projections using the slightly higher rate which is more precautioniary.

[c] 100 ÷ observer coverage. E.g., for 2008, 100/21.70 = 4.61.

[d] (Observed interactions) x (Expansion factor). E.g., for 2008, 1(4.61) = 5.

[e] Average is calculated by dividing total number by 6.25 since the data only includes a quarter of the year for 2014.

The Hawaii deep-set longline fishery interacts primarily with the Bird's Head component of the western Pacific leatherback turtle population. Studies suggest that the western Pacific population has a clear separation of migratory destinations for summer vs. winter nesters. Summer nesters move into the temperate North Pacific Ocean or into tropical waters of the South China Sea, whereas winter nesters move into temperate and tropical large marine ecosystems of the southern hemisphere (Benson et al. 2011). The majority of nesting in the summer months occurs on Bird's Head peninsula of Papua Barat, Indonesia, which includes four main nesting sites described above in section 5.3.1 (Dutton et al. 2007, Benson et al. 2011), which may account for up to 90 percent of the summer nesters for the western Pacific metapopulation (Hitipeuw et al. 2007, Benson pers. comm. 2011). The Bird's Head Peninsula is estimated to account for 75 percent of the total nesting activity of the western Pacific Population. There is some summer nesting occurring at other places such as PNG and the Solomons but it is at much lower numbers (Dutton et al. 2007, Benson et al. 2011). Since we can't say for certain what percentage of summer

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[24] There were no more observed leatherbacks through the second quarter, which ended June 30th so no updates were necessary for 2014 through the second quarter.

nesters come from the Jamursba-Medi component of the population with which the fishery interacts, we have considered a range of percentages from 69 percent - 90 percent (Snover 2008; Benson personal communication 2011).

7.5.3 Response

Leatherback response to predicted exposure (24 interactions annually) can be characterized as the annual number of mortalities estimated to result from this exposure. For the 35 leatherback interactions observed in the deep-set fishery since 1994, based on NMFS' post-hooking mortality criteria (Ryder et al. 2006), post-hooking mortality of leatherbacks in this fishery is 36.1 percent (NMFS 2014a). Using this post-hooking mortality rate, 24 interactions annually would lead to nine (rounded from 8.66) leatherback mortalities. However, in order to estimate the risk that the proposed action poses to the western Pacific leatherback population, we must first determine the number of adult females (ANE) that would be impacted. Adult females are the only component of the population for which data are available, from counts of adult females or nests on nesting beaches.

The deep-set fishery interacts with male and female leatherbacks, and they are predominantly juveniles (Van Houtan 2013). In order to estimate the number of adult females that would potentially be killed by 24 interactions, two adjustments must be applied to the calculation above: (1) the proportion of females in the adult population; and (2) the adult equivalent represented by each juvenile interaction. These adjustments are described in greater detail below (Table 14).

The proportion of females in the adult population.

The sex ratio of the western Pacific leatherback population is unknown, but studies of other leatherback populations suggest that sex ratio is not 50:50. Rather, these studies indicate more females than males in many sub-populations, hence NMFS estimates the sex ratio in the western Pacific population to be 65 percent female (NMFS 2008a, Snover 2008, Van Houtan 2011, Van Houtan 2013).

The adult equivalent.

Most leatherbacks interacting with the deep-set fishery are juveniles (Van Houtan 2013). The minimum size when western Pacific leatherback females first breed is estimated to be >120cm SCL[25] (Jones et al. 2011, Stewart et al. 2007, Van Houtan 2011).

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[25] Two studies found that the globabal average length of nesting was 147 cm SCl and the minimum size when turtles first start to breed is 121 cm SCL; this minimum size was used to determine adult equivalents.

Table 14. Annual Adult female leatherback mortality from the proposed action.

Variable	Estimate
Maximum annual interactions Post-hooking mortality Sex ratio	24 interactions 0.361 mortalities/capture 65:35 (female: male) = .65 females
Annual adult female mortalities # years for 1 ANE	0.20 4.9 years
Annual adult female mortalities from Bird's head component 69-90% # years for 1 ANE	0.138-0.18 7.2-5.6 years
Annual adult female mortalities from non- Bird's head component 10-31% # years for 1 ANE	0.02-0.062 50-16 years
Annual adult female mortalities Eastern Pacific component 4% # years for 1 ANE	0.008 125 years

*(24 captures)(0.361 mortalities/capture)= 8.664, round to 9 total mortalities.

To summarize, in order to estimate the response of leatherbacks to 24 interactions in terms of annual adult female mortalities, the interactions were first multiplied by the post-hooking mortality rate (0.361). In order to estimate the mortality of females/males from the proposed action, a 65/35 ratio was used.

The adult equivalent was determined using the discounting method (Van Houtan 2013), giving an estimate of 0.20 adult female mortalities annually or a single adult female mortality every 4.9 years from the Northern segment of the western Pacific nesting populations (Van Houtan 2013, 2014). Since we cannot say for certain what percentage of the summer nesters come from the Bird's head component of the population with which the fishery is interacting, we have considered a range of percentages from 69 percent - 90 percent (Snover 2008; Benson personal communication 2011). The total mortalities anticipated to the entire western Pacific population is 0.20. When we consider that 69 percent of the population is from the Bird Head's region then we anticipate 0.138 adult female mortality annually or one mortality every 7.2 years; if the Bird's head component comprises 90 percent of the interactions, the equivalent is 0.18 mortalities annually or one every 5.6 years. One adult female mortality is anticipated every 16.1 or 50 years when considering that 10-31 percent is from the non- Bird Head's region of the Western Pacific population (Table 14). Based on genetic analysis of historic interactions in the deep-set fishery there is also a possibility for interactions with leatherbacks from the eastern Pacific in very low levels. In order to be precautionary we assume that 4 percent of the leatherback interactions could be from the eastern Pacific (Table 2) and therefore consider that 4 percent of the ANE could be from this population, which equates to one adult female mortality every 125 years (Table 14).

NMFS applied precautionary assumptions to all variables to ensure that the action's maximum impact was accounted for, so as protect the listed species. Rather than using a 50:50 sex ratio based on absence of information for this population, a female ratio of 0.65 was used based on information from other leatherback populations. We used the maximum possible number of interactions per year (24) rather than a potential scenario of less than 24 interactions per year. For example, the maximum number of leatherback interactions that has occurred in a year in the deep-set fishery since 2008 is 42, which was based on six observed interaction that occurred between December 2013 and April 2014, with observer coverage at approximately 14 percent.

The mean annual number of actual interactions for the 6.25-year period 2008- April 2014 was 14.72 (Table 13). Between 2004 and 2013 the maximum number of interactions was 15 in 2013 based on three observed interactions. Between December 2013–April 2014, six leatherback turtles were observed with observer coverage at approximately 14 percent, therefore we estimate that there have been 42 interactions in that time period. Review of historic fishing effort does not indicate any differences in fishing that explains the increase during this time period. From Benson et al. 2011 and Roe et al. 2014, we know that the area where the leatherbacks were caught is a main corridor during that time frame but we have not seen the same levels of interactions in the past when fishing has occurred. It is possible that the interactions could have occurred due to turtles congregating in eddies or more turtles in the area due to unknown environmental or biological effects.

7.5.4 Risk

The response of leatherbacks to interactions with gear deployed by the Hawaii-based deep-set fishery is considered to be the mortality of 0.20 adult females annually or one adult female mortality every 4.9 years. The risk posed by this level of mortality to the Bird's Head region of the western Pacific leatherback population was assessed by Van Houtan (2013, 2014) for application to this opinion. Quantitative population viability analysis (PVA) modeling (i.e. classical PVA or a climate-based PVA as described above), works with discrete numbers; whole numbers and not partial individuals. Rounding the calculated adult female mortality estimate up to a single individual would overestimate the fishery impacts by a factor of five (Van Houtan 2013, 2014).

7.5.4.1 Bird Head's Component of the Western Pacific Population

The Bird Head's component makes up 75 percent of the western Pacific leatherback population (Dutton et al. 2007), and the majority of leatherbacks interacting with the Hawaii-based deep-set fishery are likely coming from this component due to its migration patterns (Van Houtan 2013). Previous estimates have determined that 69-90 percent of the impacts are occurring to the Bird's Head component due to migration patterns, with 10-31 percent of the impacts to the non Bird's Head component of the population. The risk assessment done by Van Houtan assigns all mortality to the Bird's Head component and determines that the ANE estimate of 0.20 represents 0.010 percent of breeding females in the western pacific (less than 1 in 9540), which has a negligible population influence (Van Houtan 2014).

7.5.4.2 Non-Bird Head's Component of the Western Pacific Population

The non-Bird Head's component makes up approximately 25 percent of the western Pacific leatherback population (Dutton et al. 2007), consisting of nesting sites in other locations of Indonesia, PNG, the Solomon Islands, and Vanuatu, as described in Section 5.4.1. As Table 14 explains, 10-31 percent of the leatherbacks interacting with the Hawaii-based deep-set fishery are expected to originate from the non-Bird Head's component, and effects to this component must be analyzed in order to determine the effects of the proposed action on the western Pacific population as a whole. Studies show that leatherbacks nesting in the winter in PNG, and the Solomon Islands, head southward to the western South Pacific Ocean or Tasman Sea when nesting is complete (Benson et al. 2011); therefore it is unlikely that they would be encountered in the action area of the proposed action. However there may also be some summer nesters at these locations as well and those turtles may move into the temperate North Pacific Ocean or into tropical waters of the South China Sea creating a potential to interact with the proposed action.

For leatherbacks nesting at sites in the Huon area of PNG, the most reliable trend information is from the 2006-2007 nesting season forward which appears to indicate a stable or slightly increasing trend, although there is not enough data to determine a reliable trend estimate. Sites currently monitored along the Huon coast occur during the winter (November to March) nesting season; however, anecdotal information from local villagers suggests some year-round nesting occurs in other locations.

No information exists regarding population trends in the Solomon Islands over time, but it is believed that local consumption of turtles and eggs has reduced nesting populations over the last few decades (Bellagio Steering Committee 2008, NMFS and USFWS 2013). Conservation activities at these sites are inconsistent and monitoring programs are still in development, hampered by limited local capacity and insufficient funding. One of the 37 foraging leatherbacks satellite tracked from California waters migrated to the Solomon Islands and nested at Santa Isabel Island in May providing evidence of summer breeding population linkage between the Solomons and California foraging habitats (Benson et al. 2011). One adult female mortality is anticipated every 16.1 or 50 years when considering that 10-31 percent is from the non- Bird Head's region of the Western Pacific population (Table 14), which is an extremely low level of impact.

7.5.4.3 Conclusion for Western Pacific Leatherbacks

In conclusion, the proposed action is not expected to reduce the western Pacific component of the population. The calculated adult nester equivalent (ANE) is anticipated to be approximately 0.20 annually or one female mortality every 4.9 years from all leatherback populations combined in the Pacific. Since the annual ANE is so small NMFS concludes that the influence on the Bird's Head component of the western Pacific population is negligible. Moreover, a quantitative model could not be run since the impact would have to be increased by five times the anticipated level to run the model with a discrete number as required (Van Houtan 2014). As a reference point the ANE represents 0.010 percent of breeding females in the western Pacific leatherback population or less than 1 in 9540 (Van Houtan 2013, 2014). That is, the Bird's Head component of the population is at a low risk of extinction as a result of the proposed action. This component of the population represents about 75 percent of the entire western Pacific population, however it accounts for the majority of leatherback interactions with the Hawaii deep-set fishery (69-90 percent). The remaining 25 percent of the population has less of a chance of interacting with the fishery because the majority of nesting occurs in the winter and these animals have different migratory routes that don't enter the proposed action area. Considering that the impact to the remainder of the western Pacific population is significantly smaller with an adult female mortality occurring every 16.1 – 50 years[26] we do not anticipate that this level of mortality would appreciably reduce the likelihood of survival and recovery of the non-Bird's Head component of the population. We believe that the proposed action will have a negligible impact on the risk to the Bird's Head component, the non-Bird's Head component, and therefore the western Pacific leatherback population as whole.

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[26] If population accounts for 31percent of anticipated ANE than a female mortality would occur every 16.1 years and if the population is 10percent then it would be equivalent to one every 50 years.

7.5.4.4 Conclusion for Eastern Pacific Leatherbacks

Eastern Pacific adult leatherbacks migrate south to foraging areas off South America in the eastern tropical Pacific and the southeastern Pacific, which is outside of the range of the action area (Shillinger et al. 2008, Bailey et al. 2012); however less is known on the behavior of non-breeding females and of males. Since there has been one confirmed interaction with a juvenile leatherback in the deep-set fishery, which occurred in 1995 we assume that 4 percent (Table 2) of the leatherback mortalities could be from the eastern Pacific leatherback which could represent individuals from nesting aggregations along the coast of Mexico, Costa Rica, or Nicaragua. This population of eastern pacific leatherbacks has declined and only a small population exists in the region. Due to the migration routes of this population and the low numbers of turtles remaining it is unlikely that there will be future interactions with this population. However to be precautionary since there has been one known interaction in the past, 20 years ago, we assume that 4 percent of the adult female mortalities could come from this population. That percentage would equal 0.008 (0.2 *4 percent) adult female mortalities, or one (1/0.008) in 125 years. That level of mortality is negligible and does not pose any appreciable risk to the population.

7.5.4.5 Conclusion for Leatherbacks

The analysis done on the smaller sup-populations of the leatherback species in the Pacific, which are the components that the action will affect, confirms that the proposed action presents negligible risk to those populations. We therefore conclude that the proposed action presents negligible risk to the overall species. We do not anticipate that the level of mortality would appreciably reduce the likelihood of survival and recovery of the species as a whole.

7.6 Olive Ridley Turtles

The stressors, exposure, response, and risk steps of the effects analysis for olive ridley turtles with regard to implementation of the proposed action are described below. The following information was used to conduct these analyses of the proposed action on olive ridleys: the 2004 BiOp (NMFS 2004a), the 2005 BiOp (NMFS 2005a), the 2006 BiOp (NMFS 2006), 2008 BiOp (NMFS 2008a), the 2012 BiOp (NMFS 2012a), and other documents cited below.

7.6.1 Stressors

Longline fishing affects olive ridleys primarily by hooking, but also by entanglement and trailing of gear. Olive ridleys are the most commonly-caught sea turtle species in the Hawaii-based deep-set longline fishery (NMFS 2014a), which fishes between 40 and 350 m of depth, and rarely interact with shallow-set gear, most likely because of a combination of deep-foraging and low density in temperate waters where fishing for swordfish occurs.

7.6.2 Exposure

Olive ridley turtles are expected to be exposed to interactions directly caused by the proposed action, due to hooking and entanglement by fishing gear deployed by the Hawaii-based deep-set longline fishery. This exposure can be quantified as the expected annual number of interactions. The proposed action would result in approximately 1,305 trips, 18,592 sets with 46,117,532 hooks annually. Based on the number of Olive ridley interactions that occurred between 2008-2012 (this time frame was selected because it had the most consistent systematic sampling of observer coverage at 15 percent for each quarter), and an analysis done by McCracken (2014b), NMFS expects up to 33 olive ridley interactions annually[27] (McCracken 2014b). Therefore, olive ridley exposure to the effects of the proposed action is considered to be 33 interactions annually.

Table 15. Table showing the number of observed interactions with olive ridley turtles, total caught estimated by McCracken where available (McCracken 2009a, 2009b, 2010, 2011, 2012, 2013, 2014a), and total estimate using expansion factor.

Year	Observed	Estimated McCracken	percent Observer Coverage	Expansion Factor[c]	Estimated Interactions[d]
2008	2	18	21.70	4.61	9
2009	4	18	20.60	4.85	19
2010 [a]	4	10	21.10	4.74	19
2011	7	36	20.30	4.93	34
2012	6	34	20.40	4.9	29
2013	9	42	20.40	4.9	44
2014[b]	5	NA	14.30,25.2	6.99;3.97	23
Total	37	158			177
Average	5.69	26.33			27.23

[a] One turtle was caught in American Samoa but the trip was included as a Hawaii longline trip by the observer program based on their departure.

[b] This is for (Jan 1-June 30, 2014). One was observed in the first quarter which =7 based on observer coverage, and four were observed in the second quarter which=16 based on observer coverage.

[c] 100 ÷ observer coverage. E.g., for 2008, 100/21.70 = 4.61.

[d] (Observed interactions) x (Expansion factor). E.g., for 2008, 2(4.61) = 9.

7.6.3 Response

Olive ridley response to predicted exposure (33 interactions annually) can be characterized as the annual number of mortalities estimated to result from this exposure. For the 98 olive ridley interactions observed in the deep-set fishery from 1994 through 2014, based on NMFS' post-hooking mortality criteria (Ryder et al. 2006), post-hooking mortality of olive ridleys in this fishery is 94.7 percent (NMFS 2014a). Using this post-hooking mortality rate, 33 interactions annually would lead to 32 (rounded from 31.25) olive ridley mortalities (either sex, all ages).

7.6.4 Risk

As shown by genetic samples of olive ridleys from the deep-set fishery (Table 2), individuals may come from either the eastern or western Pacific populations. Since we estimate a total of 32 olive ridleys will be killed annually by the proposed action, and 82 of the 106 (77 percent) genetic samples analyzed so far were from the eastern Pacific population, and 24 (23 percent) were from the western Pacific we expect 25 (32*77 percent=24.64) turtles from the eastern Pacific population to be killed every year, and 8(32*23 percent=7.36) olive ridleys from the western Pacific population to be killed every year. The eastern Pacific population has at least one million adult nesting females (Table 5). If we assume that fifty percent of the adults killed are

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[27] The analysis done by McCracken (2014b) used data from 2008-2012 and the projected number of 46,117,532 hooks was used in the model to project future levels of olive ridley interactions. In 2013, 42 olive ridleys were estimated to be caught (McCracken 2014b) and the number of hooks set was one percent higher than projected; however this interaction level falls within the bulk of the predictive densities of annual interactions in the probability distribution (Figure 10 of McCracken 2014b) suggesting that the catch rate has not increased.

female than the impact would be 0.0000013 percent of the adult female population that would be affected. This level of impact is extremely small. The western Pacific population has at least 33,500 adult nesting females (Table 4). If we assume that fifty percent of the adults killed (8) are female the impact would be 0.012 percent of the adult female population that would be affected. This level of impact is extremely small. The risk to both populations from the proposed action is considered negligible and therefore to the species.

7.7 Green Turtles

The stressors, exposure, response, and risk steps of the effects analysis for green turtles with regard to implementation of the proposed action are described below. The following information was used to conduct these analyses of the proposed action on green turtles: the 2004 BiOp (NMFS 2004a), the 2005 BiOp (NMFS 2005a), the 2008 Hawaii bottomfish BiOp, (NMFS 2008c), 2008 Shallow-set BiOp (NMFS 2008a), the 2012 BiOp (NMFS 2012a), and other documents cited below.

7.7.1 Stressors

Longline fishing affects green turtles primarily by hooking, but also by entanglement and trailing of gear. Historically, the longline fishery has been more likely to hook green turtles externally than to entangle them or hook them internally. Juvenile and adult interactions both occur (NMFS 2005). In addition, because green turtles recruit to nearshore habitat in the MHI, and green turtles are common in shallow MHI waters, fishing vessels traveling to and from port may strike a green turtle during transit (NMFS 2008c).

7.7.2 Exposure

Green turtle interactions in the deep-set fishery are rare, unpredictable events. Since 2004, there have been seven observed interactions with green turtles in the deep-set fishery. Based on the number of observed interactions between 2008 and 2012 (this time frame was selected because it had the most consistent systematic sampling of observer coverage at 15 percent for each quarter), and an analysis done by McCracken (2014b), NMFS estimates that there could be up to three[28] (rounded from 2.97) interactions with green turtles annually if the Hawaii deep-set fleet were to make approximately 1,305 trips, with18,592 sets and 46,117,532 hooks annually (McCracken 2014b).

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[28] A sensitivity analysis was done by McCracken to account for variability in the level of effort based on trips. The number analyzed was between 1,300 and 1,525 trips with the same level of sets and hooks per trip (McCracken 2014b). The level of hook effort observed in 2013 was approximately 1percent higher than used in the predictive model and this level of hook effort corresponds to 1473 trips as described in the analysis (McCraken pers comm.). This falls within the range projected for green sea turtles and below the 3 animals used in the BiOp analysis (Figure 19 of McCracken 2014b).

Table 16. Table showing the number of observed interactions with green turtles, total caught estimated by McCracken where available (McCracken 2009a, 2009b, 2010, 2011, 2012, 2013, 2014a), and total estimate using expansion factor. Expansion factor is used in instances when estimates are not yet available by McCracken. Both estimates are shown here for comparison purposes.

Year	Observed	Estimated McCracken	percent Observer Coverage	Expansion Factor [b]	Estimated Interactions[c]
2008	0	0	21.70	4.61	0
2009	0	0	20.60	4.85	0
2010	1	1	21.10	4.74	5
2011	1	5	20.30	4.93	5
2012	0	0	20.40	4.9	0
2013	1	5	20.40	4.9	5
2014 [a]	1	NA	25.20	3.97	4
Total	4	11			19
Average	0.62				2.92

[a] This is for (Jan 1-June 30, 2014). The turtle was observed caught in quarter 2, which had 25.2 percent coverage reported by the observer program.

[b] 100 ÷ observer coverage. E.g., for 2008, 100/21.70 = 4.61.

[c] (Observed interactions) x (Expansion factor). E.g., for 2010, 1(4.74) = 5.

The proposed action may also affect green turtles due to collisions with boats in nearshore waters around the MHI. The entire Hawaii-based longline fishery (deep-set and shallow-set combined) took approximately 1,500 trips annually from 2005-10, with only a small fraction shallow-set trips (≈100 trips/yr) (NMFS observer annual reports). The proposed action is expected to result in approximately 1400 trips per year. The number of green turtles likely to be killed due to boat collisions from the Hawaii bottomfish fishery was estimated in a March 18, 2008, biological opinion (NMFS 2008c). Using the 6-step methodology in the HI bottomfish opinion (Figure 3, p.25), and substituting 1400 trips per year for the 71,800 bottomfishing trips per year, then completing Steps 3 and 4, the number of annual green turtle mortalities estimated to result from boat collisions from deep-set longline boats is effectively zero (0.09).

7.7.3 Response

Green turtle response to the predicted exposure (three interactions annually) from the proposed action can be converted to the annual number of estimated mortalities resulting from this exposure. For eleven green interactions observed in the deep-set fishery from 1998 through 2014 based on NMFS' post-hooking mortality criteria (Ryder et al. 2006), post-hooking mortality of greens in this fishery was 93.2 percent (NMFS 2014a). Using this post-hooking mortality rate, three interactions annually would lead to three (3 x 0.932 = 2.796 + 0.09 mortality/yr from deep-set boat collisions = 2.886) green mortalities.

7.7.4 Risk

As shown by the 19 genetic samples of green turtles from the deep-set fishery (Table 2), individuals may come from either the western, central, or eastern Pacific nesting populations. Since we estimate a total of three individuals will be killed annually by the proposed action, two (3*68 percent=2.04) turtles from the eastern population are expected to be killed every year, and one from either the western or central pacific population annually.

The estimated number of nesting females for the eastern Pacific nesting aggregation is 20,112 (NMFS and USFWS in Press). The eastern Pacific nesting aggregation is increasing (see Section 5.2.7.1). As mentioned previously there could be two mortalities from the Eastern Pacific nesting aggregation annually. These two mortalities are from any age or sex. In order to analyze the impact to the adult female population we assume that they are adult and have a 50 percent chance of being a female. Therefore, one mortality from 20,112 nesting females represents 0.005 percent of the population (1/ 20,112 * 100= 0.005 percent). This level is barely detectable and therefore negligible to the overall nesting aggregation.

The estimated number of nesting females for the Hawaii nesting aggregation is 3,846 (NMFS and USFWS in Press). The Hawaii nesting aggregation is increasing (see Section 5.2.7.1). As mentioned previously there could be one mortality from the Hawaii nesting aggregation annually. This one mortality is from any age or sex. In order to analyze the impact to the adult female population we assume that it is an adult and has a 50 percent chance of being a female. Therefore a 0.5 mortality from 3,846 nesting females represents 0.013 percent of the population (0.5/3,846 *100=0.013). This level is barely detectable and therefore negligible to the overall nesting population and is likely an overestimate since we do not expect a mortality from this population every year, but we anticipate up to one mortality from one of four different nesting aggregations.

As described earlier the remaining western Pacific has been described as three separate regional nesting aggregations. The central west Pacific Ocean has insufficient information to adequately describe abundance and populations trends for many areas. Limited information suggests a nesting population decrease in some portions of this region like the Marshall Islands, or unknown trends in other areas. The nesting at Chichijima, Japan has shown an increasing trend and the estimated number of nesting females for this area is 6,518 (NMFS and USFWS in Press).

As mentioned previously there could be one mortality from this population annually. This one mortality is from any age or sex. In order to analyze the impact to the adult female population we assume that it is an adult and has a 50 percent chance of being a female. Therefore a 0.5 mortality from 6,518 nesting females represents 0.0077 percent of the population (0.5 female/6518). This level is negligible to the overall nesting population and is likely an overestimate since we do not expect a mortality from this aggregation every year, but we anticipate up to one mortality from only one of the four different nesting aggregations.

The estimated number of nesting females for the Southwest Pacific Ocean nesting aggregation is 83,058 (NMFS and USFWS in Press). 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 interannual 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).

As mentioned previously there could be one mortality from this nesting aggregation annually. This one mortality is from any age or sex. In order to analyze the impact to the adult female population we assume that it is an adult and has a 50 percent chance of being a female. Therefore a 0.5 mortality from 83,058 nesting females represents 0.0006 percent of the population. This level is negligible to the overall nesting population and is likely an overestimate since we do not expect a mortality from this aggregation every year, but we anticipate up to one mortality from only one of the four different nesting aggregations.

The estimated number of nesting females for the Central south Pacific nesting aggregation is 2,902 (NMFS and USFWS in Press). Green turtle temporal population trends in the Central South Pacific are poorly understood, with not even a single nesting site having five contiguous years of standardized monitoring that span entire nesting seasons. Partial and inconsistent monitoring from the largest nesting site in this aggregation, 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 percent of the population abundance at Scilly Atoll alone. Nesting abundance is reported to be stable to increasing at Tongareva Atoll (White and Galbraith 2013).

As mentioned previously there could be one mortality from this nesting aggregation annually. This one mortality is from any age or sex. In order to analyze the impact to the adult females in the population we assume that it is an adult and has a 50 percent chance of being a female. Therefore a 0.5 mortality from 2,902 nesting females represents 0.0017 percent of this nesting aggregation. This level is negligible to the overall nesting population and is likely an overestimate since we do not expect a mortality from this nesting aggregation every year, but we anticipate up to one mortality from only one of the four different nesting aggregations.

7.8 Scalloped Hammerhead Sharks

The stressors, exposure, response, and risk steps of the effects analysis for scalloped hammerhead sharks with regard to implementation of the proposed action are described below.

7.8.1 Stressors

Longline fishing affects scalloped hammerhead sharks primarily by hooking, but also by entanglement and trailing of gear. Historically, the longline fishery has very few interactions with scalloped hammerhead sharks and the majority are released alive.

7.8.2 Exposure

Scalloped hammerhead shark interactions in the deep-set fishery are rare, unpredictable events. Since 2004, there have been three observed interactions with scalloped hammerhead sharks in the deep-set fishery (Table 17). Based on the average number observed between 2004 and 2014, NMFS estimates that there could be up to two interactions with scalloped hammerhead sharks annually (Table 17).

Table 17. Table showing the number of observed interactions from 2004-March 2014, with the Indo-west Pacific scalloped hammerhead DPS, estimated, and total estimate using expansion factor.

Year	Observed	percent Observer Coverage	Expansion Factor [b]	Estimated Interactions[c]
2004	2	24.60	4.07	9
2005	0	26.10	3.83	0
2006	0	21.20	4.72	0
2007	1	20.10	4.98	5
2008	0	21.70	4.61	0
2009	0	20.60	4.85	0
2010	0	21.10	4.74	0
2011	0	20.30	4.93	0
2012	0	20.40	4.90	0
2013	0	20.40	4.90	0
2014*	0	14.30	6.99	0
Average				1.3

*Data only available through second quarter so 0.50 years used for 2014 to calculate average.

7.8.3 Response

Scalloped hammerhead shark response to the predicted exposure (2 interactions annually) from the proposed action can be converted to the annual number of estimated mortalities resulting from this exposure. Of 33 scalloped hammerhead interactions observed in the deep-set fishery since 2004 from the Central Pacific and Indo-west Pacific DPSs, 21 were released alive, 12 were dead (NMFS unpublished observer data). From this data, NMFS estimates that the mortality rate for scalloped hammerheads is 36 percent. Therefore we estimate the response to be up to one (2*36 percent=0.72) mortality annually.

7.8.4 Risk

The response of scalloped hammerheads to interactions with gear deployed by the Hawaii-based deep-set fishery is considered to be the mortality of up to one annually. The Indo-west Pacific scalloped hammerhead DPS occurs in a large area and abundance estimates for the entire DPS is unavailable. There are some areas where there are depletions of local populations such as off the coast of South Africa and Australia based on trends in abundance. Both of these areas are known to have high levels of illegal fishing that take sharks which is contributing to these decreasing trends. There is no information on the population trend in the area that the deep-set fishery operates where the Indo-west Pacific DPS is found, however there is no evidence to suggest that there is a localized depletion in the area since there are no artisanal or international shark fisheries in the action area. The effective population size is estimated to be at least 11,280 adults, therefore one mortality represent 0.009 percent (1/11,280*100=0.00886) of the population. Due to the small level of take we consider the risk to the scalloped hammerhead shark DPS from the proposed action to be negligible.

8 Cumulative Effects

"Cumulative effects", as defined in the ESA implementing regulations, are limited to the effects of future state, tribal, local, or private actions that are reasonably certain to occur in the action area considered in this opinion (50 CFR 402.02). Future federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to Section 7 of the ESA. Because the action area is primarily a swath of the North Pacific Ocean (see Figure 1) and cumulative effects, as defined in the ESA, do not include the continuation of actions described under the Environmental Baseline, few actions within the action area are expected to result in cumulative effects.

Cumulative effects on the eight species addressed by this opinion are likely to occur as a result of climate change, and any increase in fishing from Hawaii state fisheries or international fisheries, ship traffic, and other actions described in the Environmental Baseline section. Such effects could include worsening of the climate change effects described in Sections 5 and 6, and also could result in corresponding increases in fishing gear entanglements and vessel strikes of marine mammals and in fishing gear interactions of the four turtle species. In addition, any increases in marine debris could also increase entanglements or ingestion in all eight species.

Hawaii state and international fisheries that occur in the action area are expected to continue and therefore may impact cetaceans and turtles and their habitat in the future. The future effort level of these fisheries in the action area may vary considerably, although NMFS has no information to suggest that impacts to protected species will materially change from historic levels. .

Global anthropogenic climate change is expected to continue and therefore may impact whales and their habitat in the future. As stated previously, the humpback and sperm whales are cosmopolitan species ranging throughout the world's oceans and thermal and prey limitations related to climate change are unlikely to impact the range of these species (MacLeod 2009). The effects of climate change on the MHI IFKW DPS are not known. Whilst oceanic cetaceans are unlikely to be directly affected by rises in sea levels, important habitats for coastal species and species that require coastal bays and lagoons for breeding, such as humpback whales, could be adversely affected in the future (Simmonds and Elliot 2009). Humpback whales that feed in polar-regions may also encounter reduced prey.

Global anthropogenic climate change is expected to continue and to therefore continue to impact sea turtles and their habitats. Rising temperatures at nesting beaches may continue to exacerbate a female bias in hatchling sex ratio and could also increase embryonic mortality if beaches are already at the high end of thermal tolerance for sea turtle nests (Matsuzawa et al. 2002, Hays et al. 2003, Pike 2013). In addition the number of severe storms is expected to increase with warming ocean temperatures which is expected to change the shape of nesting beaches and to wipe out nests. This has been documented in the Atlantic; comparisons were made between loggerhead and green turtle nesting and cyclone intensity and they found that hatching success declines with increased cyclone intensity (Van Houtan and Bass 2007). Turtles that occur in the action area come from nesting aggregations that may be affected by impacts at their nesting beaches of origin throughout the Pacific. The best available demonstrations of the potential effects of sea level rise indicate that some sea turtle nesting beaches will lose a percentage of their current area by 2100 (Fish et al. 2005, Baker et al. 2006, Fuentes et al. 2009), however these were modeled on static systems and did not account for geomorphological dynamics, such as the natural sinking of islands or the natural growth of coral reefs to keep up with sea level rise. A quantitative analysis of physical changes in 27 atoll islands in the central Pacific over a 19 to 61 year period that corresponds with a rate of sea level rise of 2.0 mm/y shows that 86 percent of islands remained stable (43 percent) or increased in area (43 percent) while only 14 percent of study islands exhibited a net reduction in island area (Webb and Kench 2010, Van Houtan 2010), evidence that changes will not be uniform or predictable and sea level rise may or may not result in nesting beach loss.

Alterations to foraging habitats and prey resources, changes in phenology and reproductive capacity that correlate with fluctuations in SST, and potential changes in migratory pathways and range expansion (all discussed previously in Environmental Baseline) are additional ways in which sea turtles may continue to be impacted by climate change. Many marine species, including the pelagic life stages of sea turtle species in the action area, forage in areas of nutrient rich oceanic upwelling, the strength, location, and predictability of which may change with increasing global temperatures (Harwood 2001).

Recent studies have shown that several sea turtle populations are correlated with climate variability over long periods of time (Van Houtan and Halley 2010, del Monte-Luna et al. 2012). The Pacific Decadal Oscillation (PDO) and the Atlantic Multidecadal Oscillation (AMO) reflect atmospheric circulation patterns that regulate oceanographic processes and ecosystem productivity. The greatest influence appears to occur early on in a hatchling's life, when "climate is the parent," and there is high or low productivity (Van Houtan and Halley 2010). Years of high productivity are correlated later in time (when they reach maturity) with higher levels of nesters appearing at beaches, and low productivity years with the opposite for loggerheads in both the Atlantic and the Pacific (Van Houtan and Halley 2010). Another component of this study is the climate influence on nesting females, where SST temperatures have been shown to influence breeding remigration, as mentioned earlier.

Although there is much speculation about potential impacts of climate change to species and ecosystems, there are multiple layers of uncertainty associated with these analyses making it impossible to accurately predict the most likely scenario that will result and consequently what impacts species and ecosystems will face, particularly in Pacific Island countries (Barnett 2001). Effects of climate change will not be globally uniform (Walther et al. 2002) and information regarding the magnitude of future climate change is speculative and fraught with uncertainties (Nicholls and Mimura 1998). In particular, there is no comprehensive assessment of the potential impacts of climate change within the action area or specific to sea turtles, humpback whales, sperm whales, MHI IFKW, or scalloped hammerhead sharks that may be within the action area.

In addition to the uncertainty of the rate, magnitude, and distribution of future climate change and its associated impacts on temporal and spatial scales, the adaptability of species and ecosystems are also unknown. Impact assessment models that include adaptation often base assumptions on when, how, and to what adaptations occur on theoretical principles, inference from observations, and arbitrary selection, speculation, or hypothesis (see review in Smit et al. 2000). Impacts of climate change and hence its 'seriousness' can be modified by adaptations of various kinds (Tol et al. 1998). Ecological systems evolve in an ongoing fashion in response to stimuli of all kinds, including climatic stimuli (Smit et al. 2000). Sea turtles, whales, and sharks may exhibit a variety of adaptations to cope with climate change-related impacts, although it will likely take decades to centuries for both climate-related impacts and associated adaptations to occur (Limpus 2006) making it increasingly difficult to predict future impacts of climate change on these species in the action area. For example, sea turtles are known to be highly mobile and in the past have shown the ability to adapt to changes in their environment and relocate to more suitable foraging and nesting sites over the course of multiple generations. Implications of climate change at the population level are a key area of uncertainty and one of active research (e.g., Jonzén et al. 2007) and cannot currently be reliably quantified in terms of actual mortalities resulting from climate change impacts over any time scale, nor can they be qualitatively described or predicted in such a way as could be more meaningfully evaluated in the context of this biological opinion.

9 Integration and Synthesis of Effects

The purpose of this biological opinion is to determine if the proposed action is likely to have direct or indirect effects on threatened and endangered species that appreciably reduce their likelihood of surviving and recovering in the wild by reducing their reproduction, numbers, or distribution (50 CFR 402.02), otherwise known as the jeopardy determination. This is done by considering the effects of the action within the context of the ‘Status of Listed Species’ together with the ‘Environmental Baseline’ and the ‘Cumulative Effects’, as described in the Approach section (beginning of Section 7 Effects of the Action). We determine if mortality of individuals of listed species resulting from the proposed action is sufficient to reduce the viability of the populations those individuals represent (measured using changes in the populations’ abundance, reproduction, spatial structure and connectivity, growth rates, or variance in these measures to make inferences about the population’s extinction risks). In order to make that determination, we use a population’s base condition (established in the Status of Listed Species and Environmental Baseline sections of this opinion) as context for the overall effects of the action on affected populations. Finally, our opinion determines if changes in population viability, based on the Effects of the Action and the Cumulative Effects, are likely to be sufficient to reduce viability of the species those populations comprise. The following discussions summarize the probable risks the proposed action poses to the eight listed species addressed by this opinion.

9.1 Humpback Whales

As discussed in the humpback section of the Status of Listed Species (Section 5.2.1), there were an estimated 21,000 humpback whales in the entire North Pacific basin and the population has grown at approximately 4.9 to 6.8 percent per year for the last decade. The CNP stock is at approximately 10,000 individuals (Fleming and Jackson 2011, Allen and Angliss 2013).

As discussed in the humpback section of the Environmental Baseline (Section 6.1), up to seven mortalities are occurring annually from the CNP stock within the action area due to fishery interactions and vessel strikes.

As mentioned in the Environmental Baseline section (Section 6.1), significant threats to this stock include ship strikes and incidental entanglement in commercial fishing gear. NMFS PIRO PRD conducted a negligible impact determination analysis for a permit under the MMPA 101(a)(5)(E) for fisheries that affect the CNP humpback whale stock, including the Hawaii deep-set fishery, that considered the cumulative impact to this stock from all human sources (NMFS 2014e). In this analysis, NMFS found that the 5-year annual average mortality and serious injury (M&SI) to the CNP stock of humpback whales from all human-caused sources, including commercial fisheries (9.35 animals) + recreational fisheries (0.35 animals) + ship strikes (4.25 animals) + marine debris (2.25 animals), is 16.20 animals, which is 26.47 percent of this stock's PBR (61.2 animals/ year). Total human related M&SI is therefore below PBR. It should be emphasized that PBR, which represents the maximum total number of individuals that may be removed from the CNP stock while still allowing such stock to reach its optimum sustainable population, is an objective MMPA stock measure that applies only to the CNP humpback whale stock; the stock found in the action area that interacts with the deep set longline fishery. Nevertheless, consideration of this measure helps inform the jeopardy analysis for the species under ESA.

As described in the Effects of the Action (Section 7.1), if we assume that the proposed action will result in approximately 1,305 trips, with 18,592 sets and 46,117,532 hooks annually, that level of effort is expected to directly result in up to two humpback interactions annually, which could result in up to one (rounded from 0.7) mortality annually, or 0.005 percent annually from the current total North Pacific population, and 0.01 percent annually from the CNP stock. Consistent with approaches used in the stock assessment report and in negligible impact analyses that look at PBR and impacts that occur over a broader period of time, which is the 5-year average, to gauge effects, the removal of 3.5 (0.7 *5) animals over a 5-year average represents 5.7 percent of PBR on an annual basis, well below any level that would be expected to impact the ability of the CNP stock of humpback whales to survive and recover. As detailed above, the North Pacific population has continued to increase at a robust annual rate, indicating that Hawaii longline fisheries impacts are not an impediment to population growth, and the percentage of loss to the overall North Pacific population is expected to continue to decrease. Additionally, the CNP stock is one of three stocks that compose the North Pacific population, which is one of several populations that make up the humpback whale species. Because this contribution to mortality is small relative to an increasing population, and will be a decreasing fraction of what total mortality for the species might be, we do not believe that this small effect, when considered together with the environmental baseline and the cumulative effects, will have a detectable impact on the species' population dynamics.

As discussed in the Cumulative Effects section (Section 8), effects to humpback whales may occur as a result of climate change, and any increase in fishing, ship traffic, and other actions described in the Environmental Baseline section. Such effects could include worsening of the climate change effects described in Sections 5 and 6, as well as an increase in effects resulting from fishing gear interactions. In addition, any increases in marine debris could also increase entanglements. Global climate change is expected to continue and therefore may impact humpback whales and their habitat in the future. As stated previously, the humpback whale is a cosmopolitan species ranging throughout the world's oceans and thermal and prey limitations related to climate change are unlikely to impact the range of this species (MacLeod 2009). Whilst oceanic cetaceans are unlikely to be directly affected by rises in sea levels, important habitats for coastal species and species that require coastal bays and lagoons for breeding, such as humpback whales, could be adversely affected in the future (Simmonds and Elliot 2009). Humpback whales that feed in polar-regions may also encounter reduced prey. As discussed earlier in this opinion, although there is much speculation on the potential impacts of climate change to species and ecosystems, there are multiple layers of uncertainty associated with these analyses and the effects of climate change will not be globally uniform. In particular, there is no comprehensive assessment of the potential impacts of climate change within the action area or specific to humpback whales that may be within the action area. In addition to the uncertainty of the rate, magnitude, and distribution of future climate change impacts on temporal and spatial scales, the adaptability of species and ecosystems are also subject to considerable uncertainty. Implications of climate change at the population level are a key area of uncertainty and one of active research (e.g. Jonzén et al. 2007) and cannot currently be reliably quantified or analyzed in terms of actual mortalities resulting from climate change impacts over any time scale. Because any analysis of climate change impacts would involve considerable speculation, such impacts could not be meaningfully evaluated in the context of this biological opinion.

We considered to what extent the effects of the action affect survival and recovery of the humpback whale. The NMFS and USFWS' ESA Section 7 Handbook (USFWS and NMFS 1998) provides further definitions for *survival* and *recovery*, as they apply to the ESA's jeopardy standard[29].

> *Survival* means: the species' persistence beyond the conditions leading to its endangerment, with sufficient resilience to allow recovery from endangerment. Said another way, survival is the condition in which a species continues to exist into the future while retaining the potential for recovery. This condition is characterized by a species with a sufficiently large population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring, which exists in an environment providing all requirements for completion of the species' entire life cycle, including reproduction, sustenance, and shelter.
>
> *Recovery* means: improvement in the status of a listed species to the point at which listing is no longer appropriate under the criteria set out in section 4(a)(1) of the Act. Said another way, recovery is the process by which species' ecosystems are restored and/or threats to the species are removed so self-sustaining and self-regulating populations of listed species can be supported as persistent members of native biotic communities.

The NMFS (1991) humpback whale recovery plan contains a number of goals and criteria that should be met to achieve recovery. 1) To increase humpback whale population to at least 60 percent of the number existing before commercial exploitation or of current environmental carrying capacity; acceptable evidence of ongoing recovery will be statistically significant trends of population increase; 2) statistically significant trends of population increases in portions of the range known to have been occupied in historical times; 3) maintain and enhance habitat; and 4) reduce human-related mortality, injury and disturbance. The proposed action is not expected to

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[29] "Jeopardize the continued existence of" means "to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (50 CFR 402.02).

have a significant effect on the completion of these goals. The population is expected to continue to increase, and the proposed action will not hinder any efforts to maintain and enhance habitat.

As described above the proposed action's direct effects on North Pacific humpback whales, when considered together with the environmental baseline and cumulative effects, are not reasonably expected to cause an appreciable reduction in the likelihood of survival or recovery of the species. We expect the overall population to continue to grow and to maintain genetic heterogeneity, broad demographic representation, and to successfully reproduce. The proposed action will have an extremely small effect on the overall size of the population. We do not expect the proposed action to negatively affect the humpbacks' ability to meet their lifecycle requirements, or its recovery.

The ESA allows for incidental take of species resulting from otherwise lawful activities, provided that such take is not likely to result in jeopardy, and further that any such take is carried out consistent with those terms and conditions and reasonable and prudent measures that NMFS determines are necessary or appropriate to minimize the impact of take. In addition, section 101(a)(5)(E) of the MMPA, 16 U.S.C. 1361 et seq., has provisions for NMFS to issue permits for the taking of marine mammals designated as depleted because of their listing under the ESA, 16 U.S.C. 1531 et seq., by U.S. vessels. Under section 101(a)(5)(E), such a permit may be issued only after several factors are met, including an analysis has been conducted to determine if the incidental mortality and serious injury from commercial fisheries will have a negligible impact on the affected species or stock. A negligible impact determination was conducted simultaneously with this opinion for the Central North Pacific stock and NMFS determined that the annual mortality and serious injury incidental to commercial fisheries in Hawaii and Alaska (9.35) will have a negligible impact for purposes of issuing a permit under section 101(a)(5)(E) of the MMPA because, among other factors, total fisheries-related serious injuries and mortalities are greater than 10 percent of PBR and less than 100 percent of PBR, and the population is increasing.

While the proposed action would result in some incidental take of this species by the deep-set fishery, the impact of that take would be reduced to the extent possible. These takes are rare and based on descriptions from observers of past interactions from both the deep-set and shallow-set fisheries, the majority of interactions were determined to be non-lethal because the animals were either released with all gear removed or they were able to break free from the gear without noticeable impairment of the animal's ability to swim or feed. In one case the interaction was determined to be serious because the gear was not removed and was considered more likely than not to lead to mortality. Based on these past interactions, the majority of humpback whale takes from the proposed action are expected to be non-lethal, which are not expected to have any measurable impact on their numbers, reproduction, or distribution. The anticipated non-lethal takes are not expected to impact the reproductive potential, fitness, or growth of any of the incidentally caught humpback whales since they will be released unharmed shortly after capture, or released with only minor injuries from which they are expected to recover.

The incidental lethal and non-lethal takes of humpback whales associated with the proposed action are not reasonably expected to cause an appreciable reduction in the likelihood of survival and recovery of the species. Although any level of take and mortality can have an adverse effect on the population, we find that the expected level of take from the overall action, including a small number of mortalities, is extremely small when considered together with all impacts considered in the Status of the Species, Environmental Baseline and Cumulative Effects sections, including other federally authorized fisheries and foreign fisheries. As stated previously, the proposed action is expected to result in the mortality of up to one humpback annually and the affected population is increasing. Based on the best available information, we believe that the population will remain large and continue to grow to contribute to species recovery. Moreover, we do not believe that the proposed action is reasonably likely to result in an appreciable reduction in the likelihood of recovery of the species. The proposed action does not impede progress on carrying out any aspect of the recovery program or achieving recovery goals.

9.2 Sperm Whales

As discussed in the sperm whale section of the Status of Listed Species (Section 5.2.2), current sperm whale abundance is estimated to be approximately 200,000-1,500,000 worldwide (NMFS 2010c). The recent 2010 shipboard line-transect survey of the EEZ around the Hawaiian Islands resulted in an abundance estimate of 3,354 (CV = 0.34) for the Hawaii stock of sperm whales (Bradford et al 2013), including a correction factor for missed diving animals. This is currently the best available abundance estimate for this stock, which is the part of the population that the deep-set fishery interacts with directly.

As discussed in the Environmental Baseline section (Section 6.2), primary threats to the species are fishery interactions and vessel strikes. As discussed in this section, up to one mortality annually from the Hawaii stock is occurring within the action area due to fishery interactions.

NMFS PIRO PRD conducted a negligible impact determination analysis for a permit under the MMPA 101(a)(5)(E) for fisheries that affect the Hawaii sperm whale stock, including the Hawaii deep-set fishery, that considered the cumulative impact to this stock from all human sources (see NMFS 2014e). In this analysis, NMFS found that the 5-year annual average mortality and serious injury (M&SI) to the Hawaii stock of sperm whales from all human-caused sources, including commercial fisheries (0.7 animals) + recreational fisheries (0 animals) + ship strikes (0 Animals) + marine debris (0 animals), is 0.7 animals, which is 6.86 percent of this stock’s PBR (10.2 animals/ year). As noted above, PBR represents the maximum total number of individuals that may be annually removed from a marine mammal stock while still allowing such stock to reach its optimum sustainable population. Although PBR is an objective population measure for analyzing a stock, it is a useful quantitative measure to help inform the jeopardy analysis for the species conducted under ESA. In this case, total human related M&SI for the Hawaii stock of sperm whales is less than ten percent of PBR.

As described in the Effects of the Action (Section 7.2), if we assume that the proposed action will result in approximately 1,305 trips, with 18,592 sets, and 46,117,532 hooks annually, that level of effort could result in up to 3 (rounded from 2.8) sperm whale interactions annually, which could result in up to 2 (rounded from 1.4) mortalities annually from the Hawaii stock. In any given year, the loss of two represents about 19 percent (2÷10.2) of PBR for this stock. Consistent with the approaches typically used in the SAR and in the negligible impact determination analyses to compare known serious injuries/mortality to PBR and impacts that occur over a broader period of time to gauge effects, the loss of 7 (1.4/year * 5years) sperm whales over a 5-year average represents 69 percent (7/10.2 = 68.6) of PBR. less than the PBR threshold for removals that might have a significant delay on recovery for the Hawaii stock of sperm whales. Additionally, the Hawaii stock is one of several populations that make up the sperm whale species. Two mortalities per year is 0.06 (2/3,354*100=0.06) percent of the estimated current population of the Hawaii sperm whale stock and 0.005 percent of the estimated current total Pacific population on an annual basis. Because this contribution to mortality is small we do not believe that the small effect posed by lethal takes in this fishery, when considered together with the environmental baseline and the cumulative effects, is likely to appreciably reduce survival and recovery of the species.

As discussed in the Cumulative Effects section (Section 8), effects to sperm whales may occur as a result of worsening climate change, and any increase in fishing, ship traffic, and other actions described in the Environmental Baseline section. Such effects could include worsening of the climate change effects described in Sections 5 and 6, as well as an increase in effects resulting from fishing gear interactions. In addition, any increases in marine debris could also increase entanglements. Global climate change is expected to continue and therefore may impact sperm whales in the future. As stated previously, the sperm whale is a cosmopolitan species ranging throughout the world's oceans and thermal and prey limitations related to climate change are unlikely to impact the range of this species (MacLeod 2009). As discussed earlier in this opinion, although there is much speculation on the potential impacts of climate change to species and ecosystems, there are multiple layers of uncertainty associated with these analyses and the effects of climate change will not be globally uniform. In particular, there is no comprehensive assessment of the potential impacts of climate change within the action area or specific to sperm whales that may be within the action area. In addition to the uncertainty of the rate, magnitude, and distribution of future climate change impacts on temporal and spatial scales, the adaptability of species and ecosystems are also subject to considerable uncertainty. Implications of climate change at the population level are a key area of uncertainty and one of active research (e.g. Jonzén et al. 2007) and cannot currently be reliably determined in terms of actual mortalities resulting from climate change impacts over any time scale. Nor can they be qualitatively described or predicted in such a way as could be more meaningfully evaluated in the context of this biological opinion.

> We considered to what extent the effects of the action affect survival and recovery of the sperm whale. The NMFS (2010) sperm whale recovery plan contains a number of goals and criteria that should be met to achieve recovery: 1) To increase the populations in each ocean basin in which it occurs (Atlantic Ocean/Mediterranean Sea, Pacific Ocean, and Indian Ocean) and the global population has at least 1,500 mature reproductive individuals (consisting of at least 250 mature females and at least 250 mature males); 2) reduce threats to the species' habitat or range; 3) reduce threats from overutilization for commercial, recreational or educational purposes; C) reduce the threat of disease or predation; D) and reduce the inadequacy of existing regulatory mechanisms to. The proposed action is not expected to have a significant effect on the completion of these goals.

We conclude that the incidental take and resulting mortality of sperm whales associated with the proposed action are not reasonably expected to cause an appreciable reduction in the likelihood of survival or recovery of the species. We expect the overall population to continue to grow and to maintain genetic heterogeneity, broad demographic representation, and to successfully reproduce. The proposed action will have a small effect on the overall size of the population, and we do not expect it to affect the sperm whales' ability to meet their lifecycle requirements and to retain the potential for recovery.

The ESA allows for incidental take of species resulting from otherwise lawful activities (such as the proposed action), provided that such take does not result in jeopardy, and the impact of such take is minimized to the extent practicable. In addition, section 101(a)(5)(E) of the MMPA, 16 U.S.C. 1361 et seq., has provisions for NMFS to issue permits for the taking of marine mammals designated as depleted because of their listing under the ESA, 16 U.S.C. 1531 et seq., by U.S. vessels, only after an analysis has been conducted to determine if the incidental mortality and serious injury from commercial fisheries will have a negligible impact on the affected species or stock. This analysis was conducted simultaneously with this analysis for the Hawaii stock and NMFS determined that the current annual mortality and serious injury incidental to commercial fisheries in Hawaii (0.7) will have a negligible impact for purposes of issuing a permit under section 101(a)(5)(E) of the MMPA.

While the proposed action would result in some incidental take of this species by the deep-set fishery, take would be subject to existing mitigation measures to reduce its impact. These takes are rare and based on descriptions from observers of past interactions from both the deep-set and shallow-set fisheries, the majority of interactions were determined to be non-lethal because they were either released with all gear removed or they were able to break free from the gear without noticeable impairment of the animal's ability to swim or feed. In one case the interaction was determined to be serious because the gear was not removed and could potentially lead to mortality. Based on these past interactions, the majority of sperm whale takes from the proposed action are expected to be non-lethal, which are not expected to have any measurable impact on their numbers, reproduction, or distribution. The anticipated non-lethal takes are not expected to impact the reproductive potential, fitness, or growth of any of the incidentally caught sperm whales since they will be released unharmed shortly after capture, or released with only minor injuries from which they are expected to recover.

We believe that the incidental lethal and non-lethal takes of sperm whales associated with the proposed action are not reasonably expected to cause an appreciable reduction in the likelihood of survival of the species. Although any level of take and mortality can have an adverse effect on the overlying population, we find that the expected level of take from the overall action, including a small number of mortalities, is extremely small when considered together with all impacts considered in the Status of the Species, Environmental Baseline and Cumulative Effects sections, including other federally authorized fisheries and foreign fisheries. As stated previously, the proposed action could result in the mortality of seven sperm whales every five years. We believe the population will remain large and continue to grow to contribute to species recovery. Moreover, we do not believe that the proposed action is reasonably likely to result in an appreciable reduction in the likelihood of recovery of the species. The proposed action does not appreciably impede progress on carrying out any aspect of the recovery program or achieving the overall recovery strategy.

9.3 MHI Insular False Killer Whales

As discussed in the MHI IFKW section of the Status of Listed Species (Section 5.2), the best estimate of abundance is 151 animals for this DPS (Carretta et al. 2014). The MHI IFKW is believed to have declined markedly during the 1990s, although the current population trajectory is unknown (Oleson et al. 2010).

As discussed in the MHI IFKW section of the Environmental Baseline (Section 6.3), up to one mortality every four years from the DPS is occurring within the action area due to fishery interactions with domestic fleets.

As described in the Effects of the Action (Section 7.3), if we assume that the proposed action will result in 18,592 sets with 46,117,532 hooks annually, that level of effort will result in less than one (0.2696) MHI IFKW interaction annually or one every three years; this is less than one (0.246) mortality, which will result in one mortality every four years from the MHI IFKW DPS. The anticipated mortality over four years represents 0.2 percent of the estimated current total population of the MHI IFKW DPS. In Section 7 we introduced a tool of the MMPA to help provide an additional measure to assess effects to marine mammal populations. This additional tool helps inform our analysis when looking at the impacts on the recovery of the species. To help analyze the potential impacts of the proposed action, we considered the PBR for this DPS, which is the maximum number of animals, not including natural mortalities, that may be removed from the MHI IFKW DPS while allowing it to reach or maintain its optimum sustainable population. While not an ESA construct, PBR does provide us with an objective measure of the proposed action’s effects and therefore is useful to, and informs, our analysis.

In the most recent SAR, the potential annual PBR level for this stock is calculated to be 0.3 (0.258 unrounded) MHI IFKWs per year (Carretta et al. 2014). In any given year, the loss of one MHI IFKW would represent about 333 percent (1÷ 0.3) of PBR for this stock, which is greater than the estimated value that would delay the time in which the stock will achieve its optimum sustainable population by more than 10 percent (not to be confused with reducing the likelihood of recovery under the ESA jeopardy standard). However, consistent with the approaches typically used in the SAR and in the negligible impact determination analyses to compare known serious injuries/mortality to PBR and impacts that occur over a broader period of time to gauge effects, the loss of 1.2 (0.246/year * 5 years), NMFS estimates actual M&SI to be .246 moralities annually. Over a 5-year period, this level of M&SI represents 95 percent (.246/.258 = 0.95) of PBR on an annual basis, less than the PBR threshold for removals that might have a significant delay on recovery for the MHI IFKW DPS. This analysis suggests that the effects of the action are not strong enough to significantly affect recovery of the MHI IFKW.

Because this contribution to mortality is small, and because the fishery is currently managed under a TRP and implementing regulations that are intended to reduce and eliminate impacts on both MHI insular and pelagic false killer whales, and provide NMFS the ability to monitor and promptly respond to any unanticipated fishery impacts, we do not believe that the effects posed by lethal takes in this fishery, when considered together with the environmental baseline and the cumulative effects, will appreciably reduce the likelihood of survival or recovery of the species.

As discussed in the Cumulative Effects section (Section 8), effects to MHI IFKW may occur as a result of worsening climate change, and any increase in fishing, ship traffic, and other actions described in the Environmental Baseline section. Such effects could include worsening of the climate change effects described in Sections 5 and 6, as well as an increase in effects resulting from fishing gear interactions. In addition, any increases in marine debris could also increase entanglements. Global climate change is expected to continue and therefore may impact MHI IFKW and their habitat in the future. As discussed earlier in this opinion, although there is much speculation on the potential impacts of climate change to species and ecosystems, there are multiple layers of uncertainty associated with these analyses and the effects of climate change will not be globally uniform. In particular, there is no comprehensive assessment of the potential impacts of climate change within the action area or specific to MHI IFKW that may be within the action area. In addition to the uncertainty of the rate, magnitude, and distribution of future climate change and its associated impacts on temporal and spatial scales, the adaptability of species and ecosystems are also unknown. Implications of climate change at the population level are a key area of uncertainty and one of active research (e.g., Jonzén et al. 2007) and cannot currently be reliably quantified in terms of actual mortalities resulting from climate change impacts over any time scale. Nor can they be qualitatively described or predicted in such a way as could be more meaningfully evaluated in the context of this biological opinion.

Section 101(a)(5)(E) of the MMPA, has provisions for NMFS to issue permits for the taking of marine mammals designated as depleted because of their listing under the ESA, by U.S. vessels, only after an analysis has been conducted to determine if the incidental mortality and serious injury from commercial fisheries will have a negligible impact on the affected species or stock. This analysis was conducted in 2014 for the MHI IFKW DPS. NMFS determined that the annual mortality and serious injury incidental to commercial fisheries in Hawaii (0.1) will have a negligible impact for purposes of issuing a permit under section 101(a)(5)(E) of the MMPA because total fisheries-related M&SI are greater than 10 percent PBR and less than 100 percent PBR (NMFS 2014e). The estimated future M&SI also falls into the range that is considered negligible for this DPS. The MHI IFKW meets the initial conditions of Criterion 3. Total fisheries-related M&SI (0.1 animals per year) is greater than 10 percent PBR (0.03 animals) and less than PBR (0.3 animals) for the 2007-2011 time frame. Although Hawaiian insular false killer whales are believed to have declined markedly during the 1990s, their current population trajectory is unknown (Oleson et al. 2010). Accordingly, the population is not known to be stable or increasing under NID Criterion 3. However, it is expected that the longline fishery impacts, which historically were a high risk threat to this population, have been and will be further reduced through the recently implemented FKWTRP regulatory measures, which will be monitored for effectiveness (Carretta et al. 2014, NMFS 2012d).

NMFS published the FKWTRP on November 29, 2012 (77 FR 71260) to reduce the M&SI of Hawaii pelagic and MHI insular false killer whales in Hawaii’s longline fisheries. Most of the Plan’s regulations went into effect on December 31, 2012, but gear requirements for the deep-set longline fishery went into effect on February 27, 2013. The most significant measure in the plan for the protection of the MHI IFKW is a greater prohibited area closure for the longline fisheries. The longline prohibited area maintains the previous February-September boundary year-round. NMFS removed the seasonal boundary change from October 1 through January 31 that allowed longline fishing closer to the MHI. Longline fishing is prohibited within the entire core range of the MHI IFKW DPS, and, since the implementation of the FKWTRP, year-round within over 74 percent of the MHI insular/pelagic overlap zone. From 2002-2012, before the implementation of the FKWTRP and the change to the MHI Longline Fishing Prohibited Area, eight of nine observed false killer whale or unidentified blackfish interactions occurred within the seasonally-open area close to the MHI (open October through January) (Figure 4). Longline fishing is now prohibited within that area year-round (50 CFR 229.37(d)(1) and 50 CFR 665.806(a)(2)), per the requirements of the FKWTRP. The remaining unidentified blackfish interaction occurred 0.84 nm outside the area now closed to longline fishing. Recent telemetry data on a greater proportion of the MHI IFKW also shows that MHI IFKW are not traveling much beyond the core area on the windward sides of the islands (Oleson et al 2014), making it less likely that they will interact with the longline fisheries now that they operate even further offshore year round. Due to these FKWTRP changes, and telemetry data, NMFS anticipates that the future number of interactions and mortalities for the MHI IFKW is a conservative and precautionary estimate and will likely be lower.

While the proposed action would result in some incidental take of this species by the U.S. fishery, take would be subject to mitigation measures to reduce its impact. These takes are rare. The anticipated non-lethal takes are not expected to impact the reproductive potential, fitness, or growth of any of the incidentally caught MHI IFKWs since they will be released unharmed shortly after capture, or released with only minor injuries from which they are expected to recover.

We conclude that the incidental take and resulting mortality of MHI IFKW associated with the proposed action are not reasonably expected to cause an appreciable reduction in the likelihood of survival or recovery of the species. The longline fishery’s impacts are extremely small and are expected to result in one MHI IFKW mortality every four years. We do not expect that the removal of one MHI IFKW every four years will interfere with the stock’s ability to achieve its OSP. We recognize that other human-caused sources of mortality may occur that have not been quantified, such as from pollutants and state-managed fisheries. Nevertheless, under the TRP regulations, we expect that any unanticipated fishery impacts that might indicate a worsening trend for the MHI IFKW DPS will be detected and promptly managed in consultation with the FKW TRT. As a term and condition of the incidental take statement in this opinion, NMFS will require that the False Killer Whale TRT be convened following a second observed take, within a three-year period, of a FKW within the overlap zone to develop recommendations for possible emergency measures to further protect MHI IFKW’s. Accordingly, NMFS is satisfied that under the adaptive management process developed by the TRT and implemented by NMFS, the proposed action will have a small effect on the overall size of the population, and we do not expect it to affect the MHI IFKW ability to meet their lifecycle requirements and to retain the potential for recovery.

Although any level of take and mortality can have an adverse effect on the overlying population, we find that the expected level of take from the overall action, including a small number of mortalities, is extremely small when considered together with all impacts considered in the Status of the Species, Environmental Baseline and Cumulative Effects sections.

9.4 Loggerhead Turtles

As discussed in the loggerhead section of the Status of Listed Species (Section 5.2.4), nesting of North Pacific loggerheads in Japan has been variable over time, but has steadily increased from 1999 present (NMFS 2011b, STAJ 2012). In 2013 the number of nests laid was the highest on record, with 15,396 nests laid (excluding Yakushima Island which has typically represented 40 percent of nesting activity) (Y. Matsuzawa pers comm.) since comprehensive counts were started in the 1980s. While nesting trends do not necessarily reflect overall population status (NRC 2010), the nesting trend data from Japan are currently the best available information on the status of the North Pacific DPS. The increase from approximately 2,000 nests (representing approximately 500 nesting females) in 1999, to 6,500 – 14,632 nests (representing approximately 3,600 nesting females) in 2012 demonstrates that the population trend may not directly correlate with fishing mortality rates because large numbers of turtles were caught before, during, and after this time period as described in section 6. Surveys in Japan indicated a greater than three-fold (linear scale) increase from 2007–2008. While a purely demographic model could not reproduce this trend, the climate-based model that combines the PDO and winter SST, captures this dramatic increase emphasizing the potential importance (forcing) of climate on this population (Van Houtan and Halley 2011, Van Houtan 2011).

As discussed in the loggerhead section of the Environmental Baseline (Section 6.4), 50-600 juvenile and adult North Pacific loggerhead mortalities may be occurring annually due to longline fishery interactions from all vessels within the action area Thus, total fishery-related mortality of the North Pacific loggerhead population due to longline fishing, nearshore fishing in Japan, and other fisheries, is likely over several hundred annually.

As described in the loggerhead section of the Effects of the Action (Section 7.4), our analysis assumes that the proposed action will result in up to approximately 1,305 trips, with 18,592 sets and 46,117,532 hooks annually. That level of effort is expected to result in three loggerhead interactions annually, and a maximum of three mortalities annually (representing 0.18 adult female) from the North Pacific DPS. This represents one female mortality every 5.4 years or 0.003 percent of breeding females in the North Pacific DPS. The risk posed by this level of mortality to the North Pacific loggerhead DPS was assessed by Van Houtan (2013, 2014) for application to this opinion. Quantitative population viability analysis (PVA) modeling works with discrete numbers; whole numbers and not partial individuals. Rounding the calculated adult female mortality (ANE) estimate up to a single individual would overestimate the fishery impacts by a factor of 5, and therefore it was determined that the no further modeling was possible due to the extremely small impacts. NMFS has determined that the loss of one adult female every 5.4 years as a result of the proposed action will have a negligible effect on the North Pacific DPS (Van Houtan 2013, 2014).

As discussed in the Cumulative Effects section (Section 8), effects to this DPS are likely to occur as a result of worsening climate change, and any increase in fishing, marine debris, and other actions described in the Environmental Baseline section. Such effects could include worsening of the climate change effects described in Sections 5 and 6, as well as an increase effects resulting from fishing gear interactions with this DPS. In addition, any increases in marine debris could also increase entanglements. Global climate change is expected to continue and therefore may impact sea turtles and their habitats in the future. As discussed in this opinion, rising temperatures at nesting beaches may have negative consequences for incubating nests. While loggerhead nesting does not take place inside the action area, turtles that occur in the action area come from nesting aggregations that may be affected by impacts at their nesting beaches of origin throughout the Pacific, although changes will likely not be uniform or predictable. As also discussed in the Cumulative Effects section of this opinion, climate change may impact aquatic aspects of sea turtle biology and ecology, including foraging habitats and prey resources, phenology, and migration. As discussed earlier in this opinion, although there is much speculation about potential impacts of anthropogenic induced climate change to species and ecosystems, there are multiple layers of uncertainty associated with these analyses and the effects of climate change will not be globally uniform. In particular, there is no comprehensive assessment of potential impacts of anthropogenic climate change within the action area or specific to sea turtles that may be within the action area. In addition to uncertainty of the rate, magnitude, and distribution of future climate change and its associated impacts on temporal and spatial scales, the adaptability of species and ecosystems are also unknown. Implications of climate change at the population level are a key area of uncertainty and one of active research and cannot currently be reliably quantified in terms of actual mortalities resulting from climate change impacts over any time scale. Nor can they be qualitatively described or predicted in such a way as could be more meaningfully evaluated in the context of this biological opinion. Within the temporal scale of the proposed action, any future synergistic impacts of climate change in the action area that might interact with the effects of the proposed action are not considered significant. Viewed within the context of the Status of the Species, the Environmental Baseline, and the Cumulative Effects, the annual loss of the equivalent of 0.18 of an adult female, or one adult female every 5.4 years due to the proposed action (Section 7.4) is not expected to adversely affect population dynamics of the North Pacific loggerhead DPS.

We considered to what extent the effects of the action affect survival and recovery of the North Pacific loggerhead DPS sea turtles. The NMFS and USFWS' ESA Section 7 Handbook (USFWS and NMFS 1998) provides further definitions for *survival* and *recovery*, as they apply to the ESA's jeopardy standard (please refer to the humpback whale discussion in this section for definitions).

The NMFS and USFWS (1998c) 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.

Adult female nesting population size for the North Pacific DPS is conservatively estimated at approximately 6,673 (Van Houtan 2013). As discussed above, the anticipated deaths resulting from the continued authorization of the deep-set fishery results in the removal of approximately 0.18 female annually, or 0.003 percent of breeding females in the North Pacific loggerhead population (less than 1 in 36,100) (Van Houtan 2013, 2014). Because this contribution to mortality is an insignificant fraction of what total mortality for the species might be, we do not believe that the small effect posed by the lethal takes in this fishery, when considered together with the environmental baseline and the cumulative effects, will be detectable or appreciable.

We conclude that the incidental take and resulting mortality of North Pacific loggerhead turtles associated with the direct effects of the proposed action are not reasonably expected to cause an appreciable reduction in the likelihood of survival or recovery of the DPS. 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 loggerheads' ability to meet their lifecycle requirements and to retain the potential for recovery.

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 one 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.

The ESA allows for incidental take of species resulting from otherwise lawful activities (such as the proposed action), provided that such take does not result in jeopardy, and the impact of such take is minimized to the extent practicable. While the direct effects of the proposed action would result in some incidental take of this DPS by the U.S. fishery, take would be subject to existing mitigation measures to reduce its impact. We have applied the post-release mortality criteria conservatively to ensure that sea turtles that are likely to be seriously injured by capture in the fisheries are counted as lethal takes. The anticipated non-lethal takes are not expected to impact the reproductive potential, fitness, or growth of any of the incidentally caught sea turtles because they will be released unharmed shortly after capture, or released with only minor injuries from which they are expected to recover.

The proposed action will result in the mortality of less than one nesting female annually, as discussed above, this level of mortality would present negligible additional risk to the North Pacific DPS. Since it represents a negligible risk to the DPS, the proposed action would not prohibit the DPS from stabilizing or increasing, nor would it prohibit the DPS from reaching a biologically reasonable FENA based on the goal of maintaining a stable population in perpetuity.

The negligible risk to the DPS nesting population, which is the source of animals found at foraging grounds, means it would not substantially impair or prohibit increases to DPS foraging populations at key foraging grounds. The effects of the action would not prohibit or substantially impair continuing efforts to reduce mortality in commercial fisheries. Additionally, there would be no negative indirect effects to nesting females from the proposed action.

We believe that the incidental lethal and non-lethal takes of loggerhead turtles associated with the proposed action are not reasonably expected to cause an appreciable reduction in the likelihood of survival of the North Pacific DPS. Although any level of take and mortality can have an adverse effect on the overlying population, we find that the expected level of take from the action, including a small number of mortalities, is extremely small when considered together with all impacts considered in the Status of the Species, Baseline and Cumulative Effects sections, including other federally authorized fisheries and foreign fisheries. As stated previously, the proposed action is expected to result in the annual mortality of only 0.18 of an adult female equivalent. Moreover, we do not believe that the proposed action is reasonably likely to result in an appreciable reduction in the likelihood of recovery of the North Pacific DPS. The proposed action does not appreciably impede progress on carrying out any aspect of the recovery program or achieving the overall recovery strategy.

To summarize, when considering the effects of the proposed action, together with the status of the listed species, the environmental baseline, and the cumulative effects, we believe that the lethal and non-lethal takes of loggerhead sea turtles associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of both the survival and recovery of the North Pacific loggerhead DPS in the wild.

9.5 Leatherback Turtles

As discussed in the leatherback section of the Status of Listed Species (Section 5.2.5), the western Pacific leatherback population harbors the last remaining nesting aggregation of significant size in the Pacific. The total number of adult females in the Bird’s Head component of the western Pacific population was estimated at 1,949 Van Houtan (2013), which is estimated to make up 75 percent of the population (Dutton et al. 2007), giving a total number of adult females in the western Pacific population of 1,949/0.75 =2599. As discussed in the leatherback section of the Environmental Baseline (Section 6.5), 83 to 120 juvenile and adult western Pacific leatherback mortalities may be occurring annually due to interactions with foreign and domestic longline fisheries within the action area alone. Thus, total fishery-related mortality of the western Pacific leatherback population is likely at least a few hundred adults annually.

As described in the leatherback section of the Effects of the Action (Section 7.5), if we assume that the proposed action will result in up to 18,592 sets with 46,117,532 hooks annually, then that level of effort will result in 24 leatherback interactions annually, and a maximum of nine, mostly juvenile mortalities annually (representing 0.2 adult females) from the Pacific populations. The loss of 9 mostly juvenile leatherbacks is equivalent to the loss of 0.2 adult females (the adult female mortality equivalent), or one female mortality every 4.9 years (or 0.010 percent of breeding females in the Western Pacific population). The risk posed by this level of mortality to the Bird’s Head region of the western Pacific leatherback population was assessed by Van Houtan (2013, 2014) for application to this opinion. A quantitative population viability analysis

(PVA) could not be reliably applied to this scenario because of the low numbers of adult female equivalents. Modeling works with discrete numbers; whole numbers and not partial individuals. Moreover, NMFS considered rounding the calculated adult female mortality (ANE) estimate up to a single individual but determined this was not reliable because it would overestimate the fishery's impacts by a factor of 5. Therefore, given the fishery's extremely low impacts on ANE, it was determined that no further PVA modeling was possible (Van Houtan 2013, 2014). NMFS concludes that the removal of 0.2 ANE annually will have a negligible impact on the leatherback survival and recovery (Van Houtan 2013, 2014).

The Bird's Head component of the population represents about 75 percent of the entire western Pacific population, and it accounts for the majority of leatherback interactions with the Hawaii deep-set fishery. The remaining 25 percent of the population has less risk of interacting with the fishery because the majority of nesting occurs in the winter, which means these animals have different migratory routes that don't enter the proposed action area. Given that the majority of interactions resulting from the proposed action are with the Bird's Head component, the risk to it is the best measure of the risk of the proposed action to the overall population. Considering that the nesting trend for the remainder of the population is fairly stable from the data we have, and that the maximum impact would be less than 0.2 adult female mortality annually, we do not anticipate that this amount would appreciably reduce the likelihood of survival of the non Bird's Head component of the population. We believe that the proposed action will have a negligible impact on the risk to the Bird's Head component, the non-Bird's Head component, and therefore the western Pacific leatherback population as whole. Since there has been one confirmed interaction with a juvenile leatherback in the deep-set fishery, which occurred in 1995 we assume that 4 percent (Table 2) of the leatherback mortalities could be from the eastern Pacific leatherback which has declined and only a small population remains in the region. Due to the migration routes of this population and the low numbers of turtles remaining it is unlikely that there will be future interactions with this population. However to be precautionary since there has been one known interaction in the past (20 years ago) we assume that 4 percent of the adult female mortalities could come from this population. That percentage would equal 0.008 (0.2 *4 percent) adult female mortalities, or one (1/0.008) in 125 years. That level of mortality is negligible and does not pose any further risk to the population.

As discussed in the Cumulative Effects section (Section 8), effects to leatherback sea turtles are likely to occur as a result of worsening climate change, and any increase in fishing, marine debris, and other actions described in the Environmental Baseline section. Such effects could include worsening of the climate change effects described in Sections 5 and 6, as well as an increase in effects resulting from fishing gear interactions. In addition, any increases in marine debris could also increase entanglements. Global climate change is expected to continue and therefore may impact sea turtles and their habitat in the future. As discussed in this opinion, rising temperatures at nesting beaches may have negative consequences for nesting females and developing embryos. While leatherback nesting does not take place inside the action area, turtles that occur in the action area come from nesting aggregations that may be affected by impacts at their nesting beaches of origin throughout the Pacific, although changes will likely not be uniform or predictable. As also discussed in the Cumulative Effects section of this opinion, climate change may impact aquatic aspects of sea turtle biology and ecology, including foraging habitats and prey resources, phenology, and migration.

As discussed earlier in this opinion, although there is much speculation on the potential impacts of climate change to species and ecosystems, there are multiple layers of uncertainty associated with these analyses and the effects of climate change will not be globally uniform. In particular, there is no comprehensive assessment of the potential impacts of climate change within the action area or specific to sea turtles that may be within the action area. In addition to the uncertainty of the rate, magnitude, and distribution of future climate change and its associated impacts on temporal and spatial scales, the adaptability of species and ecosystems are also unknown. Implications of climate change at the population level are a key area of uncertainty and one of active research (e.g., Jonzén et al. 2007, Saba et al. 2012) and cannot currently be reliably quantified in terms of actual mortalities resulting from climate change impacts over any time scale. Nor can they be qualitatively described or predicted in such a way as could be more meaningfully evaluated in the context of this biological opinion.

We considered to what extent the effects of the action affect survival and recovery of the leatherback sea turtle. The NMFS and USFWS' ESA Section 7 Handbook (USFWS and NMFS 1998) provides further definition for *survival* and *recovery*, as they apply to the ESA's jeopardy standard (please refer to the humpback whale discussion in this section for definitions).

The NMFS and USFWS (1998b) 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 (FENA) 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.

Adult female nesting population size for the Bird's head component of the western Pacific population is estimated at approximately 1949 (Van Houtan 2013). This represent approximately 75 percent of the nesting females in the western Pacific, therefore there are approximately 2598 nesting females in the entire western Pacific population. As discussed above, the anticipated deaths resulting from the continued authorization of the deep-set fishery results in the removal of approximately nine turtles annually, or 0.2 adult female nester mortalities, or 0.008 percent of the population of western Pacific nesters. Additionally, the western Pacific population is one of several populations that make up the leatherback species. Because this contribution to mortality is small fraction of what total mortality for the species might be, we do not believe that the small effect posed by the lethal takes in this fishery, when considered together with the environmental baseline and the cumulative effects, will be detectable or appreciable.

We conclude that the incidental take and resulting mortality of western and eastern Pacific leatherback sea turtles associated with the direct effects of the proposed action are not reasonably expected to cause an appreciable reduction in the likelihood of survival or recovery of the species. We expect the overall population to grow and to maintain genetic heterogeneity, broad demographic representation, and successfully reproduce. The proposed action will have a small effect on the overall size of the populations and therefore the species, and we do not expect it to affect the leatherbacks’ ability to meet their lifecycle requirements and to retain the potential for recovery.

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 one 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 (FENA) 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; 5) reduce incidental mortality in commercial, recreational fisheries.

The ESA allows for incidental take of species resulting from otherwise lawful activities (such as the proposed action), provided that such take does not result in jeopardy, and the impact of such take is minimized to the extent practicable. While the direct effects of the proposed action would result in some incidental take of this species by the U.S. fishery, take would be subject to mitigation measures to reduce its impact. The majority of the leatherback sea turtle takes from the proposed action are expected to be non-lethal, which are not expected to have any measurable impact on their numbers, reproduction, or distribution. We have applied the post-release mortality criteria conservatively to ensure that sea turtles that are likely to be seriously injured by capture in the fisheries are counted as lethal takes. The anticipated non-lethal takes are not expected to impact the reproductive potential, fitness, or growth of any of the incidentally caught sea turtles because they will be released unharmed shortly after capture, or released with only minor injuries from which they are expected to recover.

The proposed action is anticipated to result in the mortality of up to one nesting female every 4.9 years, as discussed above, this level of mortality would present negligible additional risk to the leatherback sea turtle. Since it represents a negligible risk to the species, the proposed action would not prohibit the species nesting populations from increasing as predicted, nor would it prohibit the species from reaching a biologically reasonable FENA based on the goal of maintaining a stable population in perpetuity. The negligible risk to the species nesting population, which is the source of animals found at foraging grounds, means it would not substantially impair or prohibit increases to leatherback foraging populations at key foraging grounds. The effects of the action would not prohibit or substantially impair continuing efforts to reduce mortality in commercial fisheries. Additionally, there would be no negative indirect effects to nesting females from the proposed action.

We believe that the incidental lethal and non-lethal takes of leatherback sea turtles associated with the proposed action are not reasonably expected to cause an appreciable reduction in the likelihood of survival and recovery of the species. Although any level of take and mortality can have an adverse effect on the overlying population, we find that the expected level of take from the overall action, including a small number of mortalities, is extremely small when considered together with all impacts considered in the Status of the Species, Baseline and Cumulative Effects sections, including other federally authorized fisheries and foreign fisheries. As stated previously, the proposed action is expected to result in the annual mortality of 0.02 adult females. However, the population that is affected the most is expected to increase (Van Houtan 2011). We believe the populations and therefore the species, will remain large enough to retain the potential to contribute to species recovery. Moreover, the proposed action does not appreciably impede progress on carrying out any aspect of the recovery program or achieving the overall recovery strategy, discussed above.

To summarize, when considering the effects of the proposed action, together with the status of the listed species, the environmental baseline, and the cumulative effects, we believe that the lethal and non-lethal takes of leatherback sea turtles associated with the proposed action are not expected to appreciably reduce the reproduction, numbers, or distribution of the western or eastern Pacific leatherback populations, and thus the leatherback sea turtle as a species. The proposed action is not expected to cause an appreciable reduction in the likelihood of both the survival and recovery of the leatherback sea turtle in the wild.

9.6 **Olive Ridley Turtles**

As discussed in the olive ridley section of the Status of Listed Species (Section 5.2.6), nesting of eastern Pacific olive ridleys steadily increased from 1991 to present up to over 1 million nests annually. The western Pacific olive ridley population is a smaller, widely-scattered population with the largest nesting occurring in the Indian ocean; an estimate of 100,000 turtles nested in 2012 in Orissa (IOSEA 2013).

As discussed in the olive ridley section of the Environmental Baseline (Section 6.6), hundreds of juvenile and adult eastern Pacific olive ridley mortalities may be occurring annually due to longline fishery interactions within the action area alone. Thus, total fishery-related mortality of the eastern Pacific olive ridley population is likely at least several hundred adults annually and a fraction of that may be from the western Pacific population.

As described in the olive ridley section of the Effects of the Action (Section 7.6), if we assume that the proposed action will result in approximately 1,305 trips, with 18,592 sets and 46,117,532 hooks annually, then that level of effort will result in 33 olive ridley interactions annually (McCracken 2014b), which will result in approximately 32 juvenile or adult mortalities annually. Twenty-four are expected to be from the larger eastern Pacific population and 8 from the western Pacific population. The eastern Pacific population has at least one million adult nesting females (Table 5). If we assume that fifty percent of the adults killed are female the impact would be 0.0000013 percent of the adult female population that would be affected. This level of impact is extremely small. The western Pacific population has at least 33,500 adult nesting females (Table 5). If we assume that fifty percent of the adults killed (8) are female the impact would be 0.012 percent of the adult female population that would be affected. This level of impact is extremely small. Therefore risk to both populations from the proposed action is considered negligible.

As discussed in the Cumulative Effects section (Section 8), effects to this species are likely to occur as a result of worsening climate change, and any increase in fishing, ship traffic, and other actions described in the Environmental Baseline section. Such effects could include worsening of the climate change effects described in Sections 5 and 6, as well as an increase effects resulting from fishing gear interactions with this species. In addition, any increases in marine debris could also increase entanglements. Global climate change is expected to continue and therefore may impact sea turtles and their habitat in the future. As discussed in this opinion, rising temperatures at nesting beaches may have negative consequences for nesting females and developing embryos. While olive ridley nesting does not take place inside the action area, turtles that occur in the action area come from nesting aggregations that may be affected by impacts at their nesting beaches of origin throughout the Pacific, although changes will likely not be uniform or predictable. Olive ridleys in the east Pacific Ocean are seemingly adaptable to fluctuating environmental conditions. They possess the ability to shift from an unproductive habitat to one where the waters are biologically productive, which may minimize the impacts of climate change (Plotkin 1994 and 2010 in NMFS and USFWS 2014).

As also discussed in the Cumulative Effects section of this opinion, climate change may impact aquatic aspects of sea turtle biology and ecology, including foraging habitats and prey resources, phenology, and migration. As discussed earlier in this opinion, although there is much speculation on the potential impacts of climate change to species and ecosystems, there are multiple layers of uncertainty associated with these analyses and the effects of climate change will not be globally uniform. In particular, there is no comprehensive assessment of the potential impacts of climate change within the action area or specific to sea turtles that may be within the action area. In addition to the uncertainty of the rate, magnitude, and distribution of future climate change and its associated impacts on temporal and spatial scales, the adaptability of species and ecosystems are also unknown. Implications of climate change at the population level are a key area of uncertainty and one of active research and cannot currently be reliably quantified in terms of actual mortalities resulting from climate change impacts over any time scale. Nor can they be qualitatively described or predicted in such a way as could be more meaningfully evaluated in the context of this biological opinion.

We considered to what extent the effects of the action affect survival and recovery of the olive ridley sea turtle. The NMFS and USFWS' ESA Section 7 Handbook (USFWS and NMFS 1998) provides further definition for *survival* and *recovery*, as they apply to the ESA's jeopardy standard (please refer to the humpback whale discussion of this section for definitions).

The NMFS and USFWS (1998d) olive ridley 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; foraging populations are statistically significantly increasing at several key foraging grounds within each stock region; all females estimated to nest annually (FENA) at "source beaches" are either stable or increasing for over 10 years; a management plan based on maintaining sustained populations for turtles is in effect; international agreements are in place to protect shared stocks.

As discussed above, the anticipated deaths resulting from the continued authorization of the deep-set fishery results in the removal of approximately 32 turtles (either sex) annually. Viewed within the context of the Status of the Species, the Environmental Baseline, the Effects of the Action, and the Cumulative Effects, we believe the annual mortality of 32 olive ridleys caused by the proposed action will not adversely affect the population dynamics of either eastern Pacific or western Pacific olive ridley turtles. Moreover, we do not expect the proposed action to appreciably reduce the reproduction, numbers, or distribution of either population, and thus the species.

We conclude that the incidental take and resulting mortality of olive ridley turtles associated with the proposed action are not reasonably expected to cause an appreciable reduction in the likelihood of survival and recovery of the species. 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 negligible effect on the overall size of the population, and we do not expect it to affect the olive ridleys' ability to meet their lifecycle requirements and to retain the potential for recovery.

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 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) ensuring foraging populations are statistically significantly increasing at several key foraging grounds within each stock region; and 2) all females estimated to nest annually (FENA) at "source beaches" are either stable or increasing for over 10 years.

Although the proposed action would result in the mortality of up to 32 olive ridley turtles annually, as discussed above, this level of mortality would present negligible risk to the species. Since it represents a negligible risk to the species, the proposed action would not prohibit the species from stabilizing or increasing, nor would it prohibit the species from reaching a biologically reasonable FENA based on the goal of maintaining a stable population in perpetuity. The negligible risk to the olive ridley nesting population, which is the source of animals found at foraging grounds, means it would not substantially impair or prohibit increases to olive ridley foraging populations at key foraging grounds. The effects of the action would not prohibit or substantially impair continuing efforts to reduce mortality in commercial fisheries. Additionally, there would be no negative indirect effects to the species from the proposed action.

We believe that the incidental lethal and non-lethal takes of olive ridley sea turtles associated with the proposed action are not reasonably expected to cause an appreciable reduction in the likelihood of survival of the species. Although any level of take and mortality can have an adverse effect on the overlying population, we find that the expected level of take from the overall action, including a small number of mortalities, is small when considered together with all impacts considered in the Status of the Species, Baseline and Cumulative Effects sections, including other federally authorized fisheries and foreign fisheries. Notwithstanding the expected annual mortalities resulting from the proposed action, we believe that the population will remain large enough to retain the potential for recovery. Moreover, we do not believe that the proposed action is reasonably likely to result in an appreciable reduction in the likelihood of recovery of the olive ridley sea turtle. The proposed action does not appreciably impede progress on carrying out any aspect of the recovery program or achieving the overall recovery strategy.

To summarize, when considering the effects of the proposed action, together with the status of the listed species, the environmental baseline, and the cumulative effects, we believe that the lethal and non-lethal takes of olive ridley sea turtles associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of both the survival and recovery of the Pacific olive ridley sea turtle, including the eastern and western subpopulations, in the wild.

9.7 Green Turtles

As discussed in the green turtle section of the Status of Listed Species (Section 5.2.7), nesting of eastern Pacific population, and of the Hawaii component of the Central Pacific population, have increased in the last decade. As described earlier the remaining western Pacific has been described in this opinion as three separate regional aggregations. The central west Pacific Ocean has insufficient information to adequately describe abundance and populations trends for many areas. Limited information suggests a nesting population decrease in some portions of this region like the Marshall Islands, or unknown trends in other areas. The nesting Chichijima, Japan has shown an increasing trend (NMFS and USFWS in Press). In the Southwest Pacific Ocean 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 interannual 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). In the Central south Pacific green turtle temporal population trends are poorly understood, with not even a single nesting site having five contiguous years of standardized monitoring that span entire nesting seasons. Partial and inconsistent monitoring from the largest nesting site in this aggregation, 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 percent of the population abundance at Scilly Atoll alone. Nesting abundance is reported to be stable to increasing at Tongareva Atoll (White and Galbraith 2013).

As discussed in the green turtle section of the Environmental Baseline (Section 6.7), several dozen green turtles are likely to be killed annually by longlining in the action area alone. Thus, total fishery-related mortality of green turtles in the Pacific Ocean is likely a few hundred annually, with the majority belonging to the eastern Pacific population. In addition, up to several dozen green turtles from the Hawaii component of the central Pacific population are killed annually by nearshore activities such as fishing and boat collisions within the action area, although since 1982 only two turtles have been observed stranded as a result of boat collisions.

As described in the green turtle section of the Effects of the Action (Section 7.7), if we assume that the proposed action will result in approximately 1,305 trips, with 18,592 sets and 46,117,532 hooks annually, then that level of effort is expected to result in three green turtle interactions annually. These interactions are expected to include two juvenile or sub-adult mortalities annually from the eastern Pacific aggregation and one mortality from the western Pacific or the Hawaii aggregations.

The two annual mortalities from the Eastern Pacific nesting aggregation are expected to be from any age or sex. In order to analyze the impact to the adult female population, we assume that they are adult and have a 50 percent chance of being female. Therefore, one mortality from

20,112 nesting females represents 0.005 percent of the population (1/ 20,112 * 100= 0.005 percent). This represents an insignificant fraction of the overall nesting population.

There could be one annual mortality from the Hawaii nesting aggregation, of any age or sex. In order to analyze the impact to the adult female population, we assume that it is an adult and has a 50 percent chance of being a female. Therefore a 0.5 mortality from 3,846 nesting females represents 0.013 percent of the population (0.5/3,846 *100=0.013). This represents an insignificant fraction of the overall nesting population, and represents the maximum impact to the population since we anticipate up to one mortality from one of four different nesting aggregations, which includes Hawaii and the three aggregations described for the Western Pacific.

There could be one mortality from the central west Pacific Ocean aggregation annually. This one mortality is from any age or sex. In order to analyze the impact to the adult female population we assume that it is an adult and has a 50 percent chance of being a female. Therefore a 0.5 mortality from 6,518 nesting females represents 0.0077 percent of the population (0.5/6,518 *100=0.0077). This level is an insignificant fraction of the overall nesting population. This estimate represents the maximum impact to the population since we anticipate up to one mortality from one of four different nesting aggregations, which includes Hawaii and the three aggregations described for the Western Pacific.

There could be one mortality from southwest Pacific Ocean nesting aggregation annually. This one mortality is from any age or sex. In order to analyze the impact to the adult female population we assume that it is an adult and has a 50 percent chance of being a female. Therefore a 0.5 mortality from 83,058 nesting females represents 0.0006 percent of the population (0.5/83,058 * 100=0.0006). This level is an insignificant fraction of the overall nesting population. This estimate represents the maximum impact to the population since we anticipate up to one mortality from one of four different nesting aggregations, which includes Hawaii and the three aggregations described for the Western Pacific.

There could be one mortality from the central south Pacific nesting aggregation annually. This one mortality is from any age or sex. In order to analyze the impact to the adult female population we assume that it is an adult and has a 50 percent chance of being a female. Therefore a 0.5 mortality from 2,902 nesting females represents 0.017 percent of the population (0.5/2902 *100=0.017). This level is an insignificant fraction of the overall nesting population. This estimate represents the maximum impact to the population since we anticipate up to one mortality from one of four different nesting aggregations, which includes Hawaii and the three aggregations described for the Western Pacific.

As discussed in the Cumulative Effects section (Section 8), effects to this species are likely to occur as a result of worsening climate change, and any increase in fishing, marine debris, and other actions described in the Environmental Baseline section. Such effects could include worsening of the climate change effects described in Sections 5 and 6, as well as an increase effects resulting from fishing gear interactions with this species. In addition, any increases in marine debris could also increase entanglements or ingestion impacts. Global climate change is expected to continue and therefore may impact sea turtles and their habitat in the future. As discussed in this opinion, rising temperatures at nesting beaches may have negative consequences for nesting females and developing embryos. Turtles that occur in the action area come from nesting aggregations that may be affected by impacts at their nesting beaches of origin throughout the Pacific, although changes will likely not be uniform or predictable. As also discussed in the Cumulative Effects section of this opinion, climate change may impact aquatic aspects of sea turtle biology and ecology, including foraging habitats and prey resources, phenology, and migration. Although there is much speculation on the potential impacts of anthropogenic climate change to species and ecosystems, there are multiple layers of uncertainty associated with these analyses and the effects of climate change will not be globally uniform. In particular, there is no comprehensive assessment of the potential impacts of climate change within the action area or specific to sea turtles that may be within the action area. In addition to the uncertainty of the rate, magnitude, and distribution of future climate change and its associated impacts on temporal and spatial scales, the adaptability of species and ecosystems are also unknown. Implications of climate change at the population level are a key area of uncertainty and one of active research and cannot currently be reliably quantified in terms of actual mortalities resulting from climate change impacts over any time scale. Nor can they be qualitatively described or predicted in such a way as could be more meaningfully evaluated in the context of this biological opinion.

We considered to what extent the effects of the action affect survival and recovery of the green sea turtle. The NMFS and USFWS' ESA Section 7 Handbook (USFWS and NMFS 1998) provides further definition for *survival* and *recovery*, as they apply to the ESA's jeopardy standard (please refer to the humpback whale discussion of this section for definitions).

The NMFS and USFWS (1998e) Green Turtle and NMFS and USFWS (1998a) East Pacific Green Turtle recovery plans contain 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 (FENA) 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 to maintain sustained populations of turtles is in place; and international agreements are in place to protect shared stocks.

As discussed above, the anticipated mortalities resulting from the continued authorization of the deep-set fishery results in the removal of approximately two juvenile or sub-adult turtles (either sex) annually, from the eastern Pacific population, and one (juvenile or sub-adult) from either the western or central Pacific populations. Viewed within the context of the Status of the Species, the Environmental Baseline, the Effects of the Action, and the Cumulative Effects, we believe the annual mortality of three green sea turtles caused by the proposed action is insufficient to adversely affect the population dynamics of the eastern Pacific, central Pacific, or western Pacific green sea turtles and therefore the species. We do not expect the proposed action to reduce the reproduction, numbers, or distribution of the species.

We conclude that the incidental take and resulting mortality of green turtles associated with the direct effects of the proposed action are not reasonably expected to cause an appreciable reduction in the likelihood of survival or recovery of the species. We expect the overall populations to remain large enough to maintain genetic heterogeneity, broad demographic representation, and successful reproduction. The direct effect of the proposed action will have a small effect on the overall size of the populations and therefore the species, and we do not expect it to affect the green turtles' ability to meet their lifecycle requirements and to retain the potential for recovery.

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 one 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 (FENA) 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 fisheries.

The ESA allows for incidental take of species resulting from otherwise lawful activities (such as the proposed action), provided that such take does not result in jeopardy, and the impact of such take is minimized to the extent practicable. While the direct effects of the proposed action would result in some incidental take of this species by the U.S. fishery, take would be subject to mitigation measures to reduce its impact. We have applied the post-release mortality criteria conservatively to ensure that sea turtles that are likely to be seriously injured by capture in the fisheries are counted as lethal takes. The anticipated non-lethal takes are not expected to impact the reproductive potential, fitness, or growth of any of the incidentally caught sea turtles because they will be released unharmed shortly after capture, or released with only minor injuries from which they are expected to recover. Individual takes may occur anywhere in the action area and turtles would be released within the general area where they are caught.

Although the proposed action would result in the mortality of up to three green sea turtles annually, as discussed above, this level of mortality would present negligible additional risk to the species. Since it represents a negligible risk to the species, the proposed action would not prohibit the species from stabilizing or increasing, nor would it prohibit the species from reaching a biologically reasonable FENA based on the goal of maintaining a stable population in perpetuity. The negligible potential risk to the green nesting population, which is the source of animals found at foraging grounds, means it would not substantially impair or prohibit increases to green sea turtle foraging populations at key foraging grounds. The effects of the action would not prohibit or substantially impair continuing efforts to reduce mortality in commercial fisheries. Additionally, there would be no negative indirect effects to the species from the proposed action.

We believe that the incidental lethal and non-lethal takes of green sea turtles associated with the proposed action are not reasonably expected to cause an appreciable reduction in the likelihood of survival of the species. Although any level of take and mortality can have an adverse effect on the overlying population, we find that the expected level of take from the overall action, including a small number of mortalities, is extremely small when considered together with all impacts considered in the Status of the Species, Baseline and Cumulative Effects sections, including other federally authorized fisheries and foreign fisheries. Moreover, we do not believe that the proposed action is reasonably likely to result in an appreciable reduction in the likelihood of recovery of the green sea turtle. The proposed action does not appreciably impede progress on carrying out any aspect of the recovery program or achieving the overall recovery strategy.

To summarize, when considering the effects of the proposed action, together with the status of the listed species, the environmental baseline, and the cumulative effects, we believe that the lethal and non-lethal takes of green sea turtles associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of both the survival and recovery of the eastern Pacific, western Pacific, or central Pacific green sea turtle nesting aggregations individually, or the species as a whole in the wild.

9.8 Indo-west Pacific scalloped hammerhead shark DPS

As discussed in the hammerhead section of the Status of Listed Species the population size of the Indo-west Pacific DPS is unknown but is thought to have localized depletion in some parts of the range including off South Africa and Australia. These declines are attributed to overutilization by industrial/commercial fisheries, artisanal fisheries, and illegal fishing of the scalloped hammerhead shark as the most serious threats to the persistence of this DPS. Current effective population sizes are estimated to be at least 11,280 from studies in several areas around the region.

As discussed in the scalloped hammerhead section of the Environmental Baseline (Section 6.8), a very small number of scalloped hammerheads are likely to be killed annually by the deep-set fishery in the action area. However we do not have estimates of takes from non-U.S. fisheries that operate in the action area. Within the action area the Hawaii-based deep-set longline fishery catches scalloped Hammerhead sharks as bycatch at very low levels. Since 1995, approximately 16 scalloped hammerheads were caught by the deep-set fishery in the range of the Indo-west Pacific DPS. Since 2004, there have been 3 observed scalloped hammerhead sharks caught in the deep-set fishery in the Indo-west pacific DPS (PIRO Observer Program, unpublished data). Based on the three observed interactions and the observer coverage levels in those years, we estimate that the total catch of scalloped hammerheads from the Indo-Pacific DPS was approximately 14, which is equivalent to an annual average of 2 (rounded up from 1.4) for the 2004-2014 time period when they were observed (Table 17). Therefore, we estimate that there are no more than 2 scalloped hammerheads caught annually by the Hawaii deep-set longline fishery. As discussed earlier the historic level of reporting by most fisheries was minimal, and where it did exist it was not precise enough to determine species of hammerhead, or even shark in most cases. Therefore we do not have an estimate of take in the action area by non U.S. vessels.

As described in the Scalloped hammerhead shark section of the Effects of the action, response to the predicted exposure (two interactions annually) from the proposed action can be converted to the annual number of estimated mortalities resulting from this exposure. We estimate the response to be the mortality of one (rounded from 0.72) annually. This level of mortality represents approximately 0.009 percent of the population, which is an insignificant level. In addition, this is based on a likely underestimate of the effective population size in the region.

As discussed in the cumulative effects section, an increase in fishing could cause the greater impacts on the DPS since overutilization by fisheries is considered the greatest threat to the species. However there has been a trend recently for greater protection of sharks throughout the region through shark sanctuaries, prohibited shark fishing, and the prohibition of shark fins at all official reception dinners in China. In addition the Shark Conservation Act of 2010 requires all fisherman harvesting sharks to land the carcass intact.

We conclude that the incidental take and resulting mortality of scalloped hammerhead sharks associated with the direct effects of the proposed action are not reasonably expected to cause an appreciable reduction in the likelihood of survival or recovery of the species. We expect the overall DPS to remain large enough to maintain genetic heterogeneity, broad demographic representation, and successful reproduction. The direct effect of the proposed action will have a small effect on the overall size of the DPS, and we do not expect it to affect the scalloped hammerhead sharks' ability to meet their lifecycle requirements and to retain the potential for recovery.

Although the proposed action would result in the mortality of one scalloped hammerhead shark annually, as discussed above, this level of mortality would present negligible additional risk to the species. Since it represents a negligible risk to the species, the proposed action would not prohibit the species from stabilizing or increasing.

We believe that the incidental lethal and non-lethal takes of scalloped hammerhead sharks associated with the proposed action are not reasonably expected to cause an appreciable reduction in the likelihood of survival of the species. Although any level of take and mortality can have an adverse effect on the overlying population, we find that the expected level of take from the overall action, including a small number of mortalities, is extremely small when considered together with all impacts considered in the Status of the Species, Baseline and Cumulative Effects sections, including other federally authorized fisheries and foreign fisheries. We believe that the population will remain large enough to retain the potential for recovery.

To summarize, when considering the effects of the proposed action, together with the status of the listed species, the environmental baseline, and the cumulative effects, we believe that the lethal and non-lethal takes of scalloped hammerhead sharks associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of both the survival and recovery of the Indo-western Pacific scalloped hammerhead shark DPS.

10 Conclusion

The purpose of this biological opinion is to determine if the proposed action is likely to jeopardize the continued existence of listed species (i.e., jeopardy determination) or result in destruction or adverse modification of designated critical habitat. "Jeopardize the continued existence of" means "to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (50 CFR 402.02). After reviewing the current status of ESA-listed humpback whales, sperm whales, MHI insular false killer whale DPS, North Pacific loggerhead DPS, leatherback sea turtles, olive ridley sea turtles, green sea turtles, and the Indo-west Pacific scalloped hammerhead DPS, the environmental baseline for the action area, the effects of the proposed action, and the cumulative effects, it is NMFS' biological opinion that the proposed action is not likely to jeopardize the continued existence of these eight species, and since no critical habitat will be adversely affected the action is not likely to destroy or adversely modify designated critical habitat.

11 Conservation Recommendations

Section 7(a)(1) of the ESA directs Federal agencies to utilize their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or develop information.

The following conservation recommendations are provided pursuant to section 7(a)(1) of the ESA for developing management policies and regulations, and to encourage multilateral research efforts which would help in reducing adverse impacts to listed species in the Pacific Ocean.

1. NMFS should continue to promote reduction of sea turtle bycatch in Pacific fisheries by supporting:
    a. The Inter-American Convention for the Protection and Conservation of Sea Turtles;
    b. The WCPFC and IATTC sea turtle conservation and management measures for commercial longline fisheries operating in the Pacific;
    c. The wide dissemination and implementation of NMFS Sea Turtle Handling Guidelines that increase post-hooking turtle 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 and eastern Pacific;
    e. Continuation of ecological, habitat use, and genetics studies of all sea turtles occurring in foraging and migratory habitats in the Pacific, continue monitoring impacts through stranding programs, and promote mitigation studies and handling measures for fisheries operating in these waters, and;
    f. Continuation of bycatch reduction efforts in the western and eastern Pacific to reduce commercial and artisanal fishery impacts (e.g., mitigation of Japan poundnets and other fisheries operating in the South China and Sulu Sulawesi Seas, and bycatch reduction in coastal gillnet and trawl fisheries).

2. NMFS should continue to encourage, support and work with Regional partners to implement long-term sea turtle monitoring, conservation, and recovery programs at important nesting habitats.

3. NMFS should continue to investigate long term climate variability and its impacts to turtle populations.

4. NMFS should continue to promote tools, like Turtle Watch, to help fishermen avoid longline fishery interactions.

5. NMFS should continue to investigate and promote programs to reduce fishery interactions with leatherback turtles in the action area.

12 Reinitiation Notice

This concludes formal consultation on the continued operation of the Hawaii deep-set longline tuna 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.

13 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. “Take” is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, collect, or attempt to engage in any such conduct. “Incidental take” is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. 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)).

13.1 MMPA Authorization

A marine mammal species or population stock that is listed as threatened or endangered under the ESA is, by definition, depleted under the MMPA. The ESA allows incidental 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 MMPA, 16 U.S.C. 1361 *et seq.*, has provisions for 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 ESA, 16 U.S.C. 1531 *et seq.*, by U.S.

vessels and those vessels which have valid fishing permits issued by the Secretary in accordance with section 204(b) of the Magnuson-Stevens Fishery Conservation and Management Act, 16 U.S.C. 1824(b), for a period of up to three years. NMFS may issue the authorization to take ESA-listed marine mammals incidental to these commercial fisheries only after the agency has determined, after notice and opportunity for public comment, that:

(1) the incidental mortality and serious injury from commercial fisheries will have a negligible impact on the affected species or stock;

(2) a recovery plan has been developed or is being developed for such species or stock under the ESA; and

(3) where required under section 118 of the MMPA, a monitoring program has been established, vessels engaged in such fisheries are registered in accordance with section 118 of the MMPA, and a take reduction plan has been developed or is being developed for such species or stock.

A draft analysis was completed by NMFS PRD on June 12, 2014 and a Notice for a proposed permit for a period of three years to authorize the incidental, but not intentional, taking of individuals of the Central North Pacific stock of endangered humpback whales, the Hawaii stock of the endangered sperm whales, and the endangered Main Hawaiian Islands insular false killer whale DPS by the Hawaii-based longline fisheries was issued (NMFS 2014e, f). The proposed authorization (79 FR 33726) was based on determinations that mortality and serious injury incidental to commercial fisheries in Hawaii will have a negligible impact for purposes of issuing a permit under section 101(a)(5)(E) of the MMPA for the CNP stock of humpback whales, the Hawaii stock of sperm whales, and the MHI IFKW stock. Recovery plans have been developed for humpback and sperm whales, and one has been initiated for the MHI IFKW DPS (78 FR 60850 October 2, 2013). A monitoring plan is in place and a take reduction plan is in place for the MHI IFKW (77 FR 71260). The CNP stock of humpback whales and Hawaii stock of sperm whales have been designated as strategic because they are ESA-listed (MMPA section 3(19)(C)) and not because direct human-caused mortality exceeds PBR (MMPA section 3(19)(A)). As determined herein, M&SI for both stocks is currently low and is not expected to adversely affect either stock through effects on annual rates of recruitment or survival. Given these factors and NMFS' priorities, developing a TRP for these stocks will be deferred under section 118 as other stocks/fisheries are a higher priority for any available funding for developing new TRPs (79 FR 33726).

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.

13.2 Anticipated Amount or Extent of Incidental Take

NMFS anticipates the following incidental takes may occur as a result of the continued operation of the Hawaii deep-set longline fishery with approximately 1,305 trips with 18,592 sets and 46,117,532 hooks annually. The annual numbers of interactions and mortalities expected to result from implementation of the proposed action are shown for a 3-year period in Table 18 below (i.e., 3-year ITSs). The interactions are a form of take which can result in mortalities; the interactions and mortalities composes the total amount of take that is anticipated under the proposed action. Annual take estimates can have high variability because of natural variation. It is unlikely that all species evaluated in this opinion will be consistently impacted year after year by the fishery. The interactions and mortalities in Table 18 have been calculated based on observed interaction rates (see Section 7) and estimated post-hooking mortality rates of sea turtles and M&SI determinations for marine mammals. Annual equivalent adult female mortalities (AFMs) are also shown for loggerheads and leatherbacks, because they were the basis for the population assessment (Van Houtan 2013, 2014). Use of a 3-year approach will allow for an accurate assessment of the deep-set fishery's impacts on protected species, while avoiding unnecessary reinitiation in response to short-term variability in interactions. An incidental take statement is not provided for those species identified in section 5.1 as not likely to be adversely affected by the proposed action, since NMFS has determined that incidental take of those species is not reasonably certain to occur. *See Arizona Cattle Growers v. USFWS*, 273 F.3d 1229 (9th Cir. 2001).

Table 18. The total number of marine mammal, turtle, and scalloped hammerhead shark interactions (i.e. take) expected from the proposed action over a three –year period. Also shown are the total mortalities (i.e. take) (males and females, adults and juveniles) expected to result from this number of interactions, and the annual equivalent adult female mortalities (AFMs) for turtles. Observed takes are extrapolated to total interactions (takes) in order to monitor the ITS by multiplying the number of confirmed observed takes by an expansion factor based on current observer coverage. Example calculation for humpbacks at 20 percent coverage: 100 ÷ observer coverage= expansion factor [100/20=5]. Expansion factor * number observed=total takes [5*1=5].

Species	3-Year		
	Interactions	Total mortalities	Equivalent AFMs
Humpback whales	6	3	N/A
Sperm whales	9	6	N/A
*MHI IFKW	1	0.74[30]	N/A
N. Pacific loggerhead turtles	9	9	0.54
Leatherback turtles	72	27	0.60
Olive ridley turtles	99	96	N/A
Green turtles	9	9	N/A
**Indo-West Pacific Scalloped Hammerhead shark	6	3	N/A

* Current protocols require NMFS to prorate false killer whale interactions to one of two stocks (i.e., MHI insular or HI pelagic stocks) determined by the location of the interaction and the probability it is a MHI IFKW, based on distance from shore and stock densities. Future models are expected to include a third stock (i.e., the NWHI FKW) in the proration scheme in the areas where they overlap. Since NMFS will not know where the interactions will occur in the overlap area (s) and therefore cannot determine what the probability will be that it is a MHI IFKW, we took an average of the probabilities in the overlap area that is open to longline fishing. The mean probability that an interaction will involve a MHI IFKW in the overlap area is 0.15 (McCracken pers comm. May 2014). One observed false killer whale would equal 5 total interactions based on 20 percent observer coverage. Therefore the total amount calculated to be a MHI IFKW would be 5 * 0.15=0.75. Accordingly, one observed MHI IFKW interaction would be less than the one that we predict over a three year time frame. A second observation in the overlap area within a three year time period would exceed the level predicted and would trigger reiniation of consultation and NMFS would convene the TRT to evaluate alternatives to prevent future takes of MHI IFKW.

**An ITS is not required to provide protective coverage for the Indo-West Pacific scalloped hammerhead shark DPS because there are no take prohibitions under ESA section 4(d) for this DPS. 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 so the action does not jeopardize the species if the level of take analyzed in the biological opinion is exceeded.

13.3 Impact of the Take

In the accompanying biological opinion, NMFS determined that the level of incidental take anticipated from the proposed action is not likely to jeopardize the humpback whale, sperm whale, MHI IFKW DPS, North Pacific loggerhead DPS, leatherback turtle, green turtle, olive ridley turtle, or the Indo-west pacific scalloped hammerhead shark DPS.

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[30] The anticipated annual M&SI is 0.2460, therefore we anticipate that there will be one mortality every four years (0.2460*4=0.981, round to one).

13.4 Reasonable and Prudent Measures

Section 7(b)(4) of the ESA requires that when an agency is found to comply with section 7(a)(2) of the ESA and the proposed action may incidentally take individuals of listed species, NMFS will issue a statement specifying the impact of any incidental taking. It also states that reasonable and prudent measures necessary to minimize impacts, and terms and conditions to implement those measures be provided and must be followed to minimize those impacts. Only incidental taking by the Federal agency or applicant that complies with the specified terms and conditions is authorized.

The incidental take expected to result from the proposed action is shown in Table 18 above for each marine mammal and sea turtle species.

NMFS has determined that the following reasonable and prudent measures, as implemented by the terms and conditions (identified in Section 13.5), are necessary and appropriate to minimize the impacts of the deep-set longline fishery, as described in the proposed action, on sea turtles and marine mammals, and to monitor the level and nature of any incidental takes. These measures are non-discretionary--they must be undertaken by NMFS for the exemption in ESA section 7(o)(2) to apply.

1. NMFS shall collect data on the capture, injury, and mortality caused by the deep-set longline fishery, and shall also collect basic life-history information, as available.

2. NMFS shall require that sea turtles 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, as practicable and in consideration of best practices for safe vessel and fishing operations.

3. NMFS 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.

4. NMFS shall require sea turtles that are dead when brought on board a vessel or that do not resuscitate be disposed of at sea unless NMFS requests retention of the carcass for sea turtle research, as practicable and in consideration of best practices for safe vessel and fishing operations.

5. NMFS shall require that all Hawaii longline vessels comply with all gear requirements and handling protocols for marine mammals implemented through the FKW TRP and any future changes implemented by the TRP.

6. NMFS shall continue to monitor FKWTRP effectiveness in reducing MHI IFKW M&SI, and will convene the False Killer Whale Take Reduction Team to provide recommendations on additional management measures that may be necessary and appropriate to avoid significant adverse impacts to MHI IFKW.

7. NMFS shall require that all activities comply with the authorization issued under MMPA section 101 (a)(5)(E).

13.5 Terms and Conditions

NMFS shall undertake and comply with the following terms and conditions to implement the reasonable and prudent measures identified in Section 13.4 above. These terms and conditions are non-discretionary, and if NMFS fails to adhere to 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:

    1A. *Observers*. NMFS shall maintain observer coverage at rates that have been determined to be statistically reliable for estimating protected species interaction rates onboard Hawaii deep-set longline vessels.

    1B. *Data Collection*. As practicable and in consideration of best practices for safe vessel and fishing operations, observers shall collect standardized information regarding the incidental capture, injury, and mortality of sea turtles for each interactions by species, gear, and set information, as well as the presence or absence of tags on the turtles. Observers shall place tags on any untagged turtles that are brought aboard a vessel. Observers shall also collect life-history information on sea turtles incidentally caught by the deep-set fishery, including measurements, (including direct measure or visual estimates of tail length), condition, skin biopsy samples, and estimated length of gear left on the turtle 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.

    1C. *Information Dissemination*. NMFS shall disseminate quarterly, summaries of the data collected by observers to the NMFS Assistant Regional Administrators of Protected Resources and Sustainable Fisheries in PIR, as well as the NMFS Sea Turtle Coordinators in PIR, West Coast Region and Headquarters.

2. The following terms and conditions implement Reasonable and Prudent Measure No. 2:

    2A. NMFS shall continue to require and conduct protected species workshops for owners and operators of vessels registered for use with Hawaii limited entry longline fishing permits, to educate vessel owners and operators in handling and resuscitation techniques to minimize injury and promote survival of hooked or entangled sea turtles, as specified in 50 CFR 665. The workshops shall include information on sea turtle biology and ways to avoid and minimize sea turtle impacts to promote sea turtle protection and conservation, including disseminating new scientific information such as TurtleWatch for loggerhead turtles.

    2B. NMFS shall continue to train observers about sea turtle biology and techniques for proper handling, dehooking, and resuscitation.

2C. NMFS shall require that deep-set longline fishermen remove hooks from turtles as quickly and carefully as possible to avoid injuring or killing the turtle, as practicable and in consideration of best practices for safe vessel and fishing operations. NMFS shall require that each Hawaii deep-set longline vessel carry a line clipper or cutter to cut the line as close to the hook as practicable and remove as much line as possible prior to releasing the turtle in the event all of the fishing gear cannot be removed (e.g., the hook is deeply ingested or the animal is too large to bring aboard).

2D. NMFS shall require that each Hawaii deep-set longline vessel carry a dip net to hoist a sea turtle onto the deck to facilitate hook removal. If the vessel is too small to carry a dipnet, sea turtles must be eased onto the deck by grasping its carapace or flippers, to facilitate the removal of the hook. Any sea turtle 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.

2E. NMFS shall require each deep-set longline 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 a turtle.

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

3A. NMFS shall require that deep-set longline 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 may perform resuscitation techniques on comatose sea turtles if the observer is available.

4. The following terms and conditions implement Reasonable and Prudent Measure No. 4:

4A. NMFS 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.

5. The following terms and conditions implement Reasonable and Prudent Measure No. 5:

5A. NMFS shall continue to provide information on marine mammal interaction and mitigation techniques at the required protected species workshops; require a NMFS-approved marine mammal handling and release informational placard to be posted onboard all longline vessels; require the captain of the longline vessel to supervise the handling and release of any hooked or entangled marine mammal; and require a NMFS-approved placard that instructs the vessel crew to notify the captain in the event of a marine mammal interaction be posted onboard all Hawaii longline vessels (77 FR 71260).

6. The following terms and conditions implement Reasonable and Prudent Measure No. 6:

   6A. During any three year period, NMFS shall convene the False Killer Whale Take Reduction Team following a second observed take of a FKW within the overlap zone to receive recommendations for possible emergency measures to protect MHI IFKW's.

7. The following terms and conditions implement Reasonable and Prudent Measure No. 7:

   7A. All activities must comply with the authorization issued under MMPA section 101(a)(5)(E).

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